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Dear participants,

Welcome to our conference "Risk Assessment in European River Basins - State of the Art and Future Challenges". During this event, organised within the frame of the Integrated Project MODELKEY (511237 GOCE, www.modelkey.org) and the Coordination Action RISKBASE (036938 GOCE, www.riskbase.info), leading scientists and European stakeholders will present keynotes on latest developments and challenges associated with the assessment of risks to European river basins.

Freshwater ecosystems provide numerous goods and services, like e.g. drinking water supply and climate regulation that are at risk of hydromorphological changes, eutrophication and toxic pollution under global change. Thus, one of the major challenges for water managers, policy makers and scientists is to assess and to mitigate these risks. The European Water Framework Directive states that "water is an inherited good that has to be protected and used in a sustainable way" and with that the scope of river basin management was defined.

This understanding resulted in funding several research projects with focus on different aspects of risk assessment in aquatic ecosystems. The Coordination Action RISKBASE aims to review and synthesise the outcome of these projects, and other initiatives, related to integrated risk assessment-based management of the water/sediment/soil system at the river basin scale.

The conference reflects this concept, i.e. the keynote and poster presentations focus on

- ecosystem goods and services in risk-based management,
- water regulation at risk under global change,
- groundwater ecosystems and drinking water supply at risk,
- hydromorphological changes and risks to biodiversity,
- eutrophication risks to biodiversity,
- risks of environmental pollutants to ecosystem and human health,
- integrated risk assessment on a basin scale,

particularly in terms of practical implementation for river basin management in order to bridge the gap between science and management practice, as you will experience during the conference and in the proceedings.

We very much hope that you will have a stimulating and rewarding meeting with many comprehensive discussions and wish you a pleasant stay in the beautiful city of Leipzig.

Werner Brack, Peter von der Ohe and Michaela Hein
Organising Committee
### Conference Programme

#### Sunday, 11 November 2007

from 19:00  Welcome Reception and Registration at KUBUS

#### Monday, 12 November 2007

from 8:00  Registration

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<td>09:15</td>
<td>Welcome</td>
<td>Georg Teutsch &amp; Werner Brack, UFZ, Germany</td>
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<tr>
<td>09:45</td>
<td>Towards Risk-Based Management of European River Basins (EC FP6 CA RISKBASE)</td>
<td>Jos Brils, TNO, The Netherlands</td>
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<td>10:10</td>
<td>MODELKEY - Providing integrated tools for the establishment of cause-effect relationships in contaminated aquatic ecosystems</td>
<td>Werner Brack, UFZ, Germany &amp; Jos van Gils, Peter von der Ohe, Mechthild Schmitt-Jansen, Helmut Segner, Dick de Zwart</td>
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<td>10:35</td>
<td>Flood risk assessment</td>
<td>Volker Meyer, UFZ, Germany</td>
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<td>11:00</td>
<td>Climate change impacts on hydrology - State of the art and future challenges</td>
<td>Fred Hattermann, PIK, Germany &amp; Heiko Apel</td>
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<td>11:25</td>
<td>Drought risks in Mediterranean river basins</td>
<td>Ana Iglesias, Universidad Politecnica de Madrid, Spain &amp; Luis Garrote</td>
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<td>12:15</td>
<td>Risk of hydromorphological changes on biodiversity - Case studies from the Baltic Sea region</td>
<td>Seppo Hellsten, Finnish Environment Institute, Finland &amp; Mika Marttunen</td>
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<td>12:40</td>
<td>Towards ecologically successful hydromorphological risk assessment and management in streams</td>
<td>Piet Verdonschot, Alterra, The Netherlands</td>
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<td>13:05</td>
<td>How to assess degradation of freshwater communities due to hydromorphological changes</td>
<td>Nikolai Friberg, Macaulay Land Use Research Institute, UK &amp; Leonard Sandin, Morten L. Pedersen</td>
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#### Water regulation at risk under global change

10:35  Flood risk assessment

11:00  Climate change impacts on hydrology - State of the art and future challenges

11:25  Drought risks in Mediterranean river basins

11:50  BREAK

#### Hydromorphological changes and risks to biodiversity

12:15  Risk of hydromorphological changes on biodiversity - Case studies from the Baltic Sea region

12:40  Towards ecologically successful hydromorphological risk assessment and management in streams

13:05  How to assess degradation of freshwater communities due to hydromorphological changes

13:30  LUNCH BREAK
**Eutrophication risks to biodiversity**

14:30  **Biodiversity and eutrophication assessment**  
Ana Cristina Cardoso, JRC, Italy & Gary Free  

14:55  **Eutrophication risks of rivers and streams under the conditions of climate change**  
Dietrich Borchardt, UFZ, Germany  

15:20  **BREAK**

**Groundwater ecosystems and drinking water supply at risk**

15:45  **Integrated groundwater risk-based management in the context of the WFD**  
Philippe Quevauviller, DG Environment, Belgium  

16:05  **Threshold values and improving risk assessment for groundwater**  
Dietmar Müller, Umweltbundesamt, Austria  

16:30  **Risks to drinking water supply in European river basins**  
Daniel Villesot, Lyonnaise des Eaux, France & Guillaume Stahl

from 17:00  **POSTER SOCIAL**

**Tuesday, 13 November 2007**

**Risks of invasive species to ecosystem goods and services**

9:00  **Biological invasions via European inland waterways: Towards development of the risk assessment tool**  
Vadim E. Panov, St. Petersburg State University, Russia & Boris Alexandrov, Kestutis Arbaciauskas, Michal Grabowski, Rob S.E.W. Leuven, Sergey Mastitksy, Dan Minchin, Stefan Nehring, Sergej Olenin, Momir Paunovic, Vitaliy Semenchenko, Mikhail Son

9:25  **Socio-economic assessment of risks of invasive species to freshwater ecosystem goods and services on a basin scale**  
Rosa Binimelis, ICTA/Universitat Autonoma de Barcelona, Spain & Beatriz Rodriguez-Labajos

**Risk of environmental pollutants to ecosystem and human health**

9:25  **Concepts for aquatic risk assessment of pesticides and ecological protection**  
Theo C.M. Broek, Alterra, The Netherlands

10:15  **Probabilistic risk assessment of pesticides**  
Ralf Schulz, University of Landau, Germany

10:40  **Uncertainty in ecological and human risk assessment: Implications for risk-based management in river basins**  
Ad Ragas, University of Nijmegen, The Netherlands & Rob S.E.W. Leuven

11:05  **BREAK**
11:30  Generic exposure models on a river basin scale - A new way to support the risk assessment for contaminants in European river basins
Jos van Gils, DELFT Hydraulics, The Netherlands & Bert van Hattum, Bernhard Westrich

11:55  Understanding and modelling of pollutant fluxes in the water-sediment-soil-system at catchment scale
Johannes Barth, University of Tübingen, Germany & Alexis Gutierrez, Nicole Baran, Phillipe Négrel, Huub Rijnaarts, Alette Langenhoff, Neslihan Tas, Miriam van Eekert, Hauke Smitd, Peter Grathwohl

12:20  “Chemical activity” and “accessibility” as key parameters for risks due to sediment-associated organic contaminants
Philipp Mayer, NERI, Denmark & Fredrik Reichenberg

12:45  Effects-directed identification of key toxicants
Kevin Thomas, NIVA, Norway & Knut-Erik Tollefsen, Merete Grung, Katherine Langford

13:10  LUNCH BREAK

14:15  Novel analytical tools - Progress on the identification and quantification of hazardous contaminants in complexly contaminated environments
Pim Leonards, Vrije Universiteit Amsterdam, The Netherlands & Timo Hamers, Ewa Skoczynska, Peter Korytar, Marja Lamoree

14:40  Assessment concepts for effects of multiple contaminants with time-variable exposure
Rolf Altenburger, UFZ, Germany

15:05  Higher Tier toxicity studies as the link between laboratory tests and ecosystem health assessment

15:30  How to link risk assessment to current understanding of community-level effects?
Mechthild Schmitt-Jansen, UFZ, Germany

15:55  Estrogenic compounds in Spanish river basins - Implications for human and ecological risk assessment
Miren López de Alda, CSIC, Spain & Damià Barceló

16:20  BREAK

Case studies
16:45  Llobregat case study: How to elucidate the role of environmental factors and toxicants on biological community?
Isabel Muñoz, Universitat de Barcelona, Spain & Damià Barceló, Rikke Brix, Helena Giusch, Miren López de Alda, Julio C. López-Doval, Marta Ricart, Anna M. Romani, Sergi Sabater

17:10  Impact of pollution on biodiversity: From basin to site scale in the Scheldt river basin
Eric de Deckere, University of Antwerp, Belgium & Sebastian Höss, Vicky Leloup, Isabel Muñoz, Claus Orendt, Claudia Schmitt, Chris Van Liefferinge, Georg Wolfram, Patrick Meire
## Risk Assessment in European River Basins - What can we learn from the Danube case study?

**Jaroslav Slobodnik, Environmental Institute, Slovak Republic & Igor Liska**

![Page 6](image)

### Risk assessment in European river basins - What can we learn from the Danube case study?

**Jaroslav Slobodnik, Environmental Institute, Slovak Republic & Igor Liska**

**17:35**

### Risks from contaminated sediments to ecosystem services of a river catchment: Improved approaches and lessons learned from the Elbe case study

**Susanne Heise, TU Hamburg-Harburg, Germany & Ulrich Förstner, Frank Krüger, Martina Baborowski**

![Page 122](image)

**18:00**

### Integrated risk assessment on a basin scale

**Wednesday, 14 November 2007**

#### Integrated risk assessment on a basin scale

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<td>Controlling pollution in river basins - Risk based approaches at different scales</td>
<td>Bob Harris, University of Sheffield, UK</td>
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<td>Managing European sediments: Can we expand our ecological risk assessment paradigms?</td>
<td>Sabine Apitz, SEA Environmental Decisions, UK</td>
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<td>10:40</td>
<td>Break</td>
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<td>11:15</td>
<td>The MODELKEY database - Towards an integrative European risk assessment</td>
<td>Peter C. von der Ohe, UFZ, Germany &amp; Werner Brack</td>
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<td>11:40</td>
<td>New diagnostic and predictive modelling tools for an advanced and integrated evaluation of chemical, ecological, and ecotoxicological monitoring data</td>
<td>Dick de Zwart, RIVM, The Netherlands &amp; Leo Posthuma, Muriel Gevrey, Eric de Deckere</td>
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<td>12:05</td>
<td>Supporting river basin management at different scales: A risk-based DPSIR framework</td>
<td>Antonio Marcomini, Consorzio Venezia Ricerche, Italy &amp; Elena Semenzin, Stefania Gottardo, Andrea Critto</td>
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<td>12:30</td>
<td>Closure</td>
<td>Werner Brack, UFZ, Germany</td>
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Extended Abstracts Speakers
Towards Risk-Based Management of European River Basins  
(EC FP6 CA RISKBASE) 

Jos Brils¹

¹TNO Built Environment and Geosciences, PO Box 80015, 3508 TA Utrecht, The Netherlands, jos.brils@tno.nl, website: www.riskbase.info

Background
In the last decade several EC RTD projects and other major research initiatives have addressed and promoted issues related to risk-assessment based management. Most of these initiatives focus on quality and management aspects of one specific compartment: water, sediment, soil or groundwater. However, all these initiatives stressed the importance of an integrative approach for understanding and managing the multi-compartment system at the river-basin scale. Hence, there was a clear need to bring all soil-water RTD efforts together through a Coordination Action (CA) as a stepping stone towards further policy development and the development of a research agenda related to risk based management of that system. This CA is the FP6 project on Risk Based Management of River Basins (acronym: RISKBASE; EC project No. GOCE 036938).

RISKBASE
In RISKBASE, leading European scientists and representatives of major, European stakeholder groups review and synthesise the outcome of EC RTD Framework Program projects, and other major initiatives, related to integrated risk assessment-based management of the water/sediment/soil system at the river-basin scale. The synthesis will lead to the development of integrated risk assessment-based management approaches enabling the prevention and/or reduction of impacts caused by human activities on that system.

RISKBASE aims to deliver: (1) An overarching concept, generic approach and guiding principles to integrated risk based management of river basins; (2) Recommendations towards evolution and implementation of risk based management operations in national and community policies and towards implementation in management; and (3) A proposal for the European research agenda related to risk based management.

Risk object: ecosystem services and resilience
Logically risk is always connected to an object or area of concern. Within RISKBASE this risk object is defined as the goods and services provided by the biophysical soil-sediment-water system (ecological system), with a specific focus on the resilience of the system. Furthermore, RISKBASE focuses on the goods and services that are directly (or immediately indirectly) affected by rivers, lakes and groundwater. Thus the risk-based management approach to be developed should be helpful in: (1) identifying the resilience thresholds (or early warning indicators for that) and (2) defining measures for increasing the resilience (or decreasing the vulnerability) of river systems. Measures should be aimed at decreasing the likelihood that these thresholds will be crossed and hence to prevent the system moving to a different regime or state (Brils et al., submitted).

First findings
RISKBASE aims for an adaptive management approach where continuous learning (improved social-ecological system understanding) takes a central role (Figure 1). Because no fundamental, river system oriented policies have been developed in Europe so far, existing river-basin management policies are seen as supportive to this adaptive management framework rather than this framework being supportive to the strict implementation of existing policies (Brils et al., submitted).
Fig. 1: RISKBASE (draft) concept for a Risk-Based Management framework of River Basins (Brils et al., submitted). Within RISKBASE the ‘receptor’ is defined as the goods and services provided by the biophysical soil-sediment-water system (ecological system), with a specific focus on the system’s resilience.

**Challenges and future perspective**

It will be a big challenge to bring the framework to an operational level. This may be done by identifying indicators for the ‘resilience state’ of the system and by identifying or (further) developing tools and methods for assessment (measure/monitor and valuate) of the indicators. If this is successful, the insight can then be used to identify and develop practical measures aimed to increase the resilience of river systems and their ecosystem services.

This framework is further developed and tested in practice through river basin case studies, in a series of public workshops, general assemblies and other activities (co)organised by RISKBASE. Further information on these events can be found at the RISKBASE website at: www.riskbase.info

**References**


Jos Brils is working as a senior researcher with TNO as initiator and manager of river-basin management related projects. Jos has a background in ecotoxicology. He is the coordinator of RISKBASE and initiator and coordinator of the FP5 funded European Sediment Network SedNet (www.sednet.org). Current key-topics studied by Jos are: science-policy interfacing, resilience of social/ecological systems and adaptive management approaches.
MODELKEY - Providing integrated tools for the establishment of cause-effect relationships in contaminated aquatic ecosystems

Werner Brack¹, Jos van Gils², Peter von der Ohe¹, Mechthild Schmitt-Jansen¹, Helmut Segner³, Georg Streck¹, Dick de Zwart⁴

¹Helmholtz Centre for Environmental Research – UFZ, Permoserstrasse 15, 04318 Leipzig, Germany, werner.brack@ufz.de
²WL | Delft Hydraulics, P.O. Box 177, 2600 MH, Delft, The Netherlands
³University of Berne, Laenggass-Straße 122, 3012 Berne, Switzerland
⁴National Institute for Public Health and the Environment, P.O. Box 1, 3720 BA Bilthoven, The Netherlands

Background

Water is an inherited good that has to be protected and used in a sustainable way. Based on this understanding the EU Water Framework Directive demands for a good ecological status of European waters by 2015. This is a reasonable but challenging task for European water managers. In order to address their needs to identify, rank and mitigate driving forces for insufficient ecological quality a new Integrated Project called MODELKEY (Models for Assessing and Forecasting the Impact of Environmental Key Pollutants on Marine and Freshwater Ecosystems and Biodiversity (http://www.modelkey.org/)) was granted by the EU and started in 2005 [1]. 26 European institutes from 14 different countries under the co-ordination of the Helmholtz Centre for Environmental Research – UFZ in Leipzig (Germany) were brought together for 5 years to develop and integrate advanced analytical and bioanalytical tools as well as interlinked diagnostic and predictive models to unravel cause-effect relationships between contamination and ecological quality and to assess risks to aquatic ecosystems. This goal is addressed exemplarily in the three pilot river basins Llobregat (Spain), Scheldt (France, Belgium and The Netherlands), and Elbe (Czech Republic and Germany).

Thesis 1: Providing weight of evidence for cause-effect relationships is a prerequisite for efficient river basin management

Fig. 1: Results of the evaluation of the ecological status for surface waters in Germany (Water Framework Directive Scoreboard 2004/2005).

The EU Water Framework Directive (WFD) demands for a good ecological status for European rivers by 2015. For many river basins this goal is uncertain or unlikely to be achieved as demonstrated for Germany (Fig. 1). While morphological changes are assumed as a major cause for insufficient ecological status a rigorous assessment of other possible driving forces including toxic pollution is
missing. MODELKEY aims to provide tools to close this gap since efficient mitigation measures should be based on a reliable identification and evaluation of all relevant risks to the ecological status.

**Thesis 2: Biotesting of effects on different levels of biological organisation and the application of biomarkers provide a crucial link between the chemical and the ecological status**

There is a clear focus of the WFD on the improvement and protection of ecological status as defined by the quality of the phytoplankton, macro-phytes, invertebrates and fish communities. The protection of these communities as a goal is widely accepted. However, it is rather questionable whether the assessment of community alteration together with the analysis of priority pollutants is sufficient to identify and assess the toxic risk to the ecosystem due to contaminants. Major reasons are as follows:

1) Priority pollutants often do not explain toxic effects of environmental samples.

2) The assessment of risks requires early warning systems to predict the risk to communities. If there is observable damage in the community (e.g. disappearance of fish species [2]) it might be too late to initiate effective mitigation measures.

3) Since toxicants often occur together with other stressors such as habitat alteration, temperature change or pathogens we need tools to disaggregate toxic effects from the impact of other stressors as well as approaches to assess cumulative risk arising from the combined impact of multiple stressors. For example, in the case of lack of oxygen and habitat alteration toxic risks may be masked but may prevent recovery after improving habitat quality [3].

4) Present indices for community quality are mainly based on eutrophication and oxygen problems but do not take into account sensitivity towards toxicants.

To close the gap between ecological and chemical status MODELKEY aims to provide an assessment strategy together with a toolbox to link community risks with contamination (Fig. 2). This will include mode-of-action-directed *in vitro* testing together with related biomarkers applicable *in vivo* and *in situ*, *in vivo* tests on the level of individuals, populations and communities, indices on community quality based on sensitivity towards toxicants (SPEAR, SPEcies At Risk [4;5], effect-directed identification of key toxicants [6] and realistic measures for exposure and bioavailability [7;8].

Cause-effect-relationships may be confirmed on a community level applying Pollution Induced Community Tolerance (PICT).
towards individual toxicants [9]. This was shown e.g. for two geographically closely related rivers in the area of Bitterfeld (Fig. 3). One (Spittelwasser) is contaminated with significantly greater prometryn concentrations than the other (Mulde). Community effects of this contamination could be confirmed by photosynthesis inhibition in biofilms with a significantly greater EC$_{50}$ value of prometryn in the more contaminated river.

**Thesis 3: Effect-based toxicant identification simulating realistic exposure scenarios allows the establishment of site-specific cause-effect-relationships**

Environmental samples frequently cause adverse effects in biotest systems suggesting possible toxic hazards to biodiversity and ecosystems goods and services. Priority pollutants as defined by the Water Framework Directive often do not explain these effects. Instead, non-regulated or so far unknown toxicants may play an important role for measurable effects. This was shown for example for the industrial area of Bitterfeld as a major source of contamination for the lower Elbe river. Several non-regulated polycyclic aromatic compounds were identified as key toxicants [10-12] (For examples see Fig. 4). Thus, priority pollutant analysis alone often is an insufficient tool to direct source control and mitigation measures for a reduction of toxic burdens in aquatic ecosystems. MODELKEY aims to provide approaches and tools that are needed to identify those compounds that actually cause effects on different levels of biological complexity. Effect-directed analysis (EDA) integrating effect analysis with fractionation procedures and chemical analysis helps to identify and confirm cause-effect relationships. The integration of bioavailability-directed extraction [8] and dosing techniques help to simulate realistic exposure scenarios and advance EDA towards hazard and risk directed analysis. Novel candidate priority compounds to be integrated into monitoring programs are identified.

**Thesis 4: Integrated exposure and effect modelling helps to translate local hazards to the basin scale**

Regional and national monitoring programs collect enormous amounts of data on water and sediment quality at many sites in European rivers. Unfortunately, several lacks impede an optimal exploitation of these data sets for basin scale risk assessment. This includes inconsistent and incomplete data sets, inconsistent data formats, scattered data storage and a lack of integrated modelling tools to assess and predict exposure, effects and risks in the basins. To provide guidance how to solve this problem MODELKEY compiles a uniform database integrating monitoring data on habitat quality, physico-chemical parameters, contaminant concentrations, toxicity data and ecological data for three river basins (Elbe, Scheldt and Llobregat) as a possible core for later extension to other basins. MODELKEY provides guidance on parameters that need to be determined for assessment and discrimination of different stressors on ecological status.

For translation of local hazards to basin scale MODELKEY develops exposure models to predict concentrations in sediments, water and biota downstream of contamination sources under different climatic and hydrological scenarios. Recent research on nutrient management on a basin scale (EU FP5, daNUbs) has revealed that there may be "conflicts of interest" between the local water quality and basin-wide impacts: water bodies with a high dilution capacity usually do not feature severe local water problems, but they are very effective in transporting pollution downstream where it may cause severe problems in downstream lakes, estuaries and coastal waters [13]. On the other hand, water bodies with a low dilution capacity often show severe pollution problems, but the problems stay local.
and the pollution is not causing downstream impacts. Similar paradoxes may exist in the management of toxic pollution, including sediments.

For risk assessment exposure data from monitoring or modelling need to be closely integrated with effect observations and predictions. In MODELKEY this is done by integrated risk indices and effect models based on joint effects of multiple toxicants together with other stressors. Modelling approaches include estimates of bioavailability, species sensitivity distributions and modern statistical tools to assess the degree of impact as well as the contribution of different stressors. Artificial neural network technologies help to link observed and predicted impacts on aquatic communities and to forecast ecosystem responses on changing environments. The database, risk indices as well as exposure and effect models are integrated into a user-friendly Decision Support System that helps to assess hazards on different spatial scales from site to basin.

Recommendations & perspectives

Successful implementation of WFD to achieve a good ecological status in European river basins demands for a specific focus on the establishment of cause-effect relationships between stressors and effects on the aquatic ecosystems and their goods and services. Toxic chemicals should be given more attention in the assessment of the ecological status. This problem is not covered by consideration of the chemical status since the latter does not reflect the actually present set of hazardous compounds. A stronger use of biological and chemical assessment tools is required to link ecological status to chemical stressors including effect analysis on different levels of biological organisation, exposure and bioavailability assessment and novel toxicity-based indices for ecological quality. Hazard assessment on a basin scale needs to be based on site scale knowledge including exposure, effects and community alteration (TRIAD approach) together with habitat quality compiled in a European database. Integrated exposure and effect models may help to assess and extrapolate risks to downstream areas and to altered hydrological, climatic and contamination scenarios.

References

Werner Brack is an environmental chemist heading the Department Effect-Directed Analysis at the Helmholtz Centre for Environmental Research – UFZ in Leipzig. He is the co-ordinator of FP6 Integrated Project MODELKEY and the head of WP4 on “Risk Assessment and Harmonisation” of the Coordination Action RISKBASE.

Major research interests are the effect-directed analysis of toxicants in complex environmental mixtures, the establishment of cause-effect relationships between contamination and biological effects and risk assessment of complex contamination.
Flood risk assessment

Volker Meyer

1Helmholtz Centre for Environmental Research – UFZ, Department of Economics Leipzig, Germany, volker.meyer@ufz.de

Background

In recent years a shift in flood policy from the old concept of “flood protection” to the new paradigm of “flood risk management” can be recognised (Schanze 2006). To simplify matters, flood protection aims at preventing flood hazards up to a certain magnitude by providing a certain protection level (e.g. against floods of an exceedance probability once in 100 years). Such protection levels are mostly established by means of structural measures like dikes etc. In contrast, flood risk management does not only consider the hazard but also the possible consequences. It tries to adjust flood protection to the risk situation by concentrating protection efforts to areas with a high expected damage, in order to spent public funds in an economically efficient way (Messner & Meyer 2006). Thereby, the whole spectrum of flood risk reduction measures, from technical solutions to retention or warning systems is considered.

Flood risk management can be broadly divided in two steps (Schanze 2006): flood risk assessment and flood risk reduction. While the objective of risk assessment is to provide information on current or future flood risks in order to find out where these risk are unacceptable high risk reduction aims at finding measures to decrease these risks.

The term risk is hereby understood as the probability of negative consequences (ebd.) and is measured by the formula

\[ \text{risk} = \text{probability} \times \text{negative consequence} \]

In other words this is the expected annual average negative consequence of flooding, whereas “negative consequences” covers economic, social as well as environmental consequences. This formula goes back to the definition of risk introduced by Knight (1921, see e.g. Hansjürgens 2004 and Köck 2001) and is based on the assumption that risks are measurable.2

For the practical application of flood risk assessment this means that the negative consequences have to be evaluated for flood events of different probability in order to construct a damage-probability curve (see fig. 1). The risk (or the annual average damage) is shown by the area or the integral under the curve.

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1 Flood risk assessment is sometimes differentiated furthermore in a more objective risk analysis and societal risk assessment/valuation. However, like Schanze (2006) argues, this does not mean to value a (more precise) “objective” risk analysis from a “subjective” risk assessment. In fact a clear separation of both is often impossible or, like Köck (2001) states, “also the domain of a pretended purely scientific risk analysis does not go without valuations”.

2 It should be noted that this risk formula is often criticised especially in social science for several reasons (see e.g. Banse & Bechmann 1998): Firstly, it implies that an “objective risk” exists and can be measured. This is often not the case because of large uncertainties in the data, variations in time and very complex perceptions and evaluations of risks among people. Secondly, the risk formula suggests that risk is somehow naturally given. In contrast to that sociologists like e.g. Renn (1998) argue that risk is always associated with human decisions or actions: “risks refer to the possibility that human actions or events lead to consequences that affect aspects of what humans value”. With regard to flood risk this means that this current risk situation (whether it can be quantified or not) is always a product of human actions or decisions, like for example to settle in the floodplain (or not), to build up protection measures (or not) etc. These aspects should be kept in mind when assessing risks. We nevertheless use the risk formula in the following as we believe that even an uncertain estimation of a risk measure can be a valuable information basis for new human decisions.
Damage evaluation approaches usually deploy the following kind of input data in order to estimate flood damage (Messner et al. 2007):

- Inundation characteristics, i.e. data especially on the estimated area and depth of a certain flood event, calculated by hydrodynamic models.
- Information on number and type of the exposed elements at risk (people, properties, biotopes etc.), usually gathered from land use data sources.
- Information about the value of these elements at risk (either in monetary or non-monetary terms).
- Information about the susceptibility of these elements at risk, usually expressed by depth/damage-relationships.

Apart from these general components, a huge variety of damage evaluation approaches exist. Regarding their spatial scale an accuracy level the existing methods can be broadly differentiated into macro-, meso- and micro-scale approaches (ebd.). Macro scale approaches e.g. often rely on land use information with a low spatial resolution and/or low typological differentiation in order to reduce the effort of analysis and hence be able to consider large river basins as a whole (see e.g. IKSR 2001, Sayers et al. 2002). Micro-scale approaches on the other side try to achieve more accurate results by applying very detailed land use data, as well as value and susceptibility information (see e.g. Penning-Rossell et al. 2005). Of course this requires more effort which restricts these approaches often to small research areas.

This directly leads to a first problem/thesis for the application of flood risk assessment.

**Thesis 1: Uncertainties in risk assessment: a trade-off between accuracy & effort**

Even though great improvements within the methods of flood damage evaluation/risk assessment have been made during the last decades, the uncertainties in the results are still high, e.g. due to data problems in all parts of risk analysis (Nachtegael 2007). Theses uncertainties will never be reduced completely. Hence, an objective flood risk may exist but will never be exactly quantified.

However, like stated above, flood risk assessment approaches differ concerning the degree of accuracy they are able to achieve. Thus, it is a question which degree of uncertainty in flood risk assessment one is willing to accept with respect to the objective of the study. The choice of an appropriate risk assessment approach is hence always a trade-off between accuracy & effort. I.e. for a flood risk assessment on a river basin scale a macro- or meso-scale approach might be sufficient, while when decisions on concrete risk reduction measures for a specific site have to be made the more detailed micro-scale approaches should be applied.
In any case the reproach of a feigned accuracy should be counteracted by documenting and - if possible - quantifying the uncertainties within the risk assessment results. This can be either documented by a range, i.e. a lower and upper and maybe a mean value, by a standard deviation figure or by means of a probability distribution.

Thereby, the request for a transparent documentation of the uncertainties of risk assessment can be satisfied (Köck 2001). However, it should be noted that risk assessment should not be the only decision-determining criteria but only one source of information within the decision making process.

**Thesis 2: Social & environmental flood risks are often neglected**

The current practice of flood risk assessment like described in the beginning still focuses on economic damages, especially damages on buildings and their inventories. In contrast, social and environmental effects of flooding, like e.g. loss of life, stress or destruction of biotopes, are often not considered. This is partly because they are not, or at least not easily measurable in monetary terms and hence not comparable with economic damages. In consequence, flood risk assessment is often incomplete and hence biased, as an important intangible part of the overall risk is neglected.

Nevertheless, approaches exist which make it possible to include these “intangible” effects in an overall risk assessment. The traditional approach of environmental economics would be to monetise such public goods by valuation methods like contingent valuation (willingness-to-pay) or hedonic pricing. However, such valuations of life and environmental goods are still often criticised due to ethical reasons (see e.g. Hansjürgens 2004 for a discussion of such critics).

Another way of including intangibles in a non-monetary way is multi-criteria analysis (MCA, see e.g. (Keeney & Raiffa 1993; Malczewski 1999). Generally speaking, the key to the aggregation of the different risks is a weighting of the different criteria, which requires an involvement of decision makers and/or stakeholders in the assessment process.

**Thesis 3: The spatial distribution of risks and risk reducing effects is often not considered**

The spatial distribution of risks as well as of the effects of flood mitigation measures is rarely taken into account. E.g. the evaluation and selection of appropriate reduction measures is mostly based on their overall net benefit. Therefore, it is often not considered which areas benefit most from a measure and which areas do not. This may lead to spatial disparities of flood risk which are not desirable or acceptable.

Furthermore, little attention is given at present to the effects flood risk reduction measures have to areas downstream in river basin. E.g. when a dike is planned often only local risk reducing effects are considered as benefits while negative effects on the flood peak in downstream areas are neglected.

**Recommendations & perspectives**

Summarising the theses above, some recommendations can be given for the improvement of flood risk assessment in order to provide better support for flood risk management decisions:

- Flood risk assessment is always to some degree uncertain, but these uncertainties in the results should be documented in order to provide decision makers with information on the quality of the data they are using as a decision support.
- Social and environmental flood risks should be also considered in an overall risk assessment e.g. by means of multicriteria analysis.
- The spatial distribution of flood risks and risk reducing effects of mitigation measures should be shown by appropriate risk mapping approaches.

Some of these aspects will be already required in the forthcoming European “directive on the assessment and management of flood risk” (EU 2006/C 311 E/02), which demands in article 6 a risk assessment and mapping of social, economic and environmental flood risk. In our presentation we will briefly introduce a multicriteria risk mapping approach which was developed and tested at the River Mulde in Saxony, Germany (Meyer 2007).
References


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Research interests:
Flood damage evaluation & risk assessment, economic evaluation & decision support
Current projects:
FLOODsite, Flood-ERA, GLOWA Elbe
Climate change impacts on hydrology - State of the art and future challenges

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Background
The global mean surface temperature has been persistently increasing over the past decades (IPCC, 2007a). The functioning of the earth system is very complex, and there are manifold processes influencing climate, and many systems and processes are influenced by the climate. Therefore, multiple impacts along different causal chains are to be expected worldwide as a result of climate warming.

In Europe, impacts on water resources are among the main concerns. Many recent investigations highlighted the challenges that result from shifts in precipitation patterns and snow regime, changes in seasonal water availability and water quality, rise of sea level, and increase in the frequency and/or intensity of floods and droughts, all coupled with the rise in mean surface temperature. The main findings are summarized in different reports assigned by the European Commission, the European Environmental Agency and the United Nations (EEA, 2004, 2005a, 2005b; Eisenreich, 2005; IPCC, 2001, 2007).

Thesis 1: Climate change induces a change in the seasonality of river runoff in Europe
Changes in annual river discharge are projected to vary significantly across Europe, related to regional environmental settings and local changes in precipitation and temperature. As a general trend demonstrated by many modelling studies (Hattermann et al. 2007a), runoff in higher-latitude areas with increased precipitation will most likely increase in winter and decrease in spring (less precipitation as snow in winter and less snow melting). Summer flow in river basins with considerable groundwater contribution will change in accordance with changes in precipitation during the groundwater recharge period in winter (Fig. 1). In arid and semi-arid Mediterranean areas change in precipitation will be translated into runoff change. In all cases change in runoff will be considerably higher in percentage than change in precipitation due to the non-linearity of response (Hattermann et al. 2007b).

Fig. 1: Change in river runoff (river Elbe, gauge Neu Darchau) under climate change conditions: observation (average daily value 1961-90) against scenario (average day 2046-55, 100 realizations (in orange and red), Hattermann et al. 2007b).
Thesis 2: Water resources in Europe and water related sectors are sensitive to climate change

The above stated changes in water balance components under climate change translate into changes in water resources availability. Apart from climate change effects on the water cycle, human-induced pressures (mainly in domestic, industrial, and agricultural sectors) control the availability of water resources in Europe. A large proportion of Europe’s population already lives in watersheds with less than 1700 m$^3$ capita$^{-1}$ year$^{-1}$ (the water-stress threshold defined by the World Bank), and beyond the 2020s climate change would increase the number of people under this threshold by tens of millions. Information on the combined impact of climate and human water withdrawals can be also found in studies by Lehner et al. (2001), indicating increasing pressure on water resources in most river basins of southeastern, southern and parts of central Europe by 2050s compared to situation in the 1990s.

Thesis 3: Climate change will lead to an increase of extreme events

Notable changes are also projected for precipitation extremes in Europe, though the uncertainties in the prediction of extreme precipitation are still high. For example, Palmer and Raisanen (2002) project that the probability of extremely high precipitation over Europe in winter would increase by about two to five times over the next 50 to 100 years, leading to increased risk of severe floods (see also Fig. 2).

Fig. 2: Change in 100-year return level between the control and scenario periods under A2 emission scenario for European rivers with an upstream area larger than 1000 km$^2$ (based on HIRHAM climate run at 12 km. Source: European Commission, DG Joint Research Centre, personal communication).
Other studies argue that it is likely that the frequency of both intense precipitation events and summer drought risk in Europe will increase (Klein Tank et al., 2002; Parry, 2000). However, the attribution of these extremes to climate change is still uncertain due to a lack of accurate data and a full scientific understanding of the climate system functioning. Better methods to quantify uncertainty related to flood generation are therefore needed (Apel et al. 2004). Some first assessments have appeared showing that extremes are changing more than the average climate (e.g. Schär et al., 2004).

According to the Clausius-Clapeyron law, the atmosphere’s water holding capacity increases with temperature, and the potential for more intense precipitation increases. Higher and more intense precipitation has already been observed globally, and this trend is expected to increase under global warming. This is a sufficient condition for flood hazard to increase. Besides, there are other, non-climatic factors, like land use change escalating flood hazard (Apel et al. 2004). Among them are deforestation, urbanisation, elimination of floodplains and wetlands, and construction activities in flood-prone and coastal areas.

**Recommendations & perspectives**

Summarising the scientific discussion, consequences of climate change for water-related sectors could be severe. On the other hand, these sectors had to deal with climate variability in the past and this experience could help to adapt also to climate change (Kabat et al. 2002). However, it is also likely that the ongoing climate change will lead to situations or extremes never faced before, and new technical measures or management strategies or combinations of both have to be implemented. Here, experiences gained somewhere else under comparable situations may help to adapt, and the EU should support adaptation measures by its legislation.

One major problem is that the trend in climate is not steady but overlaid by climate variability. It is therefore uncertain whether a climate extreme is a climate change signal, and this uncertainty may lead to a situation where a single actor (farmer, company) may invest in inadequate adaptation measures, which could be either inadequate in space (regional impact insufficiently addressed) or inadequate in time (too early and investments do not pay off, or too late and damages are irreversible), although the sector as such will adapt to climate change over time. Therefore adaptive management as a systematic process for continually improving regional management policies and practices by learning from the outcomes of implemented management strategies is of a high priority, and a further investigation of

- Regional climate change impacts and possible adaptation measures,
- Quantification of uncertainties,
- Adaptive management flexible enough to take into account new developments, and
- Support of national and European legislation for adaptation

is important.

**References**


Dr. Fred Fokko Hattermann is a senior scientist at the Potsdam-Institute for Climate Impact Research (PIK), where he is studying the impacts of global change on the catchment and continental scale. Part of his work is the integrated analysis of climate and land-use scenarios and how they influence regional water balances, agriculture and the water quality of river basins.

He has a long-term experience in project management and coordination. Actual projects are the European project “Harmoni-CA - Harmonised modelling tools for integrated basin management”, the German EU Presidency preparation project „Climate change and water management - adaptation strategies for Europe”, and the German project GLOWA (GLObal WAter) Elbe.
Drought risks in Mediterranean river basins

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Background

The Medroplan project (Mediterranean Drought Preparedness and Mitigation Planning) has developed a systematic approach to assist in the development of drought and water scarcity management plans linking science and policy (http://www.iamz.ciheam.org/medroplan). The contribution of the Medroplan research teams and collaborators is acknowledged for their valuable input.

Droughts occur very frequently in the Mediterranean countries with severe economic and social consequences also connected to the vulnerability of the water supply systems, the agricultural systems and of society in general. Such vulnerability is due to situations of permanent water scarcity, quality deterioration and increasing water demands deriving from population growth, tourist development and irrigation needs. Thus, a policy for drought management is required based on actions aimed to improve drought preparedness and to mitigate impacts of ongoing droughts. The guidelines are designed to contribute to key social and policy questions: (1) How can water management be improved, and how best can people benefit from such changes? The present contribution argues that there are options to minimize the risk of drought impacts by promoting drought preparedness and management plans. (2) How can research help to development innovative institutional arrangements and decision-support tools? The Guidelines provide a framework and systematic approach to link academic knowledge to operational and policy aspects of drought risk management.

A multi-stakeholder dialogue is necessary to: increase the quality and acceptance of drought management plans; increase acceptance of or trust in the science that is in the basis of the planning; and to provide essential information and insights about drought preparedness since the relevant wisdom is not limited to scientific specialists and public officials. Concepts are essential to ensure results of the multi-stakeholder dialogue. Drought, aridity, water shortage, water scarcity and desertification are common and overlapping processes in Mediterranean countries and often are misinterpreted and used. Starting with clear and agreed definitions and concepts contributes to the development of clear methods and to the correct interpretation of the results for developing drought management plans.

Risk management approaches

A crisis management approach is based on the implementation of measures and actions after a drought event has started and is perceived; this approach is taken in emergency situations. This approach often results in inefficient technical and economic solutions since actions are taken with little time for evaluating optimal actions and stakeholder participation is very limited.

A preventive approach includes all the measures designed in advance, with appropriate planning tools and stakeholder participation. The proactive approach is based both on short term and long term measures and includes monitoring systems for a timely warning of drought conditions. It can be considered an approach to “manage risk”. A proactive approach consists in planning in advance the necessary measures to prevent or minimize drought impacts. Such an approach includes preparedness of planning tools which enables to avoid or reduce the consequences of a possible water emergency, and the implementation of such plans, when a drought occurs. The proactive approach foresees a continuous monitoring of hydrometeorological variables and of the status of water reserves in order to identify possible water crisis situations and to apply the necessary measures before a real water emergency occurs. Nevertheless, if it is not possible to avoid a water crisis that appears as a natural public calamity (after a government declaration), the Drought Contingency Plan is implemented until the establishment of normal conditions. It is evident that a proactive approach, even if more complex, is more efficient than the traditional approach, since it allows to define in advance drought mitigation measures (both long term and short term) improving interventions quality.
The implementation of a proactive approach implies drafting plans in which the mitigation measures are clearly defined together with the instructions for their implementation. At this end, a clear assignment of competences among the different involved institutions appears to be a key issue; therefore a legislative act which defines the responsibilities is necessary in each country. Such act could be part of national water resources policy and/or strategy to fight desertification (within the U.N. convention).

No single management action, legislation or policy can respond to all the aspects and achieve all goals for the effective drought management. Multiple collaborative efforts are needed to integrate the multidimensional effects of drought on society.

Other important aspects to take into account include: Stakeholders’ participation; management and changes in water rights legislation allowing water exchange during droughts; and definition of standards of efficiency to foster water saving and sanctions for who does not respect them.

**Institutional and legal framework for coping with drought in Europe**

The European Union Water Directive 2000/60 explicitly defines planning as the main tool to guarantee protection of water bodies and indicates mitigation of flood and drought events as main objectives. However it does not take into account criteria and actions to face drought risk, references to drought are rare and ambiguous and often misleading and mitigation measures are only considered optional.

Most European countries have not issued a legal framework to face drought risk and emergency actions are managed by Civil Protection Agencies or some legislative acts referring to natural disasters recovery. The lessons learned during recent droughts have shown the inadequacy of the legal systems, fostering towards planning of drought mitigation measures and the substitution of subsidies to cover damages in agriculture with insurances. Spain is an example of institutional support for taking these initiatives. The success in most cases is due to the management of water at the basin level, allowing coordination of policy, physical and technical aspects.

**Risk analysis**

The complexity of drought calls for complex methods of analysis. The analysis of risk needs to incorporate methods that evaluate the system: the drought hazard, the risks to different systems, the causes of risk, and the operational aspects to decrease risk. These aspects can be evaluated in isolation or in an integrated approach. The complexity of these topics suggests a wide range of possible evaluation methods. Each method has its own merit and they are usually supportive of each other. A combination of methods is usually most rewarding. The results of the risk analysis provide elements that support the controversial official declaration of drought and of its different levels of alert.

The concepts of vulnerability and risk are part of the common language and the concepts are used by most people in daily live. These concepts are used loosely in many different contexts, from medicine to poverty and development literature. In the context of natural hazards, the concepts are often derived from the social sciences since there is an explicit demand for increasing social protection to natural hazards. In contrast, the concept of risk in engineering is physically based on the computation of failure probabilities in a hydrological system.

Regardless of the nuance of risk definitions, the key concepts are: (a) Risk relates to the consequences of a perturbation, rather than its agent; and (b) Risk is a relative measure and critical levels of risk must be defined by the analyst. Risks are always created or exist within social systems, therefore it is important to consider the social contexts in which risks occur and that people therefore do not necessarily share the same perceptions of risk and their underlying causes.

In water supply systems, drought is characterized by a high level of complexity. In general, a set of performance indices, attempting to capture different aspects related to concepts such as reliability, resiliency and vulnerability, is used. Indeed, the stochastic nature of inflows, the high interconnection between the different components of the system, the presence in some cases of many conflicting demands, the uncertainty related to the actual impacts of extreme events such as droughts, make the risk assessment of a water supply system a problem that is better faced through a set of several indices and/or by analyzing the probabilities of shortages of different entities.
Quantitative evaluation of risk in water supply systems may follow several approaches as consequences of the various approaches for quantifying probabilities: (1) risk defined as the probability of an adverse event; (2) risk defined as the expected consequence or damage due to an adverse event.

**Evaluating drought management actions**

Long and short term activities and actions can be implemented to prevent and mitigate drought impacts. Such activities and actions are essential in the development of specific drought planning and response efforts. The definition and evaluation of drought management actions includes: (1) Preparedness, early warning, monitoring systems; (2) Establishing priorities of water use; (3) Defining the conditions and the thresholds to declare drought levels; (4) Establishing the management objectives in each drought level; (5) Defining the actions; and (6) Implementation of actions.

**Public review of drought management plans**

Public review has to play an important role along the process of developing a plan since the social and environmental conditions may change and aspects of risk analysis and management improve and evolve. Once the plan is developed, it may be necessary to revise an existing plan periodically certain aspects of the plan.

In all cases public revision is complex, but in most cases includes two aspects: Dissemination of the information to be revised and multi-stakeholder dialogue to revise the information. The feedback from stakeholders is may be collected by means of the responses to questionnaires, group interviews, or other methods to obtain information. The interviews may be public in order to allow the participation and discussions among all stakeholder groups.

A periodic revision of the plan by institutions and stakeholders is very advisable, as situations change and plans should be adapted to these changes. Moreover, it is obvious that an in-depth revision of a drought management plan should be making after each drought episode, analysing the response of all the aspects of the plan. This analysis would provide elements to adapt and improve the plan, in a continuous feedback process that keeps it updated.

**Recommendations & perspectives**

The Mediterranean region is undergoing rapid socio-economic and technological changes that increase the pressure on its already structural water deficit and question the ability to maintain the current management philosophy. In addition, climate change projections indicate an increased likelihood of droughts. Institutions in the region are evolving to respond to these pressures and to ensure more sustainable water resources management. There is an ongoing progress in many of these countries, which is favoured by the increasing regional cooperation, a better monitoring and management systems, and above all by the awareness of governments. The adoption of emerging technologies for using fresh or unconventional water resources more effectively is crucial for water management. Drought management measures need to be integrated into the long-term strategies for water and land uses and overall development strategies. When water resources are managed at the river basin level, there is an opportunity to respond directly to policy decisions and to the needs and problems of the natural hydrological system. Monitoring and early warning systems continue to improve and are being incorporated into the planning processes. Lastly, strengthened regional cooperation and better understanding of the resource’s dynamics and social dimension, and more efficient monitoring systems give hope for alleviating the present pressures on the water resources in the next decades.

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A complete technical Annex of the MEDROPLAN project includes several hundred references on drought management and can be downloaded at: http://www.iamz.ciheam.org/medroplan
Ana Iglesias is a Professor of Agricultural Economics at the UPM. Her research focuses in understanding the interactions of global change with agriculture and water resources. She is the Scientific Coordinator of the Drought Management Guidelines within the MEDA program of the European Union. Her collaborative work has been published in over one hundred research papers.
Risk of hydromorphological changes on biodiversity - Case studies from the Baltic Sea region

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Background

Hydropower greatly affects the status of water courses. According to the Water Framework Directive (2000/60/EC) these hydrologically and morphologically altered water bodies can be designated as heavily modified, meaning they do have lower environmental goal called good ecological potential. The assessment whether the water body is in the good ecological potential is a multi-phase and complex process. It requires that hydro-morphological mitigation measures to improve ecological status are identified and their impacts on uses are estimated. In the process those measures which cause significant harm for the uses of the water body are excluded. The WFD requires that the opportunities to improve ecological status in regulated water courses are assessed and where needed also realized. This may cause losses for hydro power production. On the other hand, the Directive on the promotion of electricity produced from renewable energy sources in the internal electricity market (2001/77/EC) demands an increase in hydropower production by 10–20 % at a national level. Therefore it is essential to evaluate the risk of hydromorphological changes on biodiversity and to establish a proper basis for determination of ecological potential.

Main aims of this study are to describe the effects of hydropower modifications of lakes and reservoirs in Fennoscandinavia and to define a good ecological potential by examples from Finland. Study is based on Interreg IIIB Baltic Sea Region project Watersketch (Principles, Tools and Systems to Extend Spatial Planning on Water Courses) and some other recent projects around Baltic Sea region.

Thesis 1: Nordic lakes are severely affected by water level regulation

Water level regulation is related to the human need to control the water levels of the lakes and flows of the rivers in such a way that benefits various users of watercourses. In a typical hydropower regulation project in the northern hemisphere, water levels during summer period are normally high or rising, while during the winter period, when the need for electricity is normally at its highest, the water level is strongly lowered. Flood prevention regulation follows a similar pattern during winter time, but in summer time some storage capacity is left empty to catch flash floods. When the major objective of the regulation is recreation or navigation, then regulated water levels are often more stable than natural ones. If the water level is regulated for water supply use, the water level fluctuation is more irregular and depends on the specific use of raw water.

There are hundreds of regulated lakes in Finland, Norway and Sweden (Marttunen et al. 2006). For instance, in Sweden, there is 563 lakes larger than 1 km² with water level regulation vary from 0.1 m to 35 meters. In Norway, there are approximately 900 reservoirs and in half of these reservoirs, the water level fluctuation is more than 5 metres. The highest regulation amplitude is 140 m. In Finland, there are more than 350 regulated lakes. Finnish lake regulations are usually relatively mild in terms of annual water level fluctuation. Half of these projects show that the annual water level fluctuation is less than 1 metre. However, in Finland, most of the largest lakes are regulated and consequently one third of the total lake area (about 11 000 km²) is regulated.

In summary, many Swedish and Norwegian reservoirs are much more heavily regulated than Finnish ones (Marttunen et al. 2006). However, the regulation amplitude itself does not directly describe the magnitude of ecological impacts of regulation. For instance, in Finland lakes are generally much shallower and their water is more coloured and consequently the productive zone is narrower than in Norwegian and Swedish lakes. Finnish regulated lakes lie in lowland areas with elevation less than 250 metres above sea level, and are relatively shallow with average depths of about 7 metres and gently sloping shores. In Norway and Sweden, many regulated lakes lie in the highlands and have oligotrophic character with secchi depth more than 5-10 metres. The slopes of these lakes are also
Due to the differences in altitude and latitude of lakes, there are also differences in the biology of the lakes. For instance, in large Finnish regulated lakes there are usually about 15-20 fish species whereas in the high land lakes in Norway the fish fauna consists of few salmonid species, or fish are not found at all.

**Thesis 2: Ecological effects of water level regulation are pronounced on littoral zone**

Changes in the water level fluctuation regime cause significant changes in the littoral, which is the most visible part of the lake ecosystem for normal lake users. When the water is transparent, the negative effects of a fluctuating water level are less severe than in the case of turbid or coloured water, due to the wider productive zone (Rørslett 1988, Palomäki 1994, Hellsten 1997).

The littoral undergoes considerable geomorphologic changes during the initial stages of lake level regulation, especially if the mean water level is raised. This leads to major geomorphologic changes, including breakdown of the organic surface layer and erosion of minerogenic matter. However, the increase in erosion has a negative effect on the littoral vegetation and benthic fauna, which leads to the decreased littoral production. These erosional processes cause destruction of vegetation and affect the successional status of the vegetation, as reported in several Scandinavian lakes (Nilsson 1981, Rørslett 1985, Hellsten 2001). As a result of winter draw-down, the ice layer may extend down to the bottom, causing the sediment to freeze and to be partly eroded by scouring (Renman 1993, Palomäki & Koskenniemi 1993, Hellsten 1997). Great losses in the production of autumn spawning fish and benthic fauna are well known to be a consequence of ice extension (Huusko et al., 1988; Tikkanen et al. 1988; Palomäki, 1996). However, there are still open questions concerning how the changes in reproduction affect the fish stocks (Sutela et al. 2002).

As the water level is kept relatively low during the early spring, the maximum water level during the spring flood has been lowered remarkably in many regulated lakes in Scandinavia and it tends to shift towards late June (Marttunen et al., 2006). In the lakes, the changes in spring flood water levels affect the reproduction of spring spawning fish, because most of the spawning areas are not accessible during the spawning period. However, it also has a negative effect on the early stages of young fish fries of autumn spawning fish, which have to find shelter among littoral vegetation.

Aquatic macrophytes are one of the best indicators of hydromorphological changes in lakes (Rørslett 1989, Hellsten 2001). Large sized isoetids (Isoetes lacustris, I. echinospora, Lobelia dortmannna, Littorella uniflora) are key species of littoral zone of soft water lakes and sensitive to the effect of descending ice. Especially winter draw down of regulated lake pronounce the degradation of large isoetids (Fig. 1).

![Figure 1: Relationship between total number of large isoetids and winter draw-down of lake water level in some selected Finnish lakes.](image)

\[ y = -0.3594x + 2.7001 \]

\[ R^2 = 0.2757 \]
Thesis 3: Definition of ecological potential includes several possibilities

The Water Framework Directive (WFD) reforms EU water legislation by introducing a new approach for water management. From an environmental point of view, the WFD’s ultimate aim is preventing further deterioration and achieving “good status” in inland waters (e.g. lakes), transitional waters (e.g. estuaries), coastal waters and groundwater. For heavily modified water bodies (HMWB) the objective is the achievement of good ecological potential, which may strongly deviate from good ecological quality, which is the objective for natural water bodies. There are several steps, which should be taken into account when the ecological potential is defined.

- First of all, ecological quality ratios for all relevant biological quality elements (macrophytes, zoobenthos, fish) should be determined. Taxonomic composition, abundance, or sensitive species should be used and one or several variables per quality element should be established.
- After determination of ecological status, there should be a clear decision of chosen mitigation measures and variables.
- Finally, an estimation of effects of mitigation measures on these variables should be carried out.

In principle there are two different possibilities to reach the goal of maximum ecological potential (Fig. 2). First option is to define all hydro-morphological mitigation measures to improve the current ecological status without causing any significant effect on the uses and to assess how they improve the ecological status (A). Second option is to start from theoretical high ecological status of water body and define measures which are needed to reach it (B). Next step includes removal of methods which have significant effect on use or are too costly. Both of these methods need a large set of data, which is not so commonly available in most of the lakes. Both methods have been tested in large regulated Lake Kemijärvi and River Oulujoki in Northern Finland (more details in www.watersketch.net). However, due to the tight time constraints and deficiencies in the biological data, for instance, more pragmatic approaches based on expert judgements will be used in the first planning period in Finland.

Fig. 2: Two possibilities for determination of maximum ecological potential. A) Derivation from current ecological status including all hydromorphological mitigation measures. B) Derivation from high ecological status removing all measures with significant effects on use.
Recommendations & perspectives

Large numbers of lakes are suffering of the consequences of water level regulation in Baltic Sea Region. Effects on biota are clear and especially focused on littoral zone. Implementation of Water Framework Directive with all environmental goals will result in systematic analysis of opportunities to improve the ecological status without causing significant harm on important uses of water courses. Unfortunately, the terms and process are difficult. Additionally, there are still many information gaps related to the knowledge about the status of water courses and the responses of the mitigation measures. This together means that the process is very challenging and it is very difficult to make accurate estimates about the impacts of measures on the ecological status. Therefore, pragmatic approaches based on the expert judgements are needed in the first planning period.

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Dr. Seppo Hellsten is currently working in Finnish Environment Institute with special focus on aquatic macrophytes and their role in ecosystem. He has been involved in several studies related to heavily modified water bodies and was a co-ordinator of Watersketch project.
Towards ecologically successful hydromorphological risk assessment and management in streams

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Background

The extensive ecological degradation of stream hydromorphy and concurrent loss of biological diversity resulting from human activities is causing great concern for conservation and restoration (Karr et al. 1985). The European Water Framework Directive and Natura 2000 strive to maintain and restore the stream ecosystems self-sustaining functioning. Making the proper choices in stream and catchment management requires understanding the functioning and interactions of the controlling factors. Stream flow can be seen as a master variable (Power et al. 1995) and is strongly correlated with physical characteristics of a stream. Sound hydromorphological risk assessment and management imply knowledge of hydromorphology-biota interactions, of spatial and temporal scale, its hierarchy (Frissell et al. 1986), and of references of their functioning.

Thesis 1: Hydromorphology best explains stream biodiversity and is spatial scale dependent

A clear understanding of spatial and temporal scale of objectives is needed to define risk assessment criteria or to prioritise restoration activities (Verdonschot 2000). Analyses of 889 samples collected in different stream types distributed all over Europe showed that hydromorphological variables explained far most of the variation in macroinvertebrate composition (Verdonschot & Nijboer 2004; Fig. 1).

Fig. 1: The explanatory strength of groups of environmental variables expressed as the average fraction of SCE grouped according to variable types. (average fraction of SCE is the ‘sum of canonical eigenvalues’ (SCE) of each separate analysis divided by the SCE of the same analysis using all environmental variables) (Verdonschot & Nijboer 2004).

Analyses of German, Dutch and Swedish stream data showed that hydromorphological variables were scale dependent and changed from catchment scale to reach or site scale (Feld 2004). Frissell et al. (1986) argued a distinct spatial hierarchy in hydromorphological stream habitats. Verdonschot & Nijboer (2004) showed that variables act more or less independently from spatial scale. Although some in-stream variables (morphological and physico-chemical) better explained the macroinvertebrate distribution in local regions and stream types, a clear hierarchy lacked. Furthermore, these results imply that any risk assessment needs a hydromorphological context.

Thesis 2: Current hydromorphological risk assessment methods should include hydrological parameters and temporal scale

Stream flow quantity and timing are critical components of water supply, hydromorphy, water quality, and the ecological integrity of stream systems (Poff et al. 1997). The five key components of flow regime are magnitude, frequency, duration, timing, and rate of change. All five directly and indirectly master the stream ecosystem structure and functioning. Four out of these five key flow regime components include a temporal aspect. An example is given by the population response of Agapetus fuscipes to discharge extremes in lowland streams (Fig. 2). The natural stream (bu) hosted a high density of A. fuscipes larvae, most turned to pupae from May to July. The semi-natural stream (st)
hosted much lower numbers, the pupae occurred earlier in the year and the recruitment was very high in July/August. An extreme flood in July diminished the recruitment in the natural stream (Fig. 2 left arrow), and a second extreme in August diminished the number of young larvae in the semi-natural stream.

**Fig. 2:** Total numbers of larvae (solid line) and pupae (dashed line) of the trichopterans *Agapetus fuscipes* in two lowland streams (indicated by code bu for the natural stream Bunderbosbeek and st for the semi-natural stream Strabeekervloedgraaf). Arrows indicate two extreme high discharge events.

**Thesis 3: Hydromorphological risk assessment is strongly based on a human (morphology) perspective**

Raven et al. (2002) compared three major hydromorphological assessment methods of streams in Europe. All three methods are based on about the same assessment categories, divided in channel, river banks and riparian zone, and floodplain features. The attributed associated with these features and assessed in the field or from maps are all visual, static, momentary morphological parameters. Even flow, subdivided in flow patterns and flow features, were assessed by visual physical parameters. Parsons et al. (2002) compared 7 hydromorphological assessment methods. Their results are summarised in Table 1.

<table>
<thead>
<tr>
<th>method</th>
<th>AUS-RIVAS</th>
<th>HAB-SCORE</th>
<th>Index of Stream Condition</th>
<th>Geomorphic River Styles</th>
<th>State of the Rivers Survey</th>
<th>Habitat Predictive Modelling</th>
<th>River Habitat Survey</th>
</tr>
</thead>
<tbody>
<tr>
<td>scale links to biota</td>
<td>Y</td>
<td>Y</td>
<td>N</td>
<td>P</td>
<td>P</td>
<td>Y</td>
<td>P</td>
</tr>
<tr>
<td>end-point is reference state</td>
<td>Y</td>
<td>Y</td>
<td>Y</td>
<td>I</td>
<td>Y</td>
<td>Y</td>
<td>I</td>
</tr>
<tr>
<td>relevant to biota (fauna)</td>
<td>Y</td>
<td>Y</td>
<td>Y</td>
<td>P</td>
<td>P</td>
<td>Y</td>
<td>Y</td>
</tr>
<tr>
<td>represent habitat processes</td>
<td>N</td>
<td>N</td>
<td>N</td>
<td>Y</td>
<td>Y</td>
<td>P</td>
<td>Y</td>
</tr>
<tr>
<td>rapid data collection/analysis</td>
<td>Y</td>
<td>Y</td>
<td>N</td>
<td>N</td>
<td>Y</td>
<td>P</td>
<td>Y</td>
</tr>
<tr>
<td>for use by non-experts</td>
<td>Y</td>
<td>Y</td>
<td>Y</td>
<td>N</td>
<td>Y</td>
<td>Y</td>
<td>Y</td>
</tr>
<tr>
<td>ease to interpret output</td>
<td>Y</td>
<td>Y</td>
<td>Y</td>
<td>P</td>
<td>Y</td>
<td>Y</td>
<td>Y</td>
</tr>
<tr>
<td>wide applicability</td>
<td>Y</td>
<td>N</td>
<td>N</td>
<td>Y</td>
<td>N</td>
<td>Y</td>
<td>P</td>
</tr>
<tr>
<td>ability to predict</td>
<td>Y</td>
<td>N</td>
<td>N</td>
<td>Y</td>
<td>N</td>
<td>Y</td>
<td>P</td>
</tr>
</tbody>
</table>

Both studies summarised the major problems encountered in the current hydromorphological assessment systems and indicated necessary improvements by including a meaningful river typology, reference conditions, different spatial and temporal scales, links to biota and hydrological parameters.

**Thesis 4: A natural hydromorphological state offers a high resilience to pressures**

A resilient, self-sustaining stream ecosystem is a moderate dynamic system with a related variability resulting from natural disturbances and has the capability for rapid recovery from change and stress (Holling 1973). For example, extreme floods can reset both the morphology as well as the biodiversity by changing bed shape and substrate patterns, and reducing numbers of organisms. Degradation of streams is thus related to the degree of change in variability. Negative examples of variability loss are bed incision, dam construction or stream straightening, whereby variability is diminished and resilience is lost. Ongoing management in such system is necessary (like, stream vegetation...
maintenance). Positive examples of natural variability improving resilience are shown by biota using refugia (Brooks 1998 [Fig. 3]).

![Fig. 3: Changes in total abundance (mean±SE, n=128) of macroinvertebrates in 4 habitats (middle of channel, behind boulders, at channel margins, and in pools) before (B) and after a major flood (duration=2d) in the Lerderderg River, Victoria. I=immediately after the flood, 7d=7 days after the flood, 14d=14 days after the flood. Arrows indicate statistically significant (P<0.05, 2-way [time by habitat] ANOVA followed by a post-hoc before-and after-contrast test) departures from preflood densities (adapted from Brooks 1998).](image)

The major refugium is provided by interstitial spaces between rocks in the stream channel. Recent studies in lowland streams showed that in these sandy systems plants and detritus accumulations functioned accordingly.

**Thesis 5: Stream conservation and restoration often fail to rehabilitate stream biodiversity**

The success of stream restoration depends on the societal and ecological potentials present in the area or catchment to be restored. The ecological potentials depend on the conditions in the catchment and the stream as well as on the reference image. An extensive inquiry on stream restoration in the Netherlands (206 projects; 1.5 million euros per year) gave interesting insights into the state of the running projects. But from the answers it became clear that though the ecological targets were high, little was done about evaluation of measures undertaken. Evaluation of three projects showed that the biodiversity either decreased or did not change (Verdonschot & Nijboer 2002). A comparable result was obtained by Jahnig (2007) as shown in Figure 4.

![Fig. 4: Restoration of 7 braiding streams in Germany. On the left the increase in amount and variability of substrates is shown (1= before, 2=after; numbers on top of bars represent the Shannon-Wiener Index of relative substrate composition). On the left the results of macroinvertebrate clustering shows that within-sections similarity is much larger than between-sections similarity, indicating no biological change (Jahnig 2007).](image)

The major conclusions were: (i) longer stretches should be restored, (ii) sources of recolonisation propagules should be identified, (iii) stepping stone measures are necessary, (iv) dispersal patterns and distances should be taken into consideration, and (v) important habitat features need to be enhanced.

**Thesis 6: Risk assessment should be based on the concept of dynamic abiotic conditions and include stochastic biotic response**

Assessment needs a guiding reference or target image that sets the upper limit for evaluation (Verdonschot 2000). This image describes the stream as a moderate dynamic, self-sustaining system resilient to external perturbations. This image may include irreversible changes made to catchment...
hydrology and morphology, such as permanent infrastructure on the floodplain (urbanisation, dikes, bank fixation). A dynamic state is one in which the abiotica (especially the hydromorphology) vary in composition, shape and configuration, and the biota vary in presence and abundance, over space and time as they do in the appropriate reference state, both due to the natural or near–natural flow variability characteristics of the (changed) stream. Such images can only be defined for specific regions. Measuring the success of ecological restoration needs measurable changes in abiotic and biotic components that move towards the guiding image. Especially, the biotic components more often need time to re-establish (Fuchs & Statzner 1990). It is advised to evaluate improvement by either comparing the variability of the restored stream with the pre-restored condition along with a comparable still degraded and more natural stream in a before-after-control-impact (BACI) design. Or by using knowledge on the functioning of the dynamic stream ecosystem and the species response in a process-based manner, like a tactics approach.

Recommendations & perspectives
Any risk assessment needs a hydromorphological context. Hydromorphological risk assessment needs:

- to be based upon an ecological river typology at regional/local scale,
- to link directly to aquatic biota,
- to include hydrological and hydraulic categories,
- to include both spatial (catchment, macro-, meso- and microhabitat) and temporal (diel to multi-annual) scale.

References


**Key research interests:** Ecological freshwater management. Ecological quality assessment of running and standing waters based on all organism groups. Development and implementation of Water Framework Directive, at a National and European level. Disturbance ecology. Restoration Ecology. Aquatic ecosystem functioning and organism life tactics. **Large projects:** EU-project EUROLIMPACS (Integrated project to evaluate the impacts of global change on European freshwater ecosystems), different National projects on lowland stream ecology and restoration.
How to assess degradation of freshwater communities due to hydromorphological changes

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²Swedish Agricultural University, Uppsala, Sweden
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Background

Man-made modifications to in-stream channel morphology are one of the main reasons that rivers do not achieve good ecological quality. Channelisation, dredging and weed cutting have seriously degraded habitat diversity thereby excluding some species and reduce numbers of others. In a Water Framework Directive context these are known as hydromorphological pressures and to mitigate these pressure are identified as being of key importance in fulfilling the Directive.

The current physical structure of streams and diversity of biological communities are closely linked past and present human activities in the entire catchment. Human activities influence stream ecosystems on multiple scales, ranging from direct manipulation of the in-stream environment on the stream reach to altering the landscape and land use in the catchment thereby influencing the hydrological pathways and morphological structure (Vannote et al., 1980; Frissell et al., 1986; Fitzpatrick et al., 2001; Allan, 2004). Past and present disturbances act simultaneously with different intensity on the stream ecosystem elements and it can thus be difficult to distinguish the exact disturbance from the individual stressors on the biotic communities (Lane & Richards, 1997; Harding et al., 1998; Allan, 2004).

Thesis 1: Physical-biological coupling are important in determining freshwater communities

The scientific literature reports many linkages between individual parameters (substratum, current velocity etc.) describing the in-stream physical environment and different attributes of the macroinvertebrate community. On a fine scale using high resolution measurements without other confounding factors, the importance of physical-biological coupling in streams have been shown albeit many aspects are still not fully understood (Hart & Finelli 1999). One reason is that numerous physical factors interact across different temporal and spatial scales, resulting in very different biotopes (riffles, pools etc.) as well as inducing variation within these biotopes. One example of the latter is shown in figure 1 as the response of two macroinvertebrate metrics (total abundance and EPT taxa abundance) to mean particle size varied considerably between to adjacent riffle biotypes in the same stream. The majority of studies have shown that diversity and abundance of macroinvertebrates increase with particle size as this increase habitat complexity and volume of interstitial spaces. In one riffle (B and D) there was the expected positive relationship between particle size and macroinvertebrate metrics whereas there was no relationship in the other riffle (A and C), the reason being that it was consolidated by fine sediments reducing habitat availability. This study illustrates that even within the same biotype that would appear completely similar when visually assessed there can be considerable differences in the physical-biological coupling.
Fig. 1: Relationship between median particle size and total macroinvertebrate abundance/EPT abundance in two riffles spaced 100 m apart in the same 3. order stream. A,C is the upstream riffle whereas B,D is the downstream riffle. From Pedersen & Friberg (2007).

Thesis 2: Multiple stressors interact and this will influence impacts of hydromorphological degradation

One key issue when assessing the influence of hydromorphology is that interpretation of results is often confounded by multiple stressors influencing freshwater communities. Typically, streams that have undergone a high degree of habitat degradation or alterations of flow regimes will be situated in areas with multiple anthropogenic pressures. A recent analysis of 1127 stream sites in three European countries showed that the strongest relationship to a biological metric (ASPT) was found when including both land use, chemistry and habitat modification in a multiple regression (Table 1). In addition, stressors can interact in a synergistically manner and increased concentrations of easy degradable organic matter will have a more detrimental impact in a habitat degraded stream because number of reactive surfaces and re-aeration capacity are reduced (Andersen, 1994).

Table 1: Relationship between the macroinvertebrate based metric ASPT and explanatory environmental variables (PTOT=total phosphorous concentration; ARABLE=percentage agriculture in the catchment; MD=morphological degradation). Data from the EU REBECCA project (SSPI-CT-2003-502158).

<table>
<thead>
<tr>
<th>Metric</th>
<th>Countries</th>
<th>Relationship</th>
<th>R²</th>
</tr>
</thead>
<tbody>
<tr>
<td>ASPT</td>
<td>Sweden</td>
<td>y = 5.26-3.33PTOT-0.014ARABLE+2.32MD</td>
<td>0.61</td>
</tr>
<tr>
<td></td>
<td>Slovakia</td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>Denmark</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>
Thesis 3: Hydromorphology can be assessed

The linkage of reach scale physical parameters and biotic samples on sites only disturbed by physical alterations are scarce. There has been moderate success in linking biota with the currently used habitat surveys which is exemplified by the generally weak relationships found when investigating the relationship between morphological degradation and number of EPT families in 1541 stream sites from four countries (Figure 2). This is partly due to the mixed nature of pressures acting on a river reach and the habitat surveys using parameters of at least two spatial scales. It furthermore reflects that methods for sampling macroinvertebrates often are on a different scale to that of the hydromorphological assessment and that the sampling strategy was developed to target mainly organic pollution. At present, a few assessment systems that target impacts of low flow (Extence et al. 1999) and degraded hydromorphology (Barbour et al. 1996, Lorenz et al. 2004) do exist but they are not generally applicable and could be further developed. Therefore, the development of more indicator systems sensitive to hydromorphological degradation, using appropriate sampling techniques, will be a key issue in future.

Fig. 2: The number of EPT families along a gradient in hydromorphological quality in 4 countries, (a) Sweden, (b) Slovakia, (c) UK, (d) Denmark. Data from the EU REBECCA project (SSPI-CT-2003-502158).

Recommendations & perspectives

Hydromorphological degradation is one the most important pressures on European and imposes a serious risk to freshwater communities either when acting alone or in combination with other pressures. Impairment of hydromorphology in river ecosystems will significantly reduce its resistance towards other pressures on the ecosystem. Hydromorphology are currently assessed using a range of techniques that are suboptimal as they lack appropriate sensitivity as well as the ability to quantify the importance of individual pressures. There is an urgent need to develop refined and updated assessment systems targeting hydromorphology.
References


Terrestrial-aquatic linkages and how recovery of riparian zones can improve ecological quality, assessment of pressures on stream ecosystems with specific emphasis on hydromorphology, interactions between organism groups and ecosystems function as indicator of impairment, global changes impacts on aquatic ecosystems and biodiversity
Biodiversity and eutrophication assessment

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Background

Management of aquatic ecosystems as prescribed by legislation has recently shifted from the regulation of emissions at their source, or setting of chemical quality standards for particular uses of water, to the establishment of ecological objectives for these ecosystems. At present, the most important EU legislation that embraces this approach is the Water Framework Directive (WFD, 2000/60/EC).

This Directive establishes a framework for the protection of inland surface waters, transitional waters, coastal waters and groundwater and, with regard to their water needs, the coupled terrestrial ecosystems and wetlands. The aim is to ensure long-term sustainable water management and to reach good quality status (i.e. good ecological and chemical status) by 2015. The assessment of the ecological status and setting of the practical management goals require several steps. The process has started with the characterisation of the river basins including identification of surface water bodies and types, and identification of significant anthropogenic pressures and impacts.

The assessment of ecological status is based on the concept that anthropogenic activities alter the physical and chemical environment, impairing the conditions of the biota, and thus degrade the functioning of a given aquatic ecosystem. By measuring the deviation of the biological parameters from pre-defined reference values, the assessment of ecological status provides an integrated, holistic and powerful tool for surface waters management.

In the WFD, the term ecological status is an expression of the quality of the structure and functioning of aquatic ecosystems and the different degrees of ecological status (i.e. quality classes) are described with narrative criteria (i.e. normative definitions). The normative definitions are the basis for identifying suitable boundary values for each of the indicator parameters (or metrics). The Directive requires that the ecological status is assessed through the analysis of various characteristics of aquatic flora and fauna (quality elements). After selecting the metric or metrics to be used to assess the condition of the quality element, the common interpretation of the normative definition will drive the setting of the boundaries for each metric. Once a boundary has been set up, the monitoring results can be used to classify the condition of the quality element. The results of the ecological quality for each biological quality element need to be expressed using a numerical scale between zero and one, the so called ‘Ecological Quality Ratio’ (EQR).

Most recently, in an initiative to develop guidance on eutrophication assessment under the WFD Common Implementation Strategy (CIS), the Working Group on Ecological Status prepared a report on the interpretation of the WFD concept of ecological status in the context of eutrophication. This report presents a proposal for a common understanding of the WFD’s normative definitions in the context of nutrient enrichment (see European Commission, 2005), which is necessary to underpin the ecological status classification in the context of eutrophication.

It is agreed that as a general rule, aquatic flora quality elements have an earlier response to nutrient conditions than other elements, i.e. benthic invertebrates or fish fauna. Thus, the interpretation of the normative definitions are based on these elements, i.e. the condition of phytoplankton, phytobenthos, macroalgae and angiosperms would not be consistent with good status unless there is a negligible probability (i.e. risk) that accelerated plant growth and/or disturbances in the balance of the taxonomic composition of a plant quality element (see Table 1) would result in a significant undesirable disturbance to the aquatic ecosystem (European Commission, 2005).

A ‘significant undesirable disturbance’ is a direct or indirect measure of anthropogenic impact on an aquatic ecosystem that appreciably degrades the health or threatens the sustainable human use of that ecosystem. A recent ruling of the European Court of Justice of the concept of “undesirable disturbances of the balance of organisms present” states that this concept means species changes involving loss of ecosystem biodiversity, nuisances due to proliferation of opportunistic macroalgae and severe outbreaks of toxic and harmful phytoplankton (European Commission, 2005).
Table 1: Examples of ecologically significant undesirable changes to the balance of taxa (from European Commission, 2005)

<table>
<thead>
<tr>
<th>Moderate conditions</th>
<th>Poor or bad conditions</th>
</tr>
</thead>
<tbody>
<tr>
<td>The composition of taxa differs moderately from type-specific reference conditions such that:</td>
<td>communities are dominated by nutrient-tolerant functional groups normally absent or rare under reference conditions</td>
</tr>
<tr>
<td>• nutrient-tolerant taxa or a functional group of taxa that are absent or rare at reference conditions is no longer rare</td>
<td>• communities are dominated by nutrient-tolerant functional groups normally absent or rare under reference conditions</td>
</tr>
<tr>
<td>• moderate number of taxa are absent or rare compared to reference conditions such that a functional group of taxa is in significant decline;</td>
<td>• one or more functional groups of taxa normally present at reference conditions has become rare or absent</td>
</tr>
<tr>
<td>or</td>
<td>• the distribution of a functional group of plant taxa is so restricted compared to reference conditions that a significant loss of function has occurred (e.g. invertebrates or fish are in significant decline because of the loss of habitats normally provided by functional groups of macrophyte; macroalgal or angiosperm taxa)</td>
</tr>
<tr>
<td>• The condition of the functional group of taxa is exhibiting clear signs of stress such that there is a significant risk of localised extinctions at the limits of its normal distributional range</td>
<td>• a group of taxa normally present at reference conditions is in significant decline</td>
</tr>
<tr>
<td>• a group of taxa normally present at reference conditions is in significant decline</td>
<td>• a group of taxa normally present at reference conditions has become rare or absent</td>
</tr>
</tbody>
</table>

The link of the WFD to biodiversity it is not only indirect through the requirements for assessment of ecological status but there is also a direct link through its requirements for protected areas (Article 4, Annex V, 1.3.5 and Annex VI, Part A). The Directive includes specific requirements regarding protected areas under the Natura 2000 network of sites (i.e. sites designated under the Habitats Directive (HD, 92/43/EEC) and Birds Directive (BD, 79/409/EEC)) to ensure conservation status of habitats and species of community importance. There are, however, cases of potential conflict between the requirements of these three Directives in particular regarding a potential mismatch between the WFD ‘good ecological status’ and the HD and BD ‘favorable conservation status’.

Also, there may be a potential mismatch between the basic units to which the Directives objectives apply, e.g. in the case of the WFD the water body is the basic unit in a river basin to which the environment objectives of the Directive apply and in case of Special Protection Areas (SPA) for birds there might be the need to integrate ecological status over larger areas.

**Thesis 1: The preservation and restoration of biodiversity requires a cross-sectoral approach at regional and catchment scale**

Biodiversity is a theme linking many policies relevant to catchment management. Its management poses opportunities to achieve synergies in meeting requirements of EU directives such as the Water Framework Directive, the Habitats Directive, the European Agricultural Fund for Rural Development (EAFRD) and the EU’s Biodiversity strategy (European Community, 1992; European Community, 2000; European Community, 2005; European Community, 2006). While habitat destruction is responsible for most biodiversity loss in Europe the latest communication from the commission regarding biodiversity: “Halting the loss of biodiversity by 2010 and beyond” has also identified eutrophication (water quality and N deposition) as a threat to biodiversity. To help monitor this, the European Environment Agency is currently developing a series of indicators for trends in biodiversity relating to water quality in aquatic ecosystems. A previous inventory of biodiversity indicators in Europe identified 43 indicators of potential relevance in the water category (EEA, 2004). In Europe’s catchments and river basin districts the EAFRD now allows for payments to farmers in support of Natura 2000 sites and the WFD (Article 36). This clearly promotes a synergistic integrated approach to landscape management linked to achieving targets in sustainability, biodiversity, conservation and water quality.
Thesis 2: The measurement of biodiversity in aquatic ecosystems presents technical challenges

Sampling biological elements in aquatic systems is typically designed to provide a measure of ecological quality and while species richness is a part of this, time and resources usually do not permit an inventory of the species present. Figure 1 shows examples of species area curves for two biological elements required to be monitored by the WFD in lakes: macrophytes and macroinvertebrates. Such species-area curves may be used to estimate whether the sampling effort recovered an adequate amount of species. Typically, such curves show a steep increase in species collected followed by a plateau where additional sampling effort only results in a few additional species being collected (McCune et al., 2002). Figure 1 shows that the typical sampling effort applied of between three and six transects per lake for macrophytes would fall short of providing a correct estimate of species number. Similarly the curve for littoral macroinvertebrates indicates that even after counting 4,464 individual invertebrates from 13 2-minute kick samples the species area curve has not reached a plateau. One method of estimating the ‘true’ number of species in a lake from a sample is to calculate a second order jack-knife estimate. This works by using the number of species that occurred once and twice in the sample (McCune et al., 2002). This method estimated that there was likely to be at least double the number of aquatic macrophyte species and 50% more macroinvertebrate species as detected by our sampling strategy. While many synoptic sampling programmes fail to provide a direct measure of species richness they can at least produce data that are indicative of biodiversity.

![Species area curves for macrophytes and littoral macroinvertebrates](https://example.com/species_area_curves.png)

Fig. 1: Species area curves for macrophytes (four lakes) and littoral macroinvertebrates (one lake). Samples were taken along transects using a grapnel (macrophytes) and using a pond-net kick-sample for 2 minutes (macroinvertebrates). Free et al. (2007).

Thesis 3: Increasing biodiversity can reflect ecosystem degradation as well as recovery

There has been an observed decline in many species in Europe. The EEA (2006) has documented declines in species such as the otter, salmon and several groups of macroinvertebrates as being partly caused by pollution and eutrophication. While the decline in macroinvertebrate diversity with the loss of sensitive species with eutrophication pressure forms the basis for well established assessment systems such as for river macroinvertebrates, it is not the case for all biological groups and other water body types. For example, lake macrophytes tend to show a unimodal response to eutrophication pressure (Murphy, 2002) (Figure 2) and Arts (2002) has suggested that higher diversity in isoetid lakes may be characteristic of an early stage of nutrient enrichment and alkalisation. Declerck et al. (2005) have postulated that this response may partly elicit a similar unimodal response by other biological groups in lakes.
Thesis 4: Changes in biodiversity may be used to set ecological status boundaries and points of intervention for restoration

The Water Framework Directive requires that ecological quality is expressed as one of five classes: High, Good, Moderate, Poor or Bad status. These classes cannot be defined by an arbitrary division of an ecological assessment metric but must conform to the definitions as given in the Water Framework Directive. For example in Ireland, total phosphorus is considered to be the main pressure affecting lakes. Ecologically relevant boundaries of TP were initially chosen based on the response of macrophyte diversity to TP. Figure 2 shows a unimodal relationship between Simpson’s diversity index and transformed (log x + 1) TP for lakes with an alkalinity greater than 20 mg l\(^{-1}\) CaCO\(_3\). Four bands of <10, 10-25, 25-70 and > 70 μg l\(^{-1}\) TP were selected to correspond to points of change in diversity with TP. The good/moderate boundary was taken to be 25 μg l\(^{-1}\) TP on the basis that it corresponds with WFD normative definitions in that it is the point where diversity starts to decrease therefore resulting in an ‘undesirable disturbance to the balance of organisms’. The increase in diversity between 10 and 25 μg l\(^{-1}\) TP may correspond to normative definitions of good status in that the change in not an ‘undesirable’ one. In the context of the WFD the point where macrophyte diversity declines in lakes in response to eutrophication may represent a suitable place to target restoration measures.

![Graph of Simpson's diversity index vs. transformed TP](image)

Fig. 2: Selection of TP bands (---) based on the loess smoothed relationship (→) between Simpson’s diversity index and transformed (Log x+1) TP. Smoothed relationship of chlorophyll \(a\) with transformed (Log x+1) TP is overlain (──). Graph refers to lakes > 20 mg l\(^{-1}\) CaCO\(_3\) only, TP values were mostly measured in Spring. From Free et al. (2007).

Thesis 5: More research needed to understand multiple environmental factors affecting multi-group biodiversity

Owing to the complexity of biodiversity, selection of surrogates for overall biodiversity is often a first step in conservation planning (Margules and Pressey, 2000). However, information on diversity across several biological groups in ecosystems is generally scarce. Although ecosystem productivity is considered to be a crucial factor affecting diversity there is generally poor concordance of diversity in biological groups in freshwater ecosystems. Declerck et al. (2005) concluded that owing to the multidimensionality of taxa richness it is less useful to try and produce a single diversity index for an entire system, rather specific indicators or groups of indicators that reflect the major gradient in richness should be selected, perhaps combining several indicators at ecosystem level. Further research is required to develop such indicators.
Recommendations & perspectives

There is a need for close integration of policy objectives in river basin districts in order to achieve a sustainable use of Europe’s environment and conserve biodiversity. While urgent action to stop habitat fragmentation and destruction is needed to meet the objective of halting biodiversity loss by 2010, further research on the development of indicators of conservation success, system biodiversity and water quality need to be developed and tested.

References


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Eutrophication risks of rivers and streams under the conditions of climate change

Dietrich Borchardt

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Background

The trophic state of rivers and streams may be divided into heterotrophic and autotrophic states (Dodds 2006). This division allows the consideration of whole stream metabolism as a conceptual framework for trophic analyses including the effects of external organic loads as well as nutrients (usually nitrogen and phosphorus). The balance of heterotrophic respiration and autotrophic production has been considered as a fundamental rule of longitudinal zonation patterns of rivers and streams with characteristic shifts from upstream to downstream regions and changing relevance of external nutrient loads related to internal bioproduction (Vannote et al. 1989). As a consequence, the native background trophic state of running waters has a certain variety which has to be considered in assessment methodologies.

The natural metabolic patterns of running waters are superimposed by anthropogenic pressures and diverse responses of receiving waters. It is still an accepted doctrine that the trophic state of running waters is mainly controlled by bottom-up processes (esp. organic or nutrient loads) and hydrological factors (esp. dilution and transport). Therefore, management approaches and mitigation measures usually focus on the reduction of organic carbon and nutrient inputs from point or diffuse sources and threshold values for physico-chemical water constituents. In Europe, this strategy has been very successful with respect to organic pollution caused by municipal wastewater and sewage treatment but has failed to limit nutrient inputs from diffuse sources and excessive primary production in many running waters and streams. In particular, recent inventories under the EU- Water Framework ranked the problem of excessive eutrophication caused by nutrient inputs as a principal reason for failing the environmental objectives for the majority of running water bodies in all EU member states (EEA 2003, Borchardt et al. 2005).

Thesis 1: The cascading effects of eutrophication have to be considered for the achievement of comprehensive risk assessment

There is a long history of effect analyses and classifications of trophic states in running waters. Adverse effects usually mentioned include increased heterotrophic or autotrophic biomass, pH-fluctuations, oxygen supersaturation or depletion, food web effects and changing species composition. However, eutrophication effects do not only have the well known impacts on surface water quality but also affect hydromorphological patterns (e.g. macrophyte growth and resulting hydraulic conditions in streams) and surface-hyporheic exchange patterns. Therefore, habitat features for aquatic biota are shaped by a complex set of processes and effects.

Examples of resulting cascading effects caused by eutrophication have been shown for a 3rd order medium gradient stream and temperate climates by Borchardt and Pusch (2007). Firstly, the vertical exchange velocity between surface flow and the hyporheic zone negatively correlated with benthic chlorophyll biomass for this river. The process behind this relation was identified as biogenic colmation caused by benthic algae biofilms (Ibisch et al. 2007). The biofilms reduced the hydrological exchange which resulted in a lowered oxygen transport to the hyporheic zone, reduced nitrogen turnover (Ingendahl et al. 2007) and therefore limited self-purification capacity. Secondly, benthic biofilms caused peak loads of biomass and organic carbon that infiltrated into the sediments in periods of algae break-downs. The resulting calamities of hyporheic oxygen conditions coincided with sensitive phases in the egg development of gravel spawning salmonids while post-larvae emerging from the sediments were trapped in algae blooms with deteriorated surface water quality (Hübner et al. 2007). Therefore, the cascading effects of eutrophication caused a series of critical phases in the life history of a fish species with alternating calamities of hyporheic and surface water qualities. These have to be known if the abundance and species composition of the salmonid populations would have to be interpreted in relation to other natural or anthropogenic effects in a meaningful way. These
cause-effect relations would be even more important, if an assessment of the “ecological quality” would be based on the fish species composition and age structure as being proposed under the EU-Water Framework Directive and the identification of ecologically effective measures for the achievement of a “good status”.

Enhanced heterotrophic and autotrophic activity induced by external matter inputs may be well separated in lotic systems and result in a decoupling of ecological responses. These can be understood a set of spatial and temporal cascading effects of eutrophication. These shape the habitat features of aquatic biota for major concerns and the ecological effects of cultural eutrophication are much more complex than usually considered. It has to be recognized, that both bottom-up and top-down controls are effective for the limitation of adverse effects in lotic systems.

As shown by the example risk assessment procedures and especially ecologically meaningful interpretation of bioindication assessment approaches as being proposed under the EU-Water Framework Directive would be limited without consideration of the cascading effects of eutrophication.

**Thesis 2: Climate change effects on eutrophication of streams and rivers will be ambivalent**

Eutrophication of rivers and streams is controlled by a series of external and internal factors. Beside increased temperatures, climate change scenarios indicate that the seasonal distribution, frequency and duration of rainfall patterns and droughts will change significantly with a wide range of regional variations. Thus climate change will alter the water cycle, aquatic and semi-aquatic ecosystems and associated biogeochemical cycles on catchment and even water body scales. However, the direction of effects may be different and either increase or mitigate eutrophication (e. g. extended periods with favourable conditions for plant growth vs. increased flushing of algae and macrophytes). With regard to excessive eutrophication and the loss of ecological integrity these effects are ambivalent and to a large extent uncertain.

Climate change may additionally influence land use. Resulting matter fluxes will be key factors for the trophic status of running water ecosystems. Predicted effects of climate change will most likely alter hydrology, water-driven sediment, phosphorus, nitrogen, organic matter and contaminant fluxes at catchment scales and may lead to alterations of running water ecosystems together with the quantity and quality of water resources. Although general relationships between land use and diffuse pollution are known, there are significant gaps in our ability to predict spatially distributed pathways of water, nutrients and contaminants and to assess their site-specific impact on aquatic ecosystems.

Understanding the full implications of climate change effects will require further knowledge of the major external and internal factors that control the trophic state of rivers and streams. More complete comprehension of external and internal interactions that influence trophic state, and determination of climate change effects on regional scales are required as a scientific basis for effective management of eutrophication of lotic waters.

**Recommendations & perspectives**

There is a wide variety of concepts for the ecological assessment of eutrophication in rivers and streams including risk assessment based on land-use patterns, nutrient mass balances and loads, physico-chemical quality standards and bioindication.

These methodologies have to be further developed under the consideration of all relevant processes that control eutrophication in streams and rivers. A process based synthesis is required and comprehensive modelling tools have to be developed that allow for the separation of natural and anthropogenic factors that control eutrophication patterns and their adverse cascading effects in rivers and streams.

Innovative management concepts are required that recognize the limitation of bottom-up controls of eutrophication in cultural landscapes and consider top-down controls as a complementary strategy for the management of adverse eutrophication effects. This concept would open new options for increased efficiency in water resources management including the ecologically efficient mitigation of climate change effects.
References


Research fields:
- Functional ecology of surface waters and ecological modelling
- Integrated water resources management
- Ecological efficiency of measures and river basin management
- Ecological economy and cost-effectiveness in water resources management

Integrated groundwater risk-based management in the context of the WFD

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Groundwater in the context of the WFD

The WFD (Directive 2000/60/EC) [1] is certainly the most advanced regulatory framework for the protection of all (surface and ground) waters that has been developed so far at international level. It is built-up along Integrated Water Resources Management (IWRM) principles, with clear objectives (achievement of ‘good status’ by 2015) to be attained on the basis of specific milestones and operational steps that have to be undertaken by Member States of the European Union. With regard to groundwater, Member States have to implement measures necessary to prevent or limit the input of pollutants into groundwater and to prevent the deterioration of the status of all bodies of groundwater. In this context, Member States have to protect, enhance and restore all bodies of groundwater, ensure a balance between abstraction and recharge, with the aim to achieve good groundwater (chemical and quantitative) status by 2015 as a general principle (taking well justified derogation clauses into account). The Directive also requires the implementation of measures necessary to reverse any significant and sustained upward trend in the concentration of any pollutant resulting from the impact of human activity in order to progressively reduce groundwater pollution.

Under this directive, water management is designed along the development of River Basin Management Plans (RBMP), which should integrate all identified pressures and impacts and identify appropriate programmes of measures necessary to prevent, protect or enhance the status of water bodies. This implies a thorough assessment of risks and a design of appropriate responses which are based on an effective implementation of parent legislations from various sectors (e.g. agriculture, industry, nature conservation etc.). Regarding groundwater, the integrated risk-based management is a stepwise development, imposing on Member States to:

- Delineate groundwater bodies within River Basin Districts and characterise them through an analysis of pressures and impacts of human activity on the status of groundwater. This first risk assessment aims to identify groundwater bodies presenting a risk of not achieving WFD environmental objectives. This characterisation work had to be carried out in 2004-2005 and reported to the European Commission. A report giving a synthesis of Member State’s report has been prepared by the European Commission and made available on the europa website in March 2007.

- Establish registers of protected areas within each river basin districts for those groundwater areas or habitats and species directly depending on water, which had to be carried out in 2004-2005. The registers have to integrate all bodies of water (and related legislations) used for the abstraction of water intended for human consumption [2] and all protected areas covered by the Bathing Water Directive 76/160/EEC [3], vulnerable zones under the Nitrates Directive 91/676/EEC [4] and sensitive areas under the Urban Wastewater Directive 91/271/EEC [5], as well as areas designated for the protection of habitats and species including relevant Natura 2000 sites designated under Directives 92/43/EEC [6] and 79/409/EEC [7]. Here again, registering protected areas is carried out with the aim to properly assess risk and take appropriate measures

- Based on the results of the (risk-based) characterisation phase, establish a groundwater monitoring network providing a comprehensive overview of groundwater chemical and quantitative status, and design a monitoring programme that takes into account the analysis of pressures and impacts.

- Set up a river basin management plan (RBMP) for each river basin district which will include a summary of pressures and impact of human activity on the groundwater status, a presentation in map form of monitoring results, a summary of the economic analysis of water use, a summary of the programme(s) of protection, control or remediation measures etc. The

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1 The views expressed in this paper are purely those of the author and may not in any circumstances be regarded as stating an official position of the European Commission
first RBPM is scheduled to be published at the end of 2009 (after a public consultation to be concluded by the end of 2008). A review is then planned by the end of 2015, and every six years thereafter.

- By 2010, take account of the principle of recovery of costs for water services, including environmental and resource costs, having regard to the economic analysis conducted under Article 5 of the WFD, and in accordance with the polluter pays principle.

- Establish a programme of measures for achieving WFD environmental objectives (e.g. abstraction control, prevent or control pollution measures) by the end of 2009, to be operational by the end of 2012. Basic measures include, in particular, controls over the abstraction of groundwater, controls (with prior authorisation) of artificial recharge or augmentation of groundwater bodies (providing that it does not compromise the achievement of environmental objectives). Point source discharges and diffuse sources liable to cause pollution are also regulated under the basic measures. Direct discharges of pollutants into groundwater are prohibited subject to a range of provisions. The programme of measures has to be reviewed and if necessary updated by 2015 and every six years thereafter.

**The new Groundwater Directive**

While quantitative status requirements are clearly covered by the Water Framework Directive, it does not include, however, specific provisions on chemical status, i.e. the different conceptual approaches to groundwater protection did not allow achieving an agreement on detailed provisions within the WFD at the conciliation. These have been developed in the newly adopted directive 2006/118/EC, which is based on three main pillars, namely:

1. Criteria linked to good chemical status evaluation, which are based on compliance to EU existing environmental quality standards (nitrates, plant protection products and biocides) and to “threshold values” (playing the same role as EQS) for pollutants representing a risk to groundwater bodies. The latter category of standards has to be established by Member States, using common methodological criteria, at the most appropriate scale (national, regional or local), taking account of hydrogeological conditions, soil vulnerability, types of pressures etc. They will have to be reported to the Commission by the end of 2008, and will be used as quality objectives for further compliance checking.

2. Criteria for the identification of sustained upward trends of pollutants in groundwater bodies characterised as being at risk. These include measurement principles and requirements regarding trend reversals.

3. Requirements on the prevention/limitation of pollutant inputs to groundwater, which will ensure a continuity of the 80/68/EEC Directive after its repeal in 2013, i.e. the same principle of prevention of hazardous substances introduction and limitation of other pollutants so as to avoid pollution will apply.

It may be noted that these three pillars all involve risk-based integrated components, requiring multidisciplinary and multisector cooperation. As highlighted below; this is well recognised in the way discussions on the directive’s implementation are conducted in the context of the Groundwater Working Group of the Common Implementation Strategy (CIS).

**A necessary participatory approach**

Indeed, the CIS Groundwater Working Group (C) aims both to clarify groundwater issues that are covered by the WFD and prepare the development of technical guidance documents and exchange best practices on several issues in the light of the orientations of the new Groundwater Directive. The Commission / DG ENV chairs the WG C which is co-chaired by Austria. The Working Group is composed of representatives of EU Member States, Associated and Candidate countries, industrial and scientific stakeholders, and NGO representatives (around 80 members in total).

The focus in the period 2003-2006 has been on the development of technical reports and guidance documents primarily focusing on the issues covered by the WFD, namely monitoring, prevent/limit measures and groundwater protected areas. In addition, a specific activity will concern exchange of views on groundwater management in the Mediterranean area (linked to the EU Water Initiative).
Activities of the WG were conceived with the view of collecting targeted data and information, avoiding duplication with existing guidance documents and ensuring an efficient use of available data and information. Three guidance documents on, respectively, groundwater monitoring, protected areas and measures to prevent/limit pollutant introduction into groundwater have been developed in 2006-2007 and recently endorsed [9-11]. The perspectives for 2007-2009 are to pursue exchanges in support of the implementation of the new Groundwater Directive along the CIS principles, focusing in particular on:

- Discussions on 'land use and groundwater', focusing in particular on agricultural pressures;
- Common methodology for the establishment of groundwater threshold values;
- Compliance, status and trend assessment;
- Recommendations for integrated risk assessment (with close links with the RISKBASE project), including conceptual modelling, and discussions on programmes of measures related to point and diffuse sources of pollution (including megasites).

The activities of the working group and published documents are regularly described in the WISE Newsletter which is published twice a year [12].

Conclusions

An effective groundwater management can only be operational if it is conceived along IWRM principles. The combined implementation of the WFD, its daughter Groundwater Directive and all the parent environmental legislations designed as programmes of measures, is the sole guarantee to enable meeting the good status objectives by 2015. This integrated approach is closely linked to the way risk assessments will be carried out, with related implications for the directive's implementation. Examples are the delineation of water bodies 'at risk' (having an impact on the way monitoring programmes are designed), economic analysis (forming the basis of the future water pricing policy), establishment of threshold values for 'risk substances' (with direct link to good status compliance), etc. The complexity of this management makes it necessary to proceed in a stepwise, iterative, manner, ensuring an effective participation of water actors and a full integration of scientific knowledge.

In this respect, the successful implementation of integrated risk-based groundwater management in the light of the WFD and its daughter Groundwater Directive will closely depend upon an efficient participatory approach and harmonised groundwater risk assessment, monitoring, and programmes of measures throughout the European Union. The CIS Working Group on Groundwater will be an indispensable element supporting this implementation, in particular in view of the preparation of the first River Basin Management Plan expected for publication at the end of 2009. This is to be seen as an opportunity to efficiently manage groundwater resources at EU level and to collaboratively tackle the challenges ahead of us for achieving good quantitative and chemical status of groundwater by 2015.

References


Main speakers' activities are turning around the effective implementation of EU groundwater policy, relying on a participatory approach. This includes close links with scientific projects such as RISKBASE, BRIDGE, AQUATERRA, HARMONICA and many others. Another field of activity concerns technical discussions and guidance on Water Framework Directive' chemical monitoring. The speaker is also responsible for establishing operational links among research and policy in the water and marine sectors.
Threshold values and improving risk assessment for groundwater

Dietmar Müller

Background

Groundwater is a key environmental resource and factor supporting sustainable regional development. This is recognised by the European Water Framework Directive (WFD) and the recently adopted Groundwater Directive. Former concepts aiming at the control of point sources and the protection of groundwater quality at a local scale, are nowadays to be completed by concepts of managing the environmental status of groundwater resources at larger management units in a regional context. Furthermore the environmental objectives defined by the WFD complete groundwater protection schemes by going beyond human use related criteria and recognising functions to support aquatic and terrestrial ecosystems.

Thesis 1: Criteria to assess Quality and Status of Groundwater are different

As for developing groundwater risk assessment towards a comprehensive approach assisting integrated resource management, criteria for quality and status assessment need to be different. To assess risks to groundwater quality is a routine, which may aim at various possible objectives like controlling (prevent or limit) human impacts at local scale, protecting drinking water supplies or conserving pristine conditions. In contrast, groundwater status assessment as requested by the WFD is a new task, lacking experiences and defined procedures, looking out for a holistic assessment of the ‘health’ of groundwater resources at regional scales and linking in aquatic and groundwater dependent ecosystems as ecological receptors. Moreover, natural occurring substances, though considered as pollutants, may be present at specific aquifers in naturally elevated concentrations. Such water may then be considered to have a poor quality (with regard to possible uses, e.g. drinking water abstraction) but still representing a good chemical status (see figure 1).

Thesis 2: Natural attenuation capacities of groundwater systems are important

To assess risks for groundwater and its quality at local scales it is quite common to make use of (natural) background levels (NBLs) and generic reference values, which until nowadays usually have been understood to be human use related.

Risk assessment for groundwater bodies need to consider general chemical quality issues as well as the fate and transport of contaminants and the intrinsic capacities of groundwater systems to cope with
various impacts. Petrographic properties of rocks in the vadose and groundwater saturated zone, regional hydrological and hydrodynamic conditions and hydrogeochemical processes controlling the behaviour of natural and anthropogenic substances are of major importance. Thus status assessment will only bring about an “added value” for groundwater management by considering attenuation criteria (dilution, diffusion, retardation and degradation) in a sound way. These criteria need to be defined specifically for each groundwater body by itself referring to the properties of contaminants, hydrogeological units governed by specific hydrogeochemical processes, and the interaction with surface waters.

**Thesis 3: Threshold values for groundwater status assessment are risk-based**

The definitions provided by the WFD are generally focused on possible impacts on receptors (associated surface waters, dependent terrestrial ecosystems, groundwater as a resource for human uses) and a qualification of the significance of these impacts (e.g. significant diminution of the ecological or chemical quality of surface water bodies). Consequently any environmental threshold applied for groundwater status assessment needs to be derived with a strong orientation to a risk-based approach and the likelihood that receptors are or might be harmed.

**Recommendations & perspectives**

To establish a receptor-oriented status assessment a tiered approach for deriving appropriate threshold values for groundwater bodies is recommended. Tiers may provide intermediate levels based on increasingly detailed understanding of a groundwater body. Therefore each tier will involve increasingly sophisticated levels of data collection and analysis. An Initial Analysis (Tier 1 and 2) can use conservative and rather simple criteria (e.g. Tier 1: check of monitoring data against natural background levels and Tier 2: check against environmental quality standards defined for associated surface waters), whereas further steps of Detailed Analysis (Tier 3 and 4) would mean a thorough evaluation of specific groundwater body characteristics (e.g. Tier 3: back calculation for an associated surface water body taking into account the contribution of the groundwater to the total pollutant burden and Tier 4: taking into account further attenuating capabilities of the subsurface environment, specifically for the aquifer, the hyporeic and the riparian zone). With respect to the heterogeneity of groundwater bodies at the one hand and the limited data availability in practice on the other hand it seems likely that the Detailed Analysis will often be limited to a rather simple third tier by describing groundwater and surface water interaction in terms of quantity relationships.

**References**


BRIDGE reports available through www.wfd-bridge.net

Main fields of interest are investigation, risk assessment and remediation of contaminated sites, and groundwater management; currently involved to 2 European coordination actions under FP6: EURODEMO – topic: acceptance and market entry of innovative technologies for soil and groundwater remediation; RISKBASE – topic: River Basin Management and new concepts for an integrated management of the soil/water/sediment-system a large scales.
Risks to drinking water supply in European river basins

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Background

The Water Framework Directive (WFD) is based upon a global approach to the management of freshwater resources. Today, seven years after its entry into force, besides the benefits to environment but problems linked to its implementation, has the WFD really reduced the risks for drinking water supply in Europe? Providing good quality water in sufficient quantity has been the objective of water utilities for decades and the results are rather satisfactory across Europe. However, the sector has changed tremendously in recent years as a response to contradictory trends: the quality of freshwater resources is continuously improving, mainly as a result of reduction in discharge of classical macro-pollutants and microbiological pathogens by domestic wastewater - while new threats (pesticides, pharmaceuticals, etc…) are constantly emerging, challenging the ability of the operator to fulfil their duties. In parallel, awareness but also consequently, expectations from consumers regarding water quality and safety are continuously growing.

New policy development of risk assessment in drinking water supply

Historical approach to water quality compliance is well encompassed in the Drinking Water Directive 98/83/EC (DWD) whose overall intention is the availability of wholesome and clean drinking water at the tap for European Union’s citizen. The directive mainly tackles water quality, and is based on an obligation of results with reference to a set of limited parametric values.

The limits of this classical approach have been pointed as follows (Medena, 2004)\textsuperscript{1}:

- the DWD is based on a rather limited list of parameters;
- samples under analysis are not representative of the volume of water produced and sensibility of analytical methods is too low;
- when contamination is detected, water has often already reached the consumer.

It is therefore a reactive rather than preventive approach. In addition, this approach can become overly reactionary, provoking requirements for extra monitoring and other studies whenever a “new” chemical is reported detected in raw water.

Since 2003, WHO has introduced risk management through “Water Safety Plans” in its 3rd edition on directives for drinking water quality\textsuperscript{2}. It is based on three main principles:

- Health-based objectives with reference to a tolerable risk;
- “Water Safety Plans”, based on HACCP principles\textsuperscript{3} (risk assessment / risk management), and encompassed also by ISO 22 000;
- Monitoring (water quality as well as satisfaction of the population).

The focus of current and future legislation with regards to drinking water on preventive risk analysis and risk management approach highlights one of the first limits of the approach to WFD. Indeed the latter is mainly concerned with environmental risks, whilst, to a certain extent, health risks can be managed independently of environmental risks. Indeed pathogens are rarely a hazard for aquatic life. Yet the other two main risks exposing population health are toxics released from algae (cf. infra), as a result of nutrient load, and micro-pollutants released in the environment, hence the need to consider the benefits of the implementation of the WFD.

\textsuperscript{1} The interaction between Quantitative Microbial Risk Assessment and risk management in the Water Safety Plan, Microrisk report, Medema & Smeets, 2004.
\textsuperscript{2} References
\textsuperscript{3} Hazard analysis and Critical Control Points
**Resource protection: benefits of the WFD & other coming legislation**

Protection of freshwater resources for drinking water should benefit fully from sound implementation of the WFD, as resource protection provides the first barrier in protection of drinking water quality. More specifically, article 7 of the WFD specifies that Member States have to designate water bodies used for drinking water supply, establish safeguard zones for water catchments areas and provide for measures to protect the water bodies used for drinking water abstraction, in order to reduce the need for treatment.

It should be recognized that the protection of the catchments and water bodies is beyond the responsibility, and legitimacy, of water suppliers. Moreover, water suppliers should not bear the administrative and monitoring burden (including land management or inspection provisions) for designated catchments which should clearly rest with river basin authorities or the relevant competent authority.

River basin authorities should however consult with the water suppliers on the measures to be enforced, with regards to all types of pollution, including diffuse pollution, in relation to the WFD article 7.3. It is clear that water suppliers can create a strong sense of ownership and joint responsibility when developing plans for improving management practices for reducing risks. Indeed, the provision of safe drinking water demands the participation of all stakeholders. Beyond the plants and networks, water suppliers are being given responsibility for developing risk management schemes, including the responsibility for taking the message out to other stakeholders (farmers, customers, highway and rail authorities, industries and other regulators).

Water suppliers have no power to enforce measures on other parties in the water supply chain to take action. And none of these other stakeholders have any obligation to participate in such discussions. While some might be willing to cooperate they inevitably question where the funding is coming from to change their practices. Without this and with no regulatory powers of enforcement the benefits of such liaison are limited in effect.

When setting Preventative Risk Management scheme, water suppliers will increasingly need acquiring an accurate knowledge of the catchments and will need to coordinate with other agencies for the planning and implementation of control measures, for a better governance. Any further transfer of responsibilities to water suppliers should be transparent, properly funded and accompanied by appropriate enforcement means.

**Limits of the WFD in reducing risks for drinking water supply**

Though its implementation is still at early stage, some limits of the Water Framework Directive can already be highlighted.

**System inertia**

The European Commission, during the WISE Conference in Brussels (April 2007), estimated that the results of the implementation of the WFD were “worse than expected”. Many water bodies will not meet their objectives of good ecological and chemical status in 2015. One of the main constraints lays with the inertia of hydraulic systems, in particular aquifers.

There has always been a strong time lag between enforcement of directives and their effects on the environment. This is particularly true with diffuse pollution. Problems that have to be dealt with today relate to practices that took place 15 to 25 years ago, while the actual effects of the measures taken today will result in improvements in another 25 years in many cases.

It is important that the timetables for delivery of programmes of measures and provisions in the “water” directives are properly integrated to allow synergies. The programme of measures and provisions to be completed will all have a positive effect on water resources intended for production of drinking water. They should show flexibility to avoid inappropriate measures and disproportionate costs, avoid duplication of investment, while properly applying the “polluter pays principle”.

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Integration of policies

Truly measuring the positive effects of the implementation of the WFD will need two steps:

- time and efforts at Member States level necessary for a better implementation of existing legislations;
- need for integration of protection and sustainable management of water into other Community policy areas such as energy, transport, agriculture, fisheries, regional policy and tourism.

Obviously, for drinking water abstraction, the integration of policies derived from WFD is at risk on several issues which are further developed in the full paper.

Risks related to water scarcity in Europe

In its Green Paper “Adapting to climate change in Europe – options for EU action”, the commission introduces four pillars and three policy options calling for early action and integrated adaptation. Climate change and the prospects of water scarcity, as well as occurrence of more extreme events (floods and droughts), is clearly a concern for drinking water. However, in this context of global change, one must question the relative importance of the different factors impacting change, when setting proper planning and risk management schemes and their associated decisions. A case study in the South of France (Rinaudo et al., 2007)\(^4\) clearly illustrates the importance of taking fully account of environmental as well as socio-economic trends.

Solutions for minimising risks from climate change lay both in the field of resource development and demand management.

As demand for water increases and water resources become scarcer, aquifer and reservoir storage as well as their management need to be further studied in order to cope with the severe impacts of climate change. It may also become necessary to treat less ‘pristine’ sources of water, such as brackish or saline waters. In some areas of Europe water reuse in its various forms will be necessary and research is needed to ensure that the economics of such schemes are fully understood. Water recycling is certainly one area of regulation which needs developing, ensuring consumers’ health and the environment are being protected.

On the demand side, the main efforts should be devoted to reducing leakage levels in distribution networks. However, water consumption in agriculture but also in watering of public spaces could be significantly optimised and reduced. Some further savings can also be found with individual consumers.

The diversity of possible strategies mentioned and the relatively short time for implementation and resulting effect show that sufficient water resources can be mobilised to face future growth in demand resulting for possible climate change impacts. This again points to the fact that quality constraints should therefore remain the prime focus of water managers.

Cost-benefit analysis

As programs of measures are being drafted, and when assessing how to manage risks related to drinking water, it must be stressed and reminded that the water sector is already faced with major challenges for improvements to wastewater treatment, as well as for the renewal of the assets.

A number of recent directives clearly impact the water sector, the WFD and daughter directives, and focus should be made in integrating them all, rather than make further changes that are likely to destabilize the efforts currently being done. In public health terms, it is unsure that new requirements are necessary to match the expectations, knowing that a permanent improvement cycle is already taking place.

\(^4\) Impact relatif du changement climatique et des tendances socio-économiques sur les équilibres offre – demande en eau : le cas de bassins versants méditerranéens en Languedoc-Roussillon, (…)

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Two aspects must be considered when dealing with drinking water, as opposed to dealing with environmental risks:

- with water quality, one should stress the importance of health based targets when dealing with cost-benefit analysis as any changes to parametric values which may be proposed in the future must be scientifically and health based, practically achievable and deliverable in a reasonable timescale, at a cost which is commensurate with the benefits that will accrue to European citizens;
- maintaining or improving water availability for all uses, including drinking water supply, seems more an environmental challenge. Strategies to maintain supply-demand balance should however be based on the appropriate geographical scale and the relevant mix of actions should consider cost-benefit analysis in the view of maximising environmental as well as social benefits.

**Recommendations and Perspectives**

From the point of view of drinking water supply, the mean basis for risk management is the reduction of health risks, which calls for different strategies than when dealing with environmental risks. One of the key aims of the European directives, but also research, should be the protection of water for public supply, which, in many cases will also contribute to the objective of protecting the environment. Solutions need to be developed especially to reduce diffuse pollution from sources such as agriculture and rural settlements. Risk assessment and management tools and quality-based water abstraction as well as models and early warning systems for pollutant control need to be further developed to improve the protection of raw water catchments. Measures and responses should be adapted in a global planning approach at the right scale and with the appropriate stakeholders.

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As President of Eureau, he has been addressing the subjects of Water Framework Directive impacts on the drinking water industry at European level.
Biological invasions via European inland waterways: Towards development of the risk assessment tool

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Background

European inland waterways have provided opportunities for the spread of aquatic invasive species (AIS) for many centuries. Canals can provide conduits for species to spread between previously separate biogeographic regions either by active movement, drift and/or as a result of ship transport. Over the past century, the potential for species to expand their range has been enhanced as a result of the construction of canals and increased trade. At present, the complex European network of inland waterways is made up of >28,000 km of navigable rivers and canals, connecting 37 countries in Europe and beyond. This aquatic web now connects the previously isolated watersheds of the Caspian Lake, the southern European seas (Azov, Black and Mediterranean sea) and the northern European seas (Baltic, North, Wadden and White sea), to provide corridors for AIS. In Europe there are thirty main canals with >100 branches and > 350 ports (Bij de Vaate et al. 2002, Ketelaars 2004, Galil et al. 2007). Recently, Leuven et al. (2008) deduced that the number of exotic species in the river Rhine in the Netherlands was significantly correlated with the cumulative area of connected river catchments within Europe. There are plans to deepen many of these canals to accommodate larger vessels and to prepare for the lower anticipated water levels arising out of climate changes. These changes are likely to have an influence on the spread of AIS.

There are four principal invasion corridors in Europe (Figure 1):

- **The Northern corridor**: has 6,500 km of navigable waterways and 21 inland ports of international importance, the corridor links the Black and Azov seas with the Caspian Sea via the Azov - Caspian waterway including the Volga-Don Canal, and with the Baltic and White seas via the Volga-Baltic waterway including the Volga-Baltic Canal, and the White Sea - Baltic Sea waterway, including the White Sea - Baltic Sea Canal.
- **The Central corridor**: connecting the Black Sea with the Baltic Sea region via Dnieper and Bug-Pripyat Canal.
- **The Southern corridor**: linking Black Sea basin with the North Sea basin via the Danube-Main-Rhine waterway.
- **The Western corridor**: linking the Mediterranean with the North Sea via the Rhône and the Rhine-Rhône Canal.

Other corridors exist, such as the Canal du Midi linking the Mediterranean Sea to the Atlantic and there are also navigable canals in Britain and Ireland.
The current invasion corridors and projected future developments of European network of inland waterways may highly facilitate the transfer of AIS across European inland waters and coastal ecosystems, which require appropriate risk assessment-based management options to address risks posed by human-mediated introductions of these species (Panov et al. 2007).

**Thesis 1: Conceptual model of qualitative risk assessment of biological invasions for the European inland navigable waterways**

The risk assessment is a part of the process of managing risks, and there are many different risk assessment approaches in different decision-making contexts and levels ranging from specific case studies to strategic regulation and policy making. These approaches can be separated into two major distinct types: quantitative risk assessments and qualitative risk assessments. However, because quantification of risks is not always possible, it is better to convey conclusions (and associated uncertainties) qualitatively than to ignore them because they are not easily understood or estimated. In our opinion, quantitative risk assessments, based on objective scientific judgments can be more applicable for the local level of decision-making in case of site-specific and/or species-specific management, while the strategic regulation and policy making both on national and international levels can be based in large extent on qualitative risk assessment, particularly if one considers the high degree of scientific uncertainty when dealing with such global and complex ecological issue as large-scale intercontinental and intra-continental introductions of AIS (Panov et al. 2007).

The specific methodologies of risk assessment of introductions of AIS include two main types: the environmental matching risk assessment and the species-specific risk assessment. Based on these two principal approaches, we have developed a conceptual model of risk assessment of biological invasions for the European inland navigable waterways, which was further tested for selected assessment units of the Northern, Central and Southern corridors.

We have identified five main components to the qualitative risk assessment of AIS for the navigable inland waterway (Panov et al. 2007): 1. identification of the principle recipient and donor areas of AIS (risk areas, or assessment units) and invasion routes; 2. identification of the main vectors of AIS introductions; 3. assessment of relative inoculation rates (e.g., propagule pressure) and factors determining establishment success of potential invaders (e.g., species life traits and environmental tolerances); 4. assessment of the vulnerability of potential recipient areas to invasions from the past...
patterns and likely environmental suitability; 5. assessment of AIS invasiveness both in the recipient risk area and in potential donor areas based on known dispersal abilities, establishment success and ecosystem impacts.

Qualitative estimations of inoculation rates and ecosystem vulnerability to invasions and species invasiveness were ranked low, medium and high, and these estimations were further used for an assessment of the integrated ecosystem risk level for the main risk areas (assessment units) within the Northern invasion corridor. Also, the analysis of these five main components was conducted for initial predictive risk assessment for selected recipient areas (assessment units) in the Northern and Central invasion corridors.

For example, an initial predictive risk assessment of possible spread and establishment of 34 high-risk AIS in aquatic ecosystems along the Northern invasion corridor has shown the Gulf of Finland (Baltic Sea) to be the most vulnerable ecosystem for invasions, with 17 AIS potentially able to invade and establish in this assessment unit (Panov et al. 2007). Also, 10 high-risk AIS has been predicted as future invaders of the Belarusian part of the Central invasion corridor (Karatayev et al. 2007), and two AIS from this list were recently recorded in this assessment unit of the corridor (Semenchenko et al. 2007).

**Thesis 2: Risk assessment tool for prevention and control of biopollution**

The qualitative approach to risk assessment of intra-continental aquatic invasions, along with qualitative approaches to assessment of impacts of AIS (“biopollution index” suggested by Olenin et al. 2007) involving a range of five levels of environmental impact, may be incorporated into the risk assessment process for European inland waterways. The biopollution approach will enable managers to concentrate on those species that have the potential to be most impacting.

We consider three core components of this risk assessment tool, which is currently under development by international group of experts with support of the European Commission 6th Framework Programme Integrated Project “Assessing LArge scale environmental Risks for biodiversity with tested Methods” (ALARM, contract GOCE-CT-2003-506675) and with support of national projects: the tested protocols of qualitative risk assessment, the distributed online database with essential data needed for assessments, and the online electronic journal “Aquatic Invasions” (ISSN: 1818-5487, http://www.aquaticinvasions.ru). The journal is serving as informational tool for support of the distributed database via protection of author rights on primary information on distribution and biology of AIS, and also as an effective early warning instrument (Panov and Gollasch 2006).

**Recommendations & perspectives**

Recommended management options, derived from initial risk assessments of biological invasions via European inland corridors, include treatment of ballast water and sludge, prevention of hull fouling and removal of hull fouling which should be undertaken before entering the main inland waterways. Control and reduction of the dispersal of AIS along inland waterways may also entail installation of barriers such as deterrent electrical systems, chloride or pH-altered locks. Our qualitative approach to risk assessment of aquatic invasions may serve as a useful tool for the integrated biopollution prevention and control in the European inland waterways.

**Acknowledgements**

This study has been supported by the European Commission 6th Framework Programme Integrated Project ALARM (contract GOCE-CT-2003-506675) and Strategic Targeted Research Project DAISIE (contract SSPI-CT-2003-511202).

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Dr Vadim E. Panov is a senior research scientist at the St. Petersburg State University, Russia. His research interests include development of approaches to management of risks posed by introductions of aquatic invasive species. Currently he is involved in the EC FP6 Integrated Project ALARM and Strategic Targeted Research Project DAISIE, coordinating research activities related to biological invasions in European inland waters. Since 2006, he is serving as Chief Editor of a new European online journal “Aquatic Invasions".
Socio-economic assessment of risks of invasive species to freshwater ecosystem goods and services on a basin scale

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Background

Biological invasions are considered the second major cause of biodiversity loss (Vitousek et al., 1997), particularly in freshwater ecosystems (García-Berthou et al., 2005). Human agency in biological invasion involves the driving forces (Perrings, 2000; Kowarik, 2003), the perception of impacts (Levine et al., 2003) and the implementation of responses, as well as the conceptualisation of the notion of invasive species (IS) as an environmental and socioeconomic problem (Shrader-Frechette, 2001). This leads to the need of incorporating the socioeconomic perspective in the assessment of the issue (Pimentel et al., 2001; 2005; Perrings et al., 2000). The objective of this communication is to develop an integrative framework to assess the impacts of aquatic invasive species from a socioeconomic perspective. For achieving this aim, we propose the utilisation of the ecosystem services approach as a key element for structuring the information on impacts.

Thesis 1: The socio-economic assessment contributes to examine the likelihood of bioinvasions and the value of their outcome

Wynne (1992) distinguishes the notion of risk from those of uncertainty, indeterminacy or ignorance. Risk assessment is thus delimited to systems which behaviour is known, and the different outcomes can be quantified by structured analysis of probabilities. The complexity of the socio-environmental processes involving IS makes bioinvasions rather a matter of uncertainty or even indeterminacy (Rauschmayer, 2003). But socioeconomic evaluation contributes at least in two ways to approach the likelihood of the introduction and establishment of invaders, and the outcome of such establishment.

1. By assessing drivers of transport, release and establishment of IS.

A wide scope of scientific literature analyzes the probability of IS arrival and establishment focussing on the invasiveness of the species or invasivility of the ecosystem (Lonsdale, 1999; Rejmanek, 1996). However, socio-economic processes are powerful mechanisms of introduction and spread that usually escape from modelling (Walker et al., 2002) and thus are ignored or underestimated in the analysis (Kowarik, 2003). Two empirical examples based in participatory research at the basin scale can be mentioned at this regard:

a) The invasion of zebra mussel (Dreissena polymorpha) and alien fish species like Wels catfish (Silurus glanis) in the Ebro River (NE Spain) were attributed to changes in the configuration of the basin due to the creation of dams, and the growth of recreational activities. External users of the river press local conditions by demanding transformations in the watershed features and increasing the affluence of transport (Rodriguez-Labajos, 2006).

b) The invasion of hydrilla (Hydrilla verticilata) in the Izabal Lake (Guatemala) also illustrates the issue of social processes shaping the distribution of the species. While the specific pathway of the introduction remains unclear - likely related to sailing - factors influencing the spread are definitely linked with inadequate fishing activities and land-use and practices (Monterroso, 2005).

2. By assessing factors that influence the perception of the impacts.

Invasive species are held responsible for causing environmental and socio-economic impacts. The value of such impacts is a relevant indication in the perception of risk by those social actors involved in the different stages of the invasion process. An alternative for valuing IS impacts is estimating either the control costs or the damage costs of the invasion (Pimentel et al., 2001; 2005). While having the advantage of providing a clear signal about the magnitude of the impact, current studies on the monetary impacts have shortcomings and their results should be carefully used (Born et al., 2005).
A key element to take into account when managing risks of bioinvasions is the assessment of the multidimensional perception of impacts. The ample range of impacts requires an inclusive framework to assess them, as a basis for an integrated assessment for the management of invasive species. A way to display such assessment is discussed in length in Thesis 2.

**Thesis 2: The ecosystem services framework allows displaying and structuring information on impacts in a comprehensive way.**

From a socio-economic perspective, impacts caused by invasive species are changes of recipient ecosystems which are perceived by humans. In that sense, by affecting the ecological processes at the genetic, species and ecosystem level, biological invasions modify the provision of ecosystem services (Charles and Dukes, 2007; Binimelis et al., 2007). For the present communication, The Millennium Ecosystem Assessment (2003) classification of ecosystem services will be used. It covers four categories: a) Supporting: those services needed for the production of all other ecosystem services, e.g. photosynthesis; b) Provisioning: products obtained from ecosystems, e.g. food or freshwater; c) Regulating: benefits supplied by self-maintenance properties of ecosystems, e.g. water quality regulation and d) Cultural: non-material benefits derived from ecosystems, e.g. aesthetic values. A fifth category, impacts caused by aquatic invasive species to human-made infrastructures and utilities, is added. The approach was implemented for the analysis of fifteen freshwater aquatic organisms in Europe, using datasets from the ALARM and DAISIE projects. They include 10 invertebrates, 3 vertebrates and 2 aquatic plants.

Table 1: Impacts of freshwater aquatic organism to ecosystem services in Europe.

<table>
<thead>
<tr>
<th>Ecosystem service</th>
<th>Impact description / effect</th>
<th>Species</th>
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<tbody>
<tr>
<td>Nutrient cycling</td>
<td>Alteration of food and oxygen availability</td>
<td>New Zealand pigmyweed (<em>Crassula helmsii</em>)</td>
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<td>Zebra mussel (<em>Dreissena polymorpha</em>)</td>
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<td>Changes in primary production</td>
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<td>Red swamp (<em>Procambarus clarkii</em>)</td>
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<td>Habitats stability</td>
<td>Changes in community structure</td>
<td>Canadian waterweed (<em>Elodea canadensis</em>)</td>
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<td>Refugia</td>
<td>Freshwater hydroid (<em>Cordylophora caspia</em>)</td>
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<td>Round goby (<em>Neogobius melanostomus</em>)</td>
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<td>Red swamp (<em>Procambarus clarkii</em>)</td>
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<td>Food</td>
<td>Loss or gain in commercial production and harvest (fisheries, aquaculture)</td>
<td>Zebra mussel (<em>Dreissena polymorpha</em>)</td>
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<td>Chinese mitten crab (<em>Eriocheir sinensis</em>)</td>
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<td>Salmon fluke (<em>Gyroactylus salaris</em>)</td>
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<td>Comb jelly, sea walnut (<em>Mniopsis leidyi</em>)</td>
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<td>Stone moroko (<em>Pseudorasbora parva</em>)</td>
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<td>Swim-bladder nematode (<em>Angullicola crassus</em>)</td>
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<td>Genetic resources</td>
<td>Threat to the viability of endangered / native species</td>
<td>Zebra mussel (<em>Dreissena polymorpha</em>)</td>
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<td>Red swamp (<em>Procambarus clarkii</em>)</td>
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<td>Brook trout (<em>Salvelinus fontinalis</em>)</td>
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<td>Water regulation and purification</td>
<td>Genetic hybridization</td>
<td>Freshwater hydroid (<em>Cordylophora caspia</em>)</td>
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<td>Killer shrimp (<em>Dikerogammarus villosus</em>)</td>
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<td>Brook trout (<em>Salvelinus fontinalis</em>)</td>
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<td>Biological control</td>
<td>Reduction of native species through displacement, predation and resource competition</td>
<td>Zebra mussel (<em>Dreissena polymorpha</em>)</td>
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<td>Fish-hook waterflea (<em>Cercopagis pengoi</em>)</td>
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<td>Brook trout (<em>Salvelinus fontinalis</em>)</td>
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<td>Disease regulation</td>
<td>Infection of native fauna/flora</td>
<td>Zebra mussel (<em>Dreissena polymorpha</em>)</td>
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<td>Fish-hook waterflea (<em>Cercopagis pengoi</em>)</td>
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<td>Red swamp (<em>Procambarus clarkii</em>)</td>
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</tbody>
</table>
### Ecosystem service | Impact description / effect | Species
--- | --- | ---
**Erosion regulation** | Intensification of soil/river banks erosion or viceversa | Red swamp (Procambarus clarkii)
**Cultural services** | Changes in recreational use of natural sites | Zebra mussel (Dreissena polymorpha)
 | | New Zealand pigmyweed (Crassula helmsii)
 | | Canadian waterweed (Elodea canadensis)
 | Affectation to eco-tourism activities | Zebra mussel (Dreissena polymorpha)
 | | Brook trout (Salvelinus fontinalis)
**Costs to human well-being** | Damage to infrastructures and utilities | Zebra mussel (Dreissena polymorpha)
 | | Canadian waterweed (Elodea canadensis)
 | | Chinese mitten crab (Eriocheir sinensis)

Structuring the information about impacts using the ecosystem services categories means that:

- All the available information is displayed in an integrated manner. Aquatic invasive species cause impacts in all ecosystem services categories, being an important driver of change. The range of impacts differs among species. Some species affect all dimensions of ecosystem services (e.g. *Dreissena polymorpha* or *Salvelinus fontinalis*), while others, as *Mnemiopsis leidyi*, are characterised by causing very specific impacts, although perceived as highly damaging.

- The non-aggregated display of information allows reflecting on uncertainty and lack of scientific knowledge, which is important both for decision-making and the design of research objectives.

- The links between ecological and SE dimensions of impacts are shown. Impacts on ecosystem services and are not only manifold but also complex, as can be illustrated with the example of the brook trout (*Salvelinus fontinalis*). It competes with other native salmonids and predates on amphibians, zooplankton and other invertebrates and causes impacts to endangered species as the freshwater pearl mussel (*Margaritifera margaritifera*) but has a positive economic impact for local communities due to sport fishing.

### Thesis 3: A multidimensional view of IS impacts can foster their management

The analysis presented above permits the distinction between direct (those caused by invasive species on ecosystem functions and human well-being) and indirect impacts (those derived from the implementation of response actions). The specific way is shown in Fig.1. This differentiation is important for designing the management strategies. In that sense, the identification of the impacts caused by invasive species allows evaluating the consequences of invasive processes and also the social agents linked to them, informing decision-making processes and supporting the implementation of management strategies (Binimelis et al., 2007).

Once organized, the information can be transferred to an impact matrix comparing different scenarios or states of nature. There the different impacts can be described through different units of measure. The information may be employed then in several ways according to the aggregation procedure. If total compensation is accepted, indexes can be obtained. If partial or no compensation is accepted
when aggregating impacts, still some ranking of alternatives may be obtained though the use of multi-criteria out-ranking methods. Other option is employing the impact matrix for accompanying processes of social deliberation around risks of biological invasions.

**Recommendations & perspectives**

Analyzing socioeconomic pathways and driving forces of biological invasion at different scales, including the basin scale, may help to understand those uncertain aspects of biological invasions that otherwise can be interpreted as ‘surprises’. The analysis of impacts with an integrative manner - as the ecosystem services framework - can help to prioritize strategic policies and actions, as well as to identify the potential stakeholders involved in the invasion process (as driving forces, affected or as the ones implementing or being influenced by the responses).

**References**


Rosa Binimelis Adell. Her research interests include ecological economics, socio-economics of biodiversity and ecosystem services as well as methods for participatory research. She is currently involved in the ALARM project (GOCE-CT-2003-506675).
Concepts for aquatic risk assessment of pesticides and ecological protection

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Background

In Europe a large part of the land surface is agricultural. Since this land surface forms the catchment area of lotic and lentic freshwater ecosystems, pollution of these surface waters with agricultural pesticides is a well-recognised environmental problem (Schulz 2004). Therefore, European authorities have set criteria to protect aquatic life from pesticide-stress. These criteria are described in the Plant Protection Product Directive (91/414/EEC) and the Water Framework Directive (2000/60/EC). The Water Framework Directive (WFD) has its main focus on relatively large bodies of water and, among other goals, aims to produce predicted-no-effect concentrations (PNECs) for all types of toxic chemicals that pose risks to aquatic ecosystems. In principle, measured concentrations of pesticides at WFD monitoring sites should not violate these PNEC values. It is, however, not clearly described in the WFD whether the officially derived water quality standards for pesticides other than the priority (hazardous) substances should always be met at all localities of freshwater in Europe (e.g. drainage ditches, ponds, small streams). The protection of water organisms within the context of Directive 91/414/EEC has its focus on water bodies that directly border agricultural fields. This Directive states that Member States shall ensure that the use of plant protection products does not have any long-term repercussions for the abundance and diversity of non-target species. In other words, the potential for recovery of sensitive populations may be taken into account when deriving acceptable concentrations in watercourses that are part of the agricultural landscape.

Thesis 1: Regulatory documents that deal with the aquatic risks of pesticides are based on ambiguous protection goals, particularly when considering their temporal and spatial context

According to Brock et al. (2006) four completely different perceptions of the ecological risks of toxicants in non-target habitats can be recognised, viz.:

1. The Pollution Prevention Principle. This principle presumes that all environmental pressure is potentially harmful. Consequently, use and emission of chemicals to non-target sites should be prevented as much as possible. An option in line with this principle is always to use conservative procedures to assess risks of chemicals with PBT properties (= Persistent, Bioaccumulative and Toxic) and/or that are very mobile in the environment. The derivation of EQS values for priority (hazardous) substances within the context of the WFD tends towards this approach.

2. The Ecological Threshold Principle. This principle assumes that the environment can absorb and tolerate a certain amount of chemical-stress before sensitive species are impacted, that the integrity of community structure depends on these sensitive species, and that protecting community structure protects ecosystem functioning. This principle may be used in deriving water quality standards for river-basin specific pollutants that are not classified as priority (hazardous) substances.

3. The Community Recovery Principle. This principle assumes that chemical-stress should be limited to a concentration that causes reversible impacts on the most sensitive populations of the community, and that recovery takes place within an acceptable time frame. Effects of chemicals that are restricted in space and time may be regarded, in certain habitats, as ecologically unimportant when they are of a smaller scale than changes caused by other natural or anthropogenic stresses. The “unless” clauses formulated within the Uniform Principles (Directive 91/414/EEC) tend towards the Community Recovery Principle, at least for the multifunctional freshwaters adjacent to agricultural fields.

4. The Functional redundancy Principle. This principle presupposes that, for sustainable functioning of an ecosystem, a decrease in biodiversity can be tolerated, as long as key(stone) species and their function are not impacted. The emphasis is on ecosystem
processes and impacts are considered acceptable when functional attributes are not changed, despite possible effects on community structure. The Functional Redundancy Principle may be used to evaluate the acceptability of the impact of chemicals in intensively used environments, such as agro-ecosystems for fish/prawn culture and for the growth of crops like rice and watercress.

It might be argued that a differentiation in the protection level may contribute to a more focused risk assessment and management of pesticides that takes into account perceived differences in functionality and intrinsic value of surface waters. The four protection goal principles presented above may facilitate the “acceptability” debate and can be used as options in the communication between risk assessors and risk managers, as well as between these risk experts and other stakeholders.

**Thesis 2: The interaction between the assessment of exposure and ecotoxicological effects in the risk assessment procedure is at a lower level of sophistication than either assessment of exposure or assessment of ecotoxicological effects**

A proper risk assessment of pesticides (and other toxic chemicals) consists of two parts; (i) assessment of effects to aquatic organisms derived from ecotoxicological experiments (= effect assessment), and, (ii) assessment of concentration levels in relevant environmental compartments by means of appropriate chemical monitoring or prediction by fate models (= exposure assessment). According to Boesten et al. (2007) current regulatory procedures lack a clear conceptual basis for the interface between the effect and exposure assessments, which may lead to a low overall scientific quality of the risk assessment. This interface is defined by Boesten et al (2007) as the type of concentration that gives a useful correlation to ecotoxicological effects and is called the Ecotoxicologically Relevant Concentration (ERC). The same type of ERC should be used consistently for both the exposure in ecotoxicological experiments and that related to exposure in the field.

A clear definition of the ERC is important because it provides the interface between the effect and exposure assessment, and thus the interface between two different fields of scientific expertise (ecotoxicology and environmental chemistry). Scientists from these two different disciplines speak “different languages” which may easily lead to confusion. To avoid this confusion the definition of the ERC has to be expressed in terms of relevant spatio-temporal scales (e.g. “maximum in time” and “interstitial water in upper 5 cm of sediment”; “highest 7-day time weighted average concentration” and “integrated column of overlying water”) (Boesten et al. 2007).

For pesticides, that usually show a high spatio-temporal variability in exposure concentrations, it will be hard to derive a proper ERC on basis of chemical monitoring data. Because of financial constraints field monitoring programmes likely fail to record the real maximum peak concentrations of pesticides. In addition, it is doubtful whether an annual average pesticide concentration really is ecotoxicologically relevant for pesticides characterised by time-variable exposure regimes and short time-to-effect periods. For this reason it can be stated that the WFD does not follow a risk assessment procedure but a hazard assessment approach. Furthermore, it seems that when deriving the short-term (MAC) and long-term water quality standards (AA-EQS) within the context of the WFD, predominantly a conservative first-tier approach is followed to compensate for the weak exposure estimates. In contrast, procedures under the umbrella of Directive 91/414/EEC are more in line with a risk assessment approach since “realistic worst-case” time-variable exposure concentrations are predicted for 10 FOCUS scenarios (representing different European climate and landscape conditions) using mechanistic models for describing emissions to and behaviour of pesticides in surface water (ditch, stream, pond). In addition, under the umbrella of 91/414/EEC several different methods for higher-tier effect assessment are developed and experimentally validated (e.g. Species Sensitivity Distribution approach on basis of the most sensitive taxonomic groups; population studies; micro/mesocosm experiments; effect models) (Brock et al. 2006; Boesten et al. 2007).
Thesis 3: For a cost-effective and accurate risk assessment of pesticides the specific toxic mode-of-action of these chemicals should be taken into account

Agricultural pesticides are chemicals deliberately released into the environment to control pest species that harm agricultural crops. These agrochemicals can only be used if they do not harm the crop, crop rotation, and beneficial organisms in agro-ecosystems (e.g. bees and earthworms). For this reason the modern pesticides developed by the agrochemical industry usually are characterised by a specific toxic mode-of-action. Aquatic ecosystems, however, contain species related to the target organisms of pesticides. It appears, for example, that in particular aquatic arthropods are most sensitive to insecticides (Maltby et al. 2005), while algae and aquatic vascular plants are more sensitive to most herbicides than other taxonomic groups (Van den Brink et al. 2006). In contrast, several fungicides are characterised by biocidal properties in that representatives of several taxonomic groups may be sensitive (Brock et al. 2006). It needs without saying that the specific toxic mode-of-action of particularly insecticides and herbicides should not be ignored when assessing the risks of these chemicals by means of the Species Sensitivity Distribution (SSD) approach and when selecting measurement endpoints in the conduct and evaluation of micro/mesocosm studies. Both the Water Framework Directive (WFD) and Directive 91/414/EEC allow the SSD approach to derive acceptable concentrations. However, different criteria are used in the construction of the SSD (number of taxa and taxonomic groups). The SSD procedure used under the umbrella of the WFD initially seems to ignore knowledge on the specific toxic mode-of-action of pesticides, which may lead to unnecessary animal testing and a less cost-effective risk assessment approach.

Thesis 4: In pesticide-stressed freshwaters the threshold levels for effects (NOEC_community) can be predicted with lower uncertainty than ecological responses caused by exposure concentrations well above these threshold levels for direct toxic effects

An important question at stake in the aquatic risk assessment of pesticides is whether lower and higher-tier toxicity data can be extrapolated in space and time. The Species Sensitivity Distribution (SSD) analysis conducted to date suggest that, although the composition of freshwater communities varies across biogeographical regions, climate zones, and habitat types, the distribution of species sensitivities to pesticides does not vary markedly. Maltby et al. (2005) assessed the influence of habitat (lentic vs. lotic) on the SSDs of arthropods to eight insecticides (carbaryl, chlorpyrifos, diazinon, fenitrothion, lambda-cyhalothrin, lindane, parathion-ethyl, permethrin). There was no consistent pattern in the relative sensitivity of lentic or lotic species and no evidence of a significant difference in HC5 (Hazardous Concentration to 5% of taxa) values among and within compounds. Therefore, there is no evidence to support the contention that the ecological risk assessments for pesticides must necessarily be based on indigenous species occurring in the watershed of concern, at least when interested in threshold concentrations of effects.

There is no question that freshwater ecosystems differing in size, complexity and community composition will also respond differently to high levels of pesticides. This may be caused by differences in fate and dynamics in exposure concentrations but also by the fact that at higher exposure concentrations also indirect effects and recovery processes are involved. Under conditions where exposure concentrations to pesticides are similar, however, it seems that threshold concentrations for direct toxic effects are more or less similar in different types of freshwater micro/mesocosms (including lentic and lotic test systems), at least when they contain enough representatives of sensitive taxonomic groups. This also accounts for similar types of micro/mesocosms that were treated in different periods of the year (e.g. Van Wijngaarden et al. 2005).
Recovery potential of affected populations (exposed to concentration levels well above the threshold level for direct toxic effects) is important in case exposure to the toxicant is not constant due to fast dissipation processes. Recovery rate, however, is highly dependent on life-cycle characteristics of the affected species of concern (e.g. generation time, offspring number, presence of dormant life stages, migration potential) and on the ecological infrastructure of the surroundings (e.g. presence of refuges and uncontaminated habitats). In other words, at temporal exposure concentrations well above the threshold level, the vulnerability of a population is not determined by its sensitivity to the pesticide alone.

**Recommendations & perspectives**

The most important recommendation of this paper is that when assessing aquatic risks of pesticides the problems associated with protection goals, linking exposure and effects, specific toxic mode-of-action and spatio-temporal extrapolation of effect data should be appropriately addressed. The thesis above mainly addressed the risk assessment of single pesticides, while multi-stress impacts may be a common phenomenon at the watershed level.

There are different, complementary approaches to assessing ecological risks of pesticides at the watershed level. The reductionist approach aims at identifying risks to populations and communities of concern on the basis of accumulated data on simple stressor-effect relationships, and by identifying the main stressors of concern. When interested in evaluating the relative toxicity and the occurrence patterns of pesticide mixtures in water courses of agricultural watersheds, an inventory of the pesticide use patterns in dominant crops in agricultural landscapes may be used to design experiments to study the combined effects of a realistic combination of pesticides in aquatic ecosystems (e.g. Arts et al. 2006). These studies seem to indicate that mixture toxicity plays a relatively small role, since most modern pesticides show a relatively fast dissipation rate in drainage ditches and streams. Also Belden et al. (2007) conclude that pesticide mixtures in streams within the “American corn-soybean pesticide landscape” tended to be less complex than would have been predicted on the number of pesticides used. Most probably, repeated stress to pesticides plays a more important role than mixture toxicity to these chemicals.

An example of a more holistic approach to assess the possible effects of pesticide-stress in freshwater ecosystems is the “species at risk (SPEAR)” concept (Liess and Von der Ohe 2005). The SPEAR concept emphasizes the importance of considering ecological traits and recolonization processes on the landscape level for ecotoxicological risk assessment.

To improve the interpretation of monitoring programmes and “ecosystem health” indicators to assess risks of pesticides in watersheds, the existing information on pesticide use patterns and aquatic biodiversity should be appropriately mined and data gaps filled, e.g. on their ecological traits. In the prediction of site-specific ecological impacts of pesticide-stress it is important to use more than one indicator to increase the discriminatory power of identifying impaired sites and to reduce the possibility of “false-negatives”. The most powerful application of watershed ecotoxicology are when experimental studies, field monitoring and model simulations are integrated in a weight-of-evidence approach.

**References**


Theo Brock participated in various ecotoxicological workgroups of the Health Council of the Netherlands, and in the organising committee of European workshops on higher-tier risk assessment of pesticides (e.g. HARAP, CLASSIC). In 2005 – 2006 he was ad hoc member of the Panel on plant health, plant production products and their residues of the European Food Safety Authority. He is regularly invited to participate as a lecturer in (post)university courses on ecological risk assessment in the Netherlands and abroad and asked to advise governmental authorities, the Dutch Board for Registration of Pesticides and the chemical industry.

Currently he is one of the Founding Editors of the scientific journal *Integrated Environmental Assessment and Management* (IEAM) and chair of the organising committee of the EU Workshop on Linking Aquatic Exposure and Effects in the Registration Procedure of Plant Protection Products (ELINK).

**Key qualifications:**
- the ecology of freshwater ecosystems
- vegetation science and wetland ecology
- microcosm/mesocosm research
- the species sensitivity distribution approach
- ecological risk assessment of pesticides and other contaminants in freshwater ecosystems
Probabilistic risk assessment of pesticides

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Background

Probabilistic risk assessment (PRA) of chemicals is based on probability functions of exposure and effects data as opposed to deterministic single-value approaches (Solomon et al. 2000). It is thus anticipated that PRA is more realistic as it includes a wider range of factors determining either the exposure or the effects side. In Germany, there is an ongoing debate mainly involving the Umweltbundesamt (UBA), the Bundesamt für Verbraucherschutz und Lebensmittelsicherheit (BVL), the Biologische Bundesanstalt (BBA) and the Industrieverband Agrar (IVA) whether PRA should be introduced for pesticides. Over the last few years, a considerable amount of work has been conducted mainly by IVA and BBA in order to develop GIS tools for exposure assessment in permanent crops (vine, orchards, hops). In January 2007, the current state of the art and the results of a research & development project have been summarized and discussed in a workshop conducted at the UBA Schulz et al. 2007.

The project had a focus on permanent crops and spray drift as a route of entry into surface waters. A total of 23 exposure-determining factors were identified and it was assessed, whether the scientific data allows their inclusion in the PRA. The suggested PRA concept itself is comprised of the following four steps:

1. Nationwide risk assessment, preferably based only on georeferenced factors
2. Hot-Spot-Analysis, including the spatial extension of contamination, the level of contamination and the tolerable effect levels
3. Refined exposure assessment, using aerial photographs and field surveys
4. Mitigation measures, with a focus on landscape-level active measures leading to effective risk reductions.

The suggested PRA concept offers the possibility to actively involve the farming community in the process of pesticide management while securing the high protection level of surface waters.

Thesis 1: A probabilistic risk assessment approach better represents reality

The current deterministic exposure assessment of pesticide spray drift assumes a virtual scenario using deterministic drift deposition values and an adjacent standing water body of 1 m width and 0.3 m depth to calculate an exposure value. This scenario is considered a considerable worst case. However, under natural conditions, many surface waters differ in their features and a large number of additional parameters contribute to the exposure of surface waters via spray drift (Fig. 1). Inclusion of these factors in the framework of a PRA considerably increases the realism on which the assumptions are based. Many, yet not all of the factors shown in Fig. 1 can be derived from regularly available governmental geodata. Others have to be monitored in the field. The use of representative areas and subsequent upscaling to larger landscapes based on distribution functions is a potential methodological approach.

Thesis 2: Probabilistic assessment should always include risk management

An important feature of a PRA concept is that it should be linked with measures to reduce identified risks. As the PRA allows for a more realistic representation of the exposure situation many rather restrictive regulations currently governing agricultural practice may not be needed any further. On the other hand, risk need to be addressed in those areas which still raises concern. It is recommended to use mainly active risk mitigation measures such as hedges or implementation of constructed wetlands (Schulz & Peall 2001; Schulz 2004) instead of passive measures such as no-spray field margins.
**Recommendations & perspectives**

- Probabilistic exposure assessment for pesticides will provide a more realistic picture.
- However, any probabilistic approach will require field validation using chemical (and potentially even biological) monitoring.
- Probabilistic risk assessment approaches should be directly linked with risk management options.

![Diagram of Factors affecting the spray drift exposure of surface waters from pesticide applications in permanent crops.](image)

**References**


The overall research interest of the Institute for Environmental Sciences, Landau, Germany is the assessment of anthropogenic stress in ecosystems. The current work concentrates on pesticide exposure and effect assessment as well as mitigation strategies for aquatic and terrestrial ecosystems (e.g. EU ArtWET). Molecular genetic techniques are used for ecosystem management purposes.
Uncertainty in ecological and human risk assessment: Implications for risk-based management in river basins

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Background

Environmental risk assessment is an organized process which aims to describe and estimate the likelihood of adverse health outcomes after exposure to environmental stressors. The outcome is a qualitative or quantitative description of risk, which is typically uncertain due to our limited knowledge about the physical, chemical and biological processes underlying the risk. This paper addresses the implications of uncertainty in risk-based management in river basins. It focuses on chemical substances, but the findings are equally applicable to other stressors.

![Fig. 1: Schematic presentation of the risk assessment process.](image)

Risk Assessment

Figure 1 provides a highly simplified, schematic presentation of the processes involved in risk assessment. It consists of four boxes: the pollution source, environmental compartments, exposure and adverse effects. The first three boxes are connected by processes such as the dispersion of the pollutant in the environment and the behaviour of the exposed organisms. The aim of exposure assessment is to quantify the extent of exposure, either by direct measurements or by modelling the underlying dispersion and behavioural processes based on emission data. The exposure and effects boxes are connected by the toxicokinetic and -dynamic processes that occur in the exposed organism after exposure. These processes are often described quantitatively by a dose-response relationship. The aim of effect assessment is to quantify the probability and extent of effects after an exposure has occurred.

The flow of events in Figure 1 runs from the left to the right. In the risk assessment process, this flow of events can be followed to estimate the likelihood of adverse effects (risk). This means a dose-response model is used in the last step. However, this is the exception rather than the rule. Often, emissions, environmental concentrations or exposure levels are compared directly with risk-based reference values or standards to obtain an indication of the risk (Leuven & Poudévigne, 2002). An example is the widespread use of the PEC/PNEC ratio: the ratio between the Predicted Environmental or Exposure Concentration (PEC) and the Predicted No Effect Concentration (PNEC). It should be stressed that values such as the PEC/PNEC ratio are risk indicators rather than risk estimates; if the ratio exceeds unity adverse effects may occur, but the likelihood and severity of these effects remain unknown.

Uncertainty

Risk assessment can be uncertain for several reasons. Here, we distinguish problem definition uncertainty, a lack of knowledge and variability. Problem definition uncertainty arises when the definition of the management problem differs from the scientific problem. For example, the management problem may be the protection of the aquatic ecosystem. This problem can be translated into several scientific problems; one being the determination of safe concentration levels for individual chemicals. Once these safe levels have been established, the management problem has been solved only partially. Even when the safe levels are met, the aquatic ecosystem may still be affected due to mixture effects or other stressors (e.g. low water levels) not considered in the scientific analysis.
Uncertainty due to a lack of knowledge is also referred to as true uncertainty. The level of knowledge can vary from complete ignorance to imprecision. Examples of true uncertainty are (model) assumptions, simplifications, approximate equations and imprecise measurements. Stochastic variation (e.g. the chance that an individual exposed to benzene develops a tumour) can also be considered a form of true uncertainty, because most stochastic events can be described mechanistically if the underlying processes are known.

Variability is a phenomenon of the real world, e.g. variation of pollutant concentrations in space and time. Another example is the variation between the individuals of a population. An important distinction between true uncertainty and variability is that true uncertainty can be reduced by additional research, but variability cannot. It can only be described. Variability becomes an issue in risk assessment when aggregate measures are used to describe reality, e.g. population indexes to characterize a group of individuals or spatial statistics to characterize an area (Leuven & Poudevigne, 2002). When detailed information about the underlying, disaggregate level is missing or lost in the aggregation process, variability also involves uncertainty, e.g. if it is known that 5% of the individuals in a population exceeds a threshold but not which individuals. In this particular case, variability is perceived as uncertainty by the individual members of the population. It illustrates that the classification of a phenomenon as either variable or (truly) uncertain depends on the perspective of the risk assessor.

There are many techniques to quantify uncertainty, especially true uncertainty and variability. Problem definition uncertainty is generally not quantified. The best option to quantify uncertainty is validation research, i.e. comparison of risk assessment outcome with measurements. However, this is often unfeasible or only partially feasible. Other options include propagation of parameter uncertainty (e.g. by Monte Carlo simulation), model comparison, scenario analysis and expert elicitation.

**Risk-based River Basin Management**

Risk assessment is an important aspect of river basin management as laid down in the European Water Framework Directive (WFD). Examples are the assessment of point source impacts on water systems and the evaluation of the chemical status of a water body. In both cases, the main aim of the assessment is to safeguard a good ecological status. This means that risk assessment is instrumental to protection. The risk estimate does not necessarily have to be realistic, as long as protection is guaranteed. This explains the use of conservative assumptions and models in many regulatory risk assessments. If a conservative assessment produces an unsatisfactory result, a tiered approach may be followed to produce more realistic and less conservative risk estimates. This is a pragmatic and practical way to deal with uncertainty, in line with the precautionary principle.

Environmental quality standards (EQSs) play an important role in the risk assessment process of the WFD. They are compared with predicted or measured concentration levels to obtain an indication of the risk, like in a PEC/PNEC ratio. If the EQS is uncertain, the risk indicator will also be uncertain. This also holds for the PEC. If the uncertainty in PEC and EQS can be quantified, the uncertainty in the risk indicator is also known. The advantage of quantification is that uncertainty can be treated in a consistent manner, i.e. the level of conservativeness of the assessment can be standardized, e.g. by adoption of the 90th percentile as a reference. However, it should be noted that quantification of uncertainty in PEC/EQS ratios does not capture problem definition uncertainty. Even if a conservative EQS of a substance is met, uncertainty about the protection of the aquatic ecosystem remains. One way to reduce this uncertainty is the use of biological indicators, as prescribed by the WFD.

**Thesis 1: Effect assessment is more uncertain than exposure assessment**

Ragas (2000) quantified uncertainty (i.e. true uncertainty and variability) in the derivation and application of EQSs. Uncertainty was quantified by means of uncertainty factors, which were calculated as the ratio between the 95th and 50th percentile of the probability distribution. Separate uncertainty factors were estimated for:

1. EQSs derived to protect humans against genotoxic carcinogens;
2. EQSs derived to protect humans against substances with a threshold effect;
3. EQSs to protect ecosystems;
4. pollutant discharge loads that were derived with water dispersion models to meet EQSs.
Table 1: Uncertainty Factors (UF) for exposure and effect assessment (Ragas, 2000).

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<th>Exposure Assessment</th>
<th>Effect Assessment</th>
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<td></td>
<td>Pollutant Loads</td>
<td>Ecosystems</td>
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<tr>
<td></td>
<td>Geometric Mean UF</td>
<td>Human-Threshold</td>
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<tr>
<td></td>
<td>2.6</td>
<td>6.5</td>
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<tr>
<td></td>
<td>Maximum UF</td>
<td>Human-Carcinogens</td>
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<td></td>
<td>3.6</td>
<td>33</td>
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</table>

The first three factors are indicative for uncertainty in effect assessment, and the fourth for exposure assessment. The results are presented in Table 1. It should be stressed that the models and assumptions used were not fully in line with the new guidelines of the WFD (Lepper, 2005), but comparable procedures were used. So it is safe to assume that the estimates give a good indication of uncertainty in effect and exposure assessment within the framework of the WFD.

Table 1 clearly shows that uncertainty is largest in the derivation of EQSs for human protection against genotoxic carcinogens. This uncertainty is mainly caused by model uncertainty (i.e. the choice of an appropriate dose-response model) and high to low dose extrapolation. However, this uncertainty is of limited relevance for river basin management, since water quality standards are rarely derived to protect humans against genotoxic carcinogens. The uncertainty in the remaining two EQS types is of comparable magnitude, i.e. a geometric mean uncertainty factor of 6.5 for ecosystem protection and 9.8 for human protection against substances with a threshold. Both uncertainty factors are large when compared to the uncertainty factor of discharge loads, which has a geometric mean of 2.6. It indicates that uncertainty in effect assessment is much larger than in exposure assessment. The difference will become even larger when direct water quality measurements instead of dispersion models are used. These results indicate that research to improve risk assessment should focus on effect assessment.

**Thesis 2: Uncertainty can be used to optimise the use of resources**

Ragas et al. (2005) performed a case study in which the costs of reducing uncertainty in EQSs were weighted against the benefits of having less stringent EQSs. The case study dealt with DDT contaminated sediments. The case is not 100% realistic, but appropriate for illustrative purposes.

In the Netherlands, polluted sediments are classified based on EQSs. Moderately and heavily polluted sediment (classes 3 and 4) are and must be disposed off at a cost of € 65.- per m³. For clean and slightly polluted sediment (classes 0-2), the costs are € 7.70 per m³; a difference of € 57.30 per m³. The water board Rivierenland in the Netherlands yearly processes approximately 63,000 m³ of class 3 and 4 sediment, which is mainly polluted with DDT from intensive fruit culture. It is obvious that a considerable amount of money could be saved if more sediment would be classified in class 2 instead of class 3 or 4.

An EQS is used to distinguish between classes 2 and 3. This EQS was derived as the middle value between the Hazardous Concentration for 50% and 5% of the species (HC₅₀ and HC₅). These values were calculated using a species sensitivity distribution (SSD) based on 3 chronic NOEC values; one for each algae, daphnia and fish. SSDs provide a conservative estimate of the HC₅ when data input is low. This means that the HC₅ tends to increase when the number of available NOECs increases. This raises the question what would happen to the EQS of DDT if a fourth NOEC would be added to the current dataset. Ragas et al. (2005) explored this issue by generating hypothetical new NOEC values based on two methods: (1) parametric bootstrap and (2) the analysis of species-specific patterns in a database with NOEC values. The first option showed an 80% probability to obtain a less stringent HC₅ value, and the second option 92%. The expected benefits due to reduced disposal costs in the second option were estimated at € 1.9 million per year. The average costs of an ecotoxicity test are € 40,000.-.

The example shows that investment in extra toxicity test can result in considerable savings. However, this only works if the amount of uncertainty in the assessment is implicitly or explicitly regulated; (e.g. a 90th percentile of a probability distribution). More data will mean smaller confidence intervals and thus less stringent standards. It is interesting to note that this does not always apply to the derivation of EQSs under the WFD. Here, safety factors are generally used to derive EQSs. If more than 3 and less than 10 NOECs are available, a safety factor of 10 is applied to the lowest NOEC. An extra NOEC can never result in a less stringent EQS, only in a more stringent. This is an example of inconsistent regulation of uncertainty and not in line with the general principle that safety margins should decrease when knowledge increases.
Thesis 3: EQSs require spatial and temporal specification

Ragas et al. (1999) applied six different discharge mixing models used to four different real life discharge situations. For each model and discharge situation, maximum allowable annual pollutant loads were calculated according to the 1998 water regulations applicable in Germany, the United Kingdom (UK), the Netherlands, and the United States of America (USA). The results revealed differences in pollutant loads due to model selection and national regulations that for some discharge situations exceeded a factor of 25. Part of this variation was caused by different regulations for dealing with spatial and temporal variability. This raises the question how EQSs should be specified in relation to space and time. The draft EU Directive of on environmental quality standards (CEC, 2006) specifies an annual arithmetic mean concentration (AA-EQS) and maximum acceptable concentration EQS (MAC-EQS). These values refer to two different temporal dimensions: an annual average and a maximum concentration that should not be exceeded any time. The latter standard poses a problem from a statistical perspective: the more samples are taken, the higher the probability that one of the samples exceeds the MAC-EQS. So, again, the water manager is punished for gathering extra information. It would be more appropriate if the MAC-EQS would be specified as a percentile value of a time series, e.g. the 99.5th percentile. Furthermore, current standards lack a spatial dimension. Water authorities are allowed to define a transitional area of exceedance (CEC, 2006), but guidelines for spatial and temporal specification of EQS exceedance are lacking. A large mixing zone may impact the integrity of the entire aquatic ecosystem. It is remarkable how little research has been done in this area, mainly because (eco)toxicological tests concentrate on homogeneous exposure regimes. More research on the impact of spatial and temporal concentration variations is necessary. In the meantime, a provision could be adopted that mixing zones should not impair ecological integrity.

Recommendations & perspectives

- Risks should be estimated with dose-response models instead of PEC/PNEC ratios.
- Research to improve risk assessment should primarily focus on effect assessment.
- Authorities should develop consistent and explicit guidelines for quantification and regulation of uncertainty in risk assessment.
- Additional research is necessary to determine the impact of spatial and temporal concentration variations on the protection level underlying EQSs.

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Ad Ragas studied biology and obtained his PhD with a thesis on uncertainty in environmental quality standards. He is now an assistant professor at the Department of Environmental Science of the Radboud University in Nijmegen, the Netherlands. He chairs the Dutch scientific advisory committee on EQSs and coordinates Research Pillar 4 on Risk Assessment of the EU FP6 NOMIRACLE project (NOvel Methods on Integrated Risk Assessment of Cumulative stressors in Europe). His main areas of interest are risk modelling, uncertainty analysis and risk perception.
Generic exposure models on a river basin scale - A new way to support the risk assessment for contaminants in European river basins

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Background

The 2000 European Water Framework Directive (WFD) calls for water quality management on a basin wide scale. In this perspective, risk assessment for contaminants also needs to be considered on a basin-wide scale. Spatial relations between causes (pollution sources) and effects (ecological risk) are affected by the geometry and the hydrology of river basins: What is the direction of transport of pollutants? How fast is their transport or degradation? What is the dilution or final fate?

Many contaminants such as heavy metals and polycyclic aromatic hydrocarbons (PAHs) are hydrophobic; they sorb to inorganic or organic sediment particles, and thus their fate becomes connected to the sedimentology of a river basin. This gives rise to similar questions as mentioned above related to the basin hydrology. One aspect is added though: sediment, including fine particles holding contaminants, can be deposited and temporarily stored in areas with a low transport capacity. It has been recognised that the (theoretical?) possibility exists that sediment deposit areas accumulate contaminants over longer periods (e.g. decades) and may release them as a result of extreme floods (Westrich & Förstner, 2007).

The problem of mobilisation of historically polluted sediments is under discussion in different river basins. Since the number of polluted sites may be high, the question of prioritisation arises. Furthermore, the concern for this kind of potential pollution problems should somehow be balanced against the control of “normal” pollution discharges, such as effluents from WWTP’s and storm water overflows.

The characterisation of the European water basins in terms of their geometry, hydrology and sedimentology is a necessary precondition for a quantitative analysis which addresses the issue above. This need has been recognised and several EU funded projects have been initiated to provide such information. The present work in the Modelkey project relies on the Catchment Characterisation and Modelling River and Catchment Database, version 2.0 CCM2 (see Fig 1, Vogt et al., 2007).

Fig. 1: Overview of catchments included in the CCM2 database (Vogt et al., 2007).
Even if the WFD addresses chemical pollution, its current form does not yet fully account for the role of sediment in relation to contaminants, nor does it explicitly take into account aspects of bioaccumulation of such contaminants which is significant for their ecological effects. The WFD therefore does not distinguish quality criteria (yet?) for sediments and biota. It is however widely recognised that risk management of contaminants should take into account these two aspects (Salomons & Brils, 2004).

**Thesis 1: Historically polluted sediments are a concern**

In an extensive study for The Port of Rotterdam, a research team compiled an inventory of historical polluted sediments in the Rhine River and its tributaries (Heise et al., 2004). Several hundreds of sites were investigated, and classified in terms of their risk of mobilisations and subsequent impact on the quality of sludge to be dredging from the harbour area. Areas presenting a “high risk with high certainty” were the sediments of the Upper Rhine barrages sections as well as the Ruhr area. As witnessed by a lecture of a Port of Rotterdam official in December 2005, the Port still considers historical polluted sediment as a major threat for its sediment management (Eisma, 2005).

In the aftermath of the Baia Mare incident, where a large Danube tributary suffered major ecological damage due to an unintentional discharge of gold mining wastes, the International Commission for the Protection of the Danube River (ICPDR) orchestrated an inventory of “Old Contaminated Sites” in potentially flooded areas (ICPDR, 2004). This inventory includes landfills, dump sites and storage facilities where harmful substances are deposited. In addition, an inventory was compiled of “Potential Accident Risk Spots”. Fig. 2 provides examples.

![Fig. 2: Example of inventory of “Old Contaminated Sites” (left) and “Potential Accident Risk Spots” (right) in the Danube River Basin (ICPDR, 2004).](image)

A concern, yes! But are water managers and other stakeholders able to quantify the risk and do they have sufficient information to take action?

**Thesis 2: Pollution sources need to be prioritised with respect to the risk of downstream impacts**

Pollution sources, among them the historical polluted sediments discussed above, are too many to be controlled all in one time. A prioritisation between types of sources (WWTP’s, storm water discharges, atmospheric deposition, polluted sediments) or between individual sites/sources is necessary. From the river basin perspective, such a prioritisation needs to take into account the (potential) basin wide impact of the sources.

Water quality management is often focusing on the local water and sediment quality, and whether or not it is in agreement with the applicable standards. Recent research on nutrient management on a basin scale (Kroiss et al., 2005) has revealed that there is sometimes a "conflict of interest" between the local water quality and basin-wide impacts: water bodies with a high dilution capacity usually do not feature severe local water problems, but they are very effective in transporting pollution downstream where it may cause problems in downstream lakes, estuaries and coastal waters. On the
other hand, water bodies with a low dilution capacity often show severe pollution problems, but the problems stay local and the pollution is not causing downstream impacts. Similar paradoxes may exist in the management of hydrophobic contaminants, where also sediments come into the picture. The prioritisation also needs to take into account the continuous versus intermittent character of pollution sources; the impact of potential accident risk spots and polluted areas washed out during extreme floods need to be balanced against the continuous impact of other sources. Finally, the assessment needs to take into account impacts on the sediment quality and the impact on different biota under the influence of the pollutant-specific bio-accumulation.

**Thesis 3: Generic exposure models on a river basin scale provide a tool to do just that**

The Modelkey project used the above Theses as a starting point and concluded that a generic exposure model for European river basins would potentially be a great help for water management. Such a model is under development as a *Rapid Assessment Tool* (RAT):

- working on limited input data from the future user
- covering all relevant aspects of the problem
- providing defaults as much as possible
- allowing the user to refine the assessment by providing more data

The approach is based on the state-of-the-art European-wide catchment geometry and characteristics database CCM2 (see Fig. 3). It relies on hydrology and sedimentology data derived from Modelkey’s BASIN database, as well as on contaminant data derived from Modelkey’s KEYTOX databases. It contains two sub models deriving the transport coefficients related to the movement of water and sediment through the catchment, a sub model computing the transport and fate of contaminants and a sub model for exposure and bioaccumulation in food webs. Another separate sub model is dedicated to estimating the risk of remobilisation of polluted sediments under variable hydrological conditions.

The generic exposure model takes into account probabilistic aspects, separated in one part related to the basin hydrology and another part related to other factors. In 2007 the first trial applications are carried out, for Modelkey’s Case Study Areas (Elbe, Scheldt, Llobregat). Some highlights will be presented during the conference. A major challenge will be to validate the selected model approach, using data collected for the Case Study Areas mentioned above.

The output from the generic exposure model is tailored to contribute to ecological effect assessment models and decision making procedures.
Recommendations & perspectives

The Modelkey related research is in the process of providing a generic exposure model, which aims to be a tool for quantitative and objective ranking of contaminants and/or pollution sources, on a basin wide scale. The tool will be available for all European basins, and will be available to all European water managers and their technical advisors.

The value of this tool would be substantially increased if it could be linked to a quantitative assessment focusing on the sources of contaminants (e.g. tools developed in the SOCOPSE project), and maybe also to atmospheric transport models.

References


I am particularly interested in the analysis of the behaviour of complex water systems; to understand why they are as they are, and to indicate what would happen if changes would be implemented.

In my 21 year career at Delft Hydraulics, I have had the opportunity to carry out this kind of work for river systems, estuaries and coastal waters in Europe and elsewhere. Presently, I am working in projects for the North Sea, the European river basins (Modelkey), the Pearl River basin (P.R. China) and the Arabian Gulf.

Jos van Gils, WL | Delft Hydraulics
Background

Organic and inorganic pollutant turnover, storage and transport in soils, sediments, ground- and surface water remains to a large extent poorly understood. This lack of understanding results in part from a limited knowledge about large-scale behaviour (catchment or regional) of pollutants and is further complicated by the complexity and heterogeneity of the systems involved. Little is also known about links between compartments such as the atmosphere, soils ground- and surface water as well as sediments. It is precisely at these interfaces where we expect the steepest biogeochemical gradients.

The understanding and quantification of material and pollutant fluxes within and between these compartments is one of the challenges that the EU 6th Framework Integrated Project “AquaTerra” addresses. The main goal of this project is to provide the foundations for an improved understanding of the behaviour of environmental pollutants.

AquaTerra integrates across various disciplines that range from biogeochemistry, environmental engineering, computer modelling and chemistry to socio-economic sciences. Activities involve researchers, but also practitioners and end-users such as policy-makers, river basin managers as well as regional and urban land planners. Field study areas are in the river basins of the Ebro, the Meuse, the Elbe and the Danube as well as the small French agricultural catchment of the Brévilles Spring. Within the first three years of the project, integrated work among the various partners has led to the collection of about 2000 soil, sediment, water and biological samples. Apart from increasing availability of specialist results, project activities also included various meetings, the publishing of internal reports and national as well as peer-reviewed manuscripts and a special issue (Barth and Grathwohl, 2005; Barth and Grathwohl, 2006; Barth et al., 2007a; Barth et al., 2007b; Brouyère, 2006; Bürger et al., 2007; Dusek et al., 2006; Eljarrat et al., 2005; Fowler et al., accepted; Gerzabek et al., 2007; Gocht et al., 2007a; Gocht et al., 2007b; Hildebrandt et al., 2007; Kalbus et al., 2006; Kolditz and Bauer, 2004; Kolditz et al., 2006; Lair et al., 2006; Marani and Zanetti, 2007; Orban et al., 2005; Roulier et al., 2006; Terrado et al., 2007; Vijver et al., 2007; Zanotti et al., 2004)). Further information about the project is available on the AquaTerra Website (http://www.eu-aquaterra.de/).

Several preliminary theses arise from the work conducted in AquaTerra and only selected ones can be presented here.

Thesis 1: Pesticides are often persistent for years to decades

The AquaTerra project hosts one of the most extensively studied agricultural areas for pesticide pollution in Europe, the Brévilles Catchment. This area has been extensively studied since 1999 already by the Pegase Project and was subject to further investigations in AquaTerra since 2004. As an important source for local water supply, the spring was disconnected from the drinking water distribution network in August 2001 because pesticide and nitrate concentrations exceeded water quality limits. The Brévilles catchment belongs to a wider aquifer system of about 12 km², that constitutes a closed system. The area has been investigated by questioning farmers about pesticide and fertilizer use, the installation of a total of 18 piezometers, tracer tests, microbiological investigations and numerical models about water and material transport. Initial results show that pesticides such as Atrazine with an application ban in 1999 (4 years before the official ban in France) can be found even years after their application in the spring (Fig. 1), thus suggesting a stability and slow transport of this
molecule and its related compounds in the subsurface. Scaling up such a pollution scenario to the catchment and basin scale would likely cause even longer time periods for pesticide turnover in subsurface environments.

Fig. 1: Time series of atrazine, deethylatrazine and deisopropylatrazine concentrations in the Brévilles spring (June 2004- April 2006) after Barth et al. (2007b).

Initial calibration results of the modelling of the Brévilles Case seems promising to establish a tool to predict pesticide and other pollutant transport in well instrumented catchments also in other parts of Europe. From these results it becomes clear that response time of the soil-water system (i.e. trend reversal in groundwater and surface water concentrations) to measures taken (stop input of pesticides) is much larger than a few years, probably a decade or longer. This time-frame is in many instances longer than most policy instruments (WFD, GWD) usually take into account.

Thesis 2: Temperature can act as a controlling factor for biodegradation

AquaTerra hosted several experiments that could show transformations of chlorinated benzenes and ethenes by sediments and soils from environments including rivers, sediments and soils. Screening environmental samples for reductive dechlorination capability showed that reduction of these compounds into less chlorinated derivatives is possible for both chlorinated benzenes and ethenes. Moreover, microorganisms known to be involved in anaerobic reductive dechlorination were previously detected in basins of the AquaTerra project. Reconnaissance experiments of this kind tested the effect of environmental parameters on pollutant degradation with soil and sediment material of the Ebro-Flix area that is well known for accumulation of several chlorinated contaminants. Initially, experiments aimed to mimic on-site conditions and compared the activity of the dechlorinating consortia in the laboratory environment. Furthermore, follow-up experiments investigated the effects of temperature and co-occurrence of chlorinated compounds in relation to the activity of dechlorinating species. First results indicted a high capacity for natural attenuation at the site with ambient temperature as one of the major controlling factors. While high temperatures of 30 degrees and above may reflect rare conditions in central European sites, temperature differences of 10 degrees and more are realistic dynamics. Therefore the results indicate that degradation may be more active during the warm seasons. These specific results show that the relation between temperature and biodegradation activity is in line with the local climate conditions (air temperatures ranging between 10-40 degrees). Microbial populations originating from regions with colder or warmer climates are likely to show other temperature-activity functions. Change of climate is therefore also likely to enforce adaptation of the microbial community to the new conditions, which may result in changes in pollutant biodegradation.
Thesis 3: Degradation of pollutants is often found in the laboratory, but real environments often show different dynamics

Often the fate and transport of organic as well as inorganic pollutants exposes inconsistencies between laboratory and field observations. For instance, it is known that data from persistent organic pollutants obtained at the field-scale indicate a high stability when based on mass balances at the catchment scale. This indicates continuous accumulation of the compounds in top soils. In contrast to this, many laboratory experiments about degradation indicate the potential to metabolize the same pollutants rapidly. Even if sorption is strong, desorption is relatively rapid compared to the time scale of the field (i.e. decades to centuries). Currently, it is still unclear which processes dominate the long-term fate of pollutants in soils.

Even though we can show degradation, transformation and immobilisation of compounds in the laboratory, their behaviour in the field under natural conditions needs to be further examined. We are currently able to determine the status quo of environmental systems with ever increasing accuracy and detail of analytical techniques. This allows good estimation of distribution patterns and to evaluate sources and fluxes of pollutants. On the other hand, when process-relevant studies are based in the laboratory, they yield crucial first information on environmental behaviour of pollutants. Nonetheless, these laboratory results need to be either adapted to the field (which remains a major challenge) or evaluated with great care when transferring the results to field sites. Some first field-based techniques such as stable isotope quantifications of turnover of organic compounds in the field are beginning to show the extent to which for instance natural attenuation actually takes place in real natural environments (Morasch et al., 2007). Yet, our understanding of such processes is still limited by too broad sample densities in space and time. These often yield highly localised information and snapshots of long-term processes and integral monitoring techniques might offer future solutions and trends in this respect.

Thesis 4: Floodplains are dynamic environments with mobilisation and storage of materials and pollutants

Floodplains are among the most dynamic biogeochemical environments of the Earth’s surface. They link terrestrial with aquatic systems and often act as important sinks and/or sources for inorganic as well as organic contaminants. Periodic flooding can cause strong fluctuations in soil redox conditions and leads to complex abiotic and biotic transformations of pollutants, organic matter, and minerals. These processes ultimately control the behaviour of contaminants in such systems. Current and new research in floodplains in selected AquaTerra Elbe, Danube, Meuse and Ebro sites are good examples on how to improve understanding of processes and quantification of fluxes between compartments. This research shows that it is equally important to advance knowledge about inorganic and organic pollutant turnover including heavy metals as well as persistent organic pollutants such as polyaromatic hydrocarbons (PAHs).

Recommendations & perspectives

Results from the detailed pesticide study in the Brévilles area have shown that application of diffuse pollutants even in small areas leads to a highly complex response in receiving ground- and surface water systems. Turnover, storage and degradation times appear much longer than expected and may affect even small systems such as the Brévilles Catchment for years to decades. When scaling up to larger catchments, even longer transport times can be anticipated because of larger distances of groundwater transport. From these results it becomes clear that response time of the soil-water system (i.e. trend reversal in groundwater and surface water concentrations, (Visser et al., 2007) to measures taken (stop input of pollutants, nutrients, of diffuse contaminants from atmospheric deposition) is much longer than a few years and often lasts up to decades or longer. This time-frame is often much longer than policy instruments (WFD, GWD) usually take into account and they must therefore adapt these time frames in order to be applied effectively. Such long time frames also have advantages. For instance, microbiological turnover of pollutants in the subsurface may have more time to remove pollutants before they are transported to vulnerable receptors like drinking water aquifers or organisms in ecosystems. However, further research is necessary to determine which metabolites would be expected under such scenarios and whether they are stable and/or harmful. Overall, results of the
Brévilles Catchment lead to recommendations that crop controls and fertilisers need to be applied with care and under consideration of the consequences for receiving water systems that often serve as drinking water supply.

First results of AquaTerra hotspot and diffuse pollution patterns show that the evaluation of large-scale and long-term pollutant behaviour needs to be further resolved under realistic field conditions with temperature changes and mixtures of pollutants, changing geochemical and microbiological conditions. More field studies under real conditions are necessary to feed results into reactive transport models. For the above, networks of passive samplers and modern monitoring techniques using stable isotope fractionation patterns may hold the key for taking laboratory results to the field (Morasch et al., 2007). Floodplains are not only interesting sites for dynamic biogeochemical research but also control pollutant storage and release with respect to soil groundwater and river interactions. Further research and exchange of knowledge needs to focus on these areas in order to evaluate how pollutants are turned over and under which water level and associated redox, pH and temperature conditions they may be mobilised or stored. AquaTerra will dedicate a special workshop about “Contaminant dynamics in periodically flooded soils” on the upcoming EUROSOIL Congress, Vienna, August 25-29, 2008. Our recommendation in this context is to have an active participation by AquaTerra and other members of the soil-sediment-ground and surface water research community at this event.

Acknowledgements

'This work was supported by the European Integrated Project AquaTerra (GOCE 505428). The project has received research funding from the Community's Sixth Framework Programme.'

References


Johannes A.C. Barth: Research interests include biogeochemical consideration of natural and anthropogenic systems with dynamics of pollutants and turnover of natural material including weathering, transport and turnover of oxygen, carbon, nitrogen. These Investigations are often combined with stable isotopes to provide extra labels about origin and turnover of materials.
“Chemical activity” and “accessibility” as key parameters for risks due to sediment-associated organic contaminants

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Background

Two sides of bioavailability

The last decades of research has led to several bioavailability concepts and to many more methods to measure bioavailability. One reason for this diversity is that they aim at two fundamentally different parameters [i] accessible quantity and [ii] chemical activity (Reichenberg and Mayer, 2006) (Figure 1).

[i] The accessible portion describes a mass of contaminant, which can become available to e.g. biodegradation and biouptake. The accessible portion can be determined with mild extraction schemes or depletive sampling techniques. [ii] The chemical activity quantifies the potential for spontaneous physico-chemical processes such as diffusion, sorption and partitioning. For instance, the chemical activity of a contaminant in a sediment determines its equilibrium partitioning concentration in sediment organisms and gradients in activity determine the direction and extent of molecular diffusion between sediment and water column. Chemical activity can be measured with equilibrium sampling devices and is theoretically closely linked to both fugacity and freely dissolved concentration.

Fig. 1: Bioavailability parameterization: Conceptually, a chemical contaminant in sediment is distributed between the pools; irreversibly bound, reversibly bound and freely dissolved. The reversibly bound and the freely dissolved are accessible. Equilibrium sampling devices can measure the chemical activity of the freely dissolved and reversibly bound contaminant. (From (Reichenberg and Mayer, 2006)).

Thesis 1: Chemical activity and freely dissolved concentrations can be measured by equilibrium sampling into thin polymer coatings

A thin polymer phase can be brought into direct contact with the sediment matrix in order to establish equilibrium between sediment and polymer. At equilibrium, the chemical activity of the contaminant is the same in the polymer as in the sediment. Its concentration in the polymer can then be measured and is proportional to the freely dissolved concentration, chemical activity and fugacity in the sediment (Mayer et al., 2000, Mayer et al., 2003, Golding et al., 2007). A number of different equilibrium sampling formats developed in recent years will be presented.
Thesis 2: The chemical activity drives the bioconcentration in sediment organisms

The chemical activity of sediment contaminants drives their bioconcentration into sediment organisms. This is the basis for the frequently applied equilibrium partitioning theory (Ditoro et al., 1991), which can be used to predict contaminant concentrations in for instance sediment dwelling worms. Research during the last decade has shown that such predictions can be inaccurate, and much of the error can be attributed to the variability of the sorptive properties of both sediment matrices and biological tissues (i.e. BSAF). Uncertainties in partition ratios can be circumvented when applying measured freely dissolved concentrations, chemical activities or fugacities in the prediction of bioconcentration (Kraaij et al., 2003, You et al., 2006, Golding et al., 2007).

Fig. 2: Measured versus predicted steady-state concentrations (C\text{B}) in Tubificidae worms. The estimated steady-state concentrations are calculated based on measured freely dissolved concentrations and bioconcentration factors. The solid line represents the perfect fit line for 16 hydrophobic organic contaminants with 4.6<log K\text{OW}<7.5. (From (Kraaij et al., 2003)).

In sediment, most hydrophobic organic contaminants are bound to the sediment matrix, and binding then controls freely dissolved porewater concentrations and chemical activities. This is crucial for the exposure in sediment suspensions and sediment dilutions, where desorption will buffer freely dissolved concentrations (ter Laak et al., 2007). This has implications for sediment toxicity tests.

Thesis 3: Baseline toxicity is exerted at chemical activities of 0.01 – 0.1

Effect concentrations for aquatic baseline toxicity generally decrease with increasing Log K\text{OW} values of up to about 5 or 6, whereas less is known about the baseline toxicity of organic chemicals with Log K\text{OW} values above 6. A physico-chemical analysis of the dissolution process for organic chemicals was combined with reported baseline toxicity data (Reichenberg and Mayer, 2006), which lead to the following conclusions. First, there is no absolute hydrophobicity cut-off for baseline toxicity at a Log K\text{OW} of 6, since aquatic baseline toxicity for fish and algae was observed for chemicals with Log K\text{OW} values above 6.5 and with effect concentrations below 10 μg/L\text{aq}. Second, the baseline toxicity of hydrophobic organic substances was exerted at a relatively constant chemical activity of 0.01 to 0.1. Finally, organic chemicals with high melting temperatures cannot provide a sufficient chemical activity to exert baseline toxicity as individual and pure agents. However, such substances are still expected to contribute to baseline toxicity when part of a complex mixture.

Recommendations & perspectives

The implementation of “bioavailability” into risk assessment and risk management requires well defined and measurable parameters. Chemical activity, freely dissolved concentration and fugacity are suited to predict equilibrium partitioning into biota and to determine the direction of diffusive mass transfer between sediment and water column. They can be measured with different techniques that are based on the equilibrium sampling into a thin polymer. In contrast, accessibility describes a pool of contaminant that can be released from the sediment and it can be measured by depletive sampling or mild extraction schemes.
References


Dr. Philipp Mayer’s research interests include (1) analytical techniques directed at the exposure of organic contaminants in various environmental matrices, (2) minimally destructive techniques that are suited for the screening of new (unknown) substances and (3) the exposure and effects of highly hydrophobic organic chemicals. He leads the work package on “available exposure” within the EU project NOMIRACLE and participates in the EU projects ALARM, OSIRIS and BIOTOOL.
Effects-directed identification of key toxicants

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Background

The monitoring and regulation of prioritised substances plays an important role in focusing the efforts of regulators and monitoring authorities in protecting the aquatic environment. However, are we as a scientific community confident that priority substances alone pose a risk to European river basins? The vast majority of chemicals used by society are not monitored in the environment and evidence to date suggests substances other to prioritised PBT (persistent, bioaccumulative and toxic) substances present in the environment are capable of exerting a range of different effects including those exerted by current priority substances. Identifying which substances are responsible for these effects is one of the greatest challenges that face environmental scientists (Muir & Howard, 2006). A number of approaches are available for identifying key toxicants which co-exist with other pollutants as complex environmental mixtures along with natural compounds, some of which are toxic and some of which are benign. Identifying which substances have the potential to cause harm is therefore a major and important challenge. Effects-directed chemical analysis (EDA or TIE) is one (of many) approach that can be used to identify key toxicants in environmental samples (See Brack, 2003 for review of approach and Brack et al., 2007 for review of application to European river basins). Through a series of theses this presentation will provide evidence to support the hypothesis that not only priority substances present a hazard to European river basins and that EDA is one suitable technique for their identification.

Thesis 1: Are there ‘unknown’ substances to be found in the aquatic environment?

Undoubtedly! The scientific literature has seen an explosion of publications on the occurrence of ‘emerging substances’ over the past decade; for example pharmaceuticals and personal care products. These compounds have been present in the environment since their introduction to society and it is indeed our awareness of their occurrence in the environment that is new. This increasing awareness has been made possible through the development of modern analytical techniques and a better focus on the substances that contaminate our environment (Muir & Howard, 2006; Ternes 1998). The number of chemical substances that exist has been estimated at over 8 million, many of which can potentially be released into the environment. The European Inventory of Existing Commercial Chemical Substances (EINECS) lists more than 100,000 substances that are currently in use today. It is therefore unsurprising that with major advances in analytical science, we are becoming increasingly aware of which chemicals contaminate the aquatic environment. In addition an improved appreciation that it is not only anthropogenic contaminants that occur in European waters but also for many classes of compound(s) there are natural sources. For example naturally occurring brominated dibenzo-p-dioxins have been shown to occur in Scandinavia (Haglund et al., 2007). One must remember that occurrence alone does not equate to hazard whilst the key toxicants in an ecosystem may not necessarily be new emerging compounds.

Thesis 2: Unknown causes of toxic effect

In Thesis 1 we have demonstrated that compounds currently unknown as environmental contaminants exist in the environment and as such may potentially exert known effects. In Thesis 2 we provide examples of observed effects where the cause is/was unknown.

Past examples of unknown lethal effects

The toxicity of surface- and pore water and sediments can not always be explained by the combined toxicity of those substances routinely monitored. An early example of this, which initiated effects-directed analysis at Cefas in the UK, was the observation that the levels of compounds detected during the UK National Marine Monitoring Programme were insufficient to explain the effects seen in the oyster (Crassostrea gigas) embryo bioassay (Thomas et al., 1999a; 1999b). Another example of where
unknown lethal effects were seen was in surface waters from headwater streams in the South East of England following precipitation events (Thomas et al., 2001). It was hypothesised here that the transient movement of pesticides was the cause; however EDA showed that this hypothesis was partly correct.

**Past examples of unknown estrogenic effects**

In the 1990s effects consistent with exposure to estrogenic chemicals (i.e. vitellogenin induction) were first reported in the UK (Purdom et al., 1994). These effects were reported in rivers and estuaries receiving treated wastewater. Through the application of in vitro receptors assays such as the yeast estrogen screen (YES) it was also possible to quantitatively measure the in vitro estrogen receptor (ER) agonist potency of wastewater treatment works (WTW) effluents, even though the identity of those ER agonist was at the time unknown. Similarly, the YES assay was used to measure the ER agonist potency in a number of different compartments and show that certain UK estuarine sediments in fact contain high concentrations of ER agonists. Application of the yeast androgen screen (YAS) assay showed that androgen receptor agonists were also present in these effluents.

**Present status**

The EU FP6 project MODELKEY (Brack et al., 2005) has used a battery of small scale bioassays to characterise the effects in samples collected from three European river basins; Elbe, Schelde and Llobregat (Table 1; Lamoree et al., 2007). These and other toxicity characterisation data demonstrate that compounds capable of exerting a multitude of different ecotoxicological effects occur in European river basins and that there is a major task in identifying the compounds responsible for these effects.

**Thesis 3: Identifying key toxicants by EDA**

Since we know both unknown compounds and unknown causes of toxicity exist it seems plausible to claim that the former must be responsible for the latter. However it is just as likely for certain mechanisms of toxicity (e.g. AR antagonists) that the effects we are observing may be due to the presence of known contaminants. Likewise specific effects, such as antibiotic activity, may be due to emerging compounds such as antibiotics/antibacterials and or active metabolites. Due to the complexity of environmental mixtures it is important to keep an open mind and consider all of the information available when identifying key toxicants.

**Table 1: Summary of the toxic effects measured in European river basins within MODELKEY.**

<table>
<thead>
<tr>
<th>Toxic mechanism</th>
<th>Bioassays</th>
<th>Effect observed</th>
</tr>
</thead>
<tbody>
<tr>
<td>Geno-toxicity</td>
<td>Green Screen</td>
<td>AMES II</td>
</tr>
<tr>
<td>Hormonal disruption ER</td>
<td>(anti)YES  UmuC</td>
<td>ER-Calux</td>
</tr>
<tr>
<td>Hormonal disruption AR</td>
<td>(anti)YAS</td>
<td>(anti)AR-Calux</td>
</tr>
<tr>
<td>Ah-based toxicity</td>
<td>ToxAlert</td>
<td>DR-Calux</td>
</tr>
<tr>
<td>Cell toxicity</td>
<td>T4-TTR</td>
<td>Microtox</td>
</tr>
<tr>
<td>Thyroid hormone</td>
<td>ABC</td>
<td></td>
</tr>
<tr>
<td>Antibiotic</td>
<td>assay</td>
<td></td>
</tr>
</tbody>
</table>

In the examples presented in Thesis 2, effects-directed analysis showed that alkyl and chlorinated phenols as well as other non-regulated compounds were responsible for some of the observed toxicity in UK estuaries (Thomas et al., 1999a; 1999b). The transient movement of nonylphenol used as a carrier in pesticides along with endosulfan sulphate and pendimethalin were shown to be responsible for the effects observed in headwater streams and not those pesticides targeted (Thomas et al., 2001). As we all know now steroid estrogens are the main ER agonists in WTW occasionally accompanied
by xeno-estrogens such as nonylphenol and bis-phenolA (Desbrow et al., 1998; Thomas et al., 2001). Most of the ER agonists in marine sediments remain unknown (Thomas et al., 2004), whilst a mixture of testosterone metabolites are the primary AR agonists in WTW effluents but we are as yet uncertain as to the cause of AR antagonist activity in both industrial and domestic waste streams (Thomas et al., 2002). These examples show how known compounds can be identified as the cause of effects, however when confronted by compounds not present in mass spectral libraries or not amenable to gas chromatography the task becomes more difficult. Many EDA studies are not successful in identifying the key toxicants but are successful in eliminating the possibility that it may be a known substance. In our opinion, this is where the current challenge lies and where current advancements in instrumental and ecotoxicological techniques are providing exciting opportunities to advance our knowledge about what contaminates our environment and what the key toxicants in European river basins.

Recommendations & perspectives

- Regulatory authorities must accept that we do not yet have a complete understanding of the contaminants that contaminate European river basins.
- Advancements in modern analytical and ecotoxicological techniques we are better placed than ever to identify what the key toxicants are
- EDA is only one of many approaches which need to be used in a complementary manner in order to optimise our ability as environmental scientists to identify the key toxicants present in European river basins.
- EDA using bioassays specific to a particular mode of action are often successful in identifying those compounds responsible. However, EDA is not the ‘magic bullet’ and is an investigative research tool and not suitable for routine application.
- Where EDA (and other approaches) are not successful in identifying the causes of toxic effect(s) alternative risk assessment strategies need to be sought.

References


Dr. Kevin Thomas is Research Manager for Ecotoxicology and Risk Assessment at NIVA. His research interests are broad however placed upon establishing which contaminants in the environment are actually hazardous and pose a risk to the environment.
Novel analytical tools - Progress on the identification and quantification of hazardous contaminants in complexly contaminated environments

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Background
Chemicals are important in modern society, from an economic, lifestyle and health point of view. In the European Union more than 100,000 chemicals are used, many of these will enter the environment, but only a limited number of chemicals are routinely monitored. To determine the risk of the complex mixture of chemicals in the environment is a challenging task. Information on exposure (identity of compounds, concentrations) and effects (mode of action, effect levels) are needed to determine the risk. Various state-of-the art analytical techniques (e.g. GCxGC-ToF-MS, LC-QTOF-MS) are evaluated in the MODELKEY project for the identification of unknown compounds in environmental samples. Examples of sediment from European basins will be highlighted to show the complexity of the samples, but also the power of the current analytical techniques. Toxicity profiling as additional tool to support the identification of compound groups will be shown. Questions that will be discussed are: What are the gaps and restrictions of the techniques, do we need more sensitive or selective analytical tools, and do we have enough identification power to identify “the” key-toxicants in the environment.

Thesis 1: Sensitive and selective analytical tools are needed
Three decades ago (1975) analytical techniques were able to detect PCBs at the 0.1 mg/kg level in sediment using gas chromatography (GC) with selective detection. The introduction of high-resolution capillary gas chromatogram further improved the limit of detection (LOD) 100x to 1µg/kg, and in the ’90 more sensitive detectors were able to detect 1 ng/kg. Currently 1 pg/kg can be detected using selective and sensitive detectors, which is 100,000,000 times better than 30-years ago. The increase in sensitivity throughout the last decades is also observed for liquid chromatography (LC) combined with selective detectors (MS). But do we need such sensitive detectors? Definitely!

Sensitivity: Natural and synthetic estrogens
Natural and synthetic estrogens can currently be detected at the low ng/l level in surface and waste water. These low detection levels are evidently needed as negative effect for ethynylestradiol in fish occur at the low ng/l range (Nash et al., 2004; Kidd et al., 2007). A recent 7-year lake studies in Ontario (Canada) with 17α-ethynylestradiol showed that chronic exposure of fish (fathead minnow) to low concentrations (5–6 ng/l) resulted in the impact on gonad development, intersex by male fish and finally a near extinction of the species in the lake (Kidd et al., 2007).

Separation power: PAHs
In three decades the separation power moved from low resolution GC (packed columns) to high resolution GC (capillary columns), finally to comprehensive to-dimensional gas chromatography (GCxGC) which has a very high separation power. GCxGC was first described 12-years ago (e.g. Philips et al., 1993, Philips and Beens, 1999) and the number of publications rapidly increased since then. The technique not only provides an enhanced sensitivity compared to conventional GC, but also provide structured chromatograms (e.g. Korytar et al., 2005) that could help to identify unknown compounds. The introduction of fast-scan time-of-flight mass spectrometers (ToF-MS) further improved the separation power but also the identification power of unknown compounds. Separation of toxic and non-toxic PAH isomers that occur in the environment can be obtained with GCxGC.
Thesis 2: Do we have enough identification power to identify “the” key-toxicants in the environment?

The last decade identification of the structure of unknown hazardous compounds in the environment has become an important topic issue beside the quantification of contaminants. New emerging compounds, e.g. PFCs, BFRs, siloxanes, have been identified (e.g. Giesy and Kannan, 2001; Muir and Howard, 2006; Schweigkofler and Niessner, 1999). Novel analytical tools, such as time-of-flight mass spectrometry combined with GC or GCxGC that are able to detect and provide structure information of compounds at low levels in the environment, are very powerful and certainly enhanced our identification capabilities more than ever. However, when compounds are not present in mass spectrometry libraries other techniques such as accurate mass spectrometry (Thurman et al. 2003) or atomic emission detection (Van Stee et al., 2002, 2003) could be used to provide information on the elemental composition of compounds. Still, most of the current identification work is based on GC-able compounds. If compounds can not be analysed with GC, identification becomes more problematic. Novel and expensive techniques as LC-QToF and LC-Orbitrap, using accurate mass spectrometry, have the potential to fill this gap.

Recommendations & perspectives

- Novel analytical tools are certainly essential and needed to fill the gap to identify and detect “the” key-toxicants in the environment.
- Identification of unknown compounds is not only a matter of novel analytical tools but also an issue of additional tools such as detailed information on the location, toxicity profiling, and Internet.
- The biggest challenge is the identification of compounds that are not GC-able.
- Combining chemical with biological tools should be future explored to develop techniques that are able to extract/separate, detect and identify active compounds from no-active compounds.

References


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Assessment concepts for effects of multiple contaminants with time-variable exposure

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Background

Organisms in the environment are typically exposed to various stressors, while risk management traditionally focuses on the assessment of individual priority compounds. Organisms that are of concern for the services they provide or as sentinel species may be exposed to multiple stressors chemical and non-chemical either simultaneously or in time sequence. Due to the large number of possible mixtures and their variability in time experimental investigation of every conceivable mixture for their adverse effects is not a viable option. Instead during the last decade modelling approaches have been discussed in ecotoxicology that allow the prediction of expected combination effects based on the knowledge of the biological activity of the individual components.

Thesis 1: Scientific concepts for risk assessment of simultaneous exposure of organisms towards mixtures of chemicals are reasonably validated and ready for use

As a canonical approach towards mixture assessment testing of environmental samples of concern has become established in cases where the mixture composition for some reason can be assumed to stay somewhat stable. This approach has become established in Germany e.g. in effluent testing.

With the Saariselkä agreement in 1992 two prediction concepts, namely concentration addition and independent action (for formula, assumptions e.g. Altenburger et al. 2003), have become established as reference models that allow calculation of expected combined effects on the basis of concentration effect information for the individual components of the mixture of concern. Both concepts are regarded as biologically plausible, though there is ongoing debate as to how pharmacodynamic information on the mode of action of compounds can be used productively to opt for a specific use. The predictions obtained account only for non-interactive joint action.

Through studies of binary and multiple mixtures of various chemicals with specific and unspecific modes of action using various biological systems and endpoints extensive knowledge as to the predictive capabilities of both concepts is available. Briefly it may be summarised as follows: (i) Concentration addition can be considered to provide reasonable worst case estimations in predictive chemical assessment, (ii) in site specific scenarios, e.g. in the confirmation step of bioassay directed toxicant identification typically independent action provides the more conservative estimates and help to avoids overlooking unresolved toxicity.

Even specific concerns, as e.g. the suitability of the reference models for predicting low dose effects (see figure 1), or the usefulness for community level effects have been experimentally studies. Moreover, several papers deal with proposals of how to translate the concepts into regulatory practise for setting of water quality criteria, for environmental sample assessment at the catchment scale, use in probabilistic assessment and others. Input requirements are well known and for concentration addition there is no need for other than the typically available ECx-type of information for the chemicals of concern.
Thesis 2: Approaches for describing combined effects from joint exposure to chemical and non-chemical stressors are under development

In recent years non-chemical stressors became of concern too and several studies have tried to adapt the above outlined approaches for the description of binary combination of a chemical and a non-chemical stressors. In particular the formulation of adequate metrics and provision of scales that allow systematic consideration have been major challenges. So far, most studies use response surface type of assessment tools (e.g. Jonker et al. 2005) and restrict their scope to data description (Heugens review) which therefore is not as yet ready for any risk assessment practise that has to rely on inference statements.

Exemptions may be found when it comes to physico-chemical interaction e.g. the photoenhanced toxicity of PAH-type compounds for which figure 2 provides an example.

Fig. 2: Regression through the complete data set when relative efficacies for photoinduced toxicity (RPE) are assumed according to Grote et al. (2005). anthracene (x), benzo[a]pyrene (●), benzo[a]anthracene (▲), fluoranthene (○), pyrene (♦), benzo[b]fluoranthene (■), benzo[k]fluoranthene (Δ), indeno[1,2,3-cd]pyrene (□), benzo[ghi]fluoranthene (◇)

\[ y = -0.5283x - 1.7917 \]
Thesis 3: Sequential exposure may also be relevant for organisms and risk assessment

Sequential exposure regularly might occur for organisms in streams and ponds adjacent to agricultural areas where spray calendars are common practice. Typically, exposure pulses for aquatic organisms follow rainfall events (Reinert et al. 2002). Also, water bodies that receive waste water typically carry contaminant loads from upstream emitting sources. The latter became evident again in the current debate on pharmaceuticals where it was shown that despite the biodegradability of individual chemical components, their widespread occurrence in freshwater bodies suggest a 'pseudo'-persistence due to input at different sites along a stream. Also, organism mobility in their environment may lead to sequential exposure against various contaminants from different local sources.

Exposure patterns on the time scale need characterisation in terms of pulse length, pulse intensity and pulse frequency which might not be accessible through routine monitoring programme. Even modern analytical approaches using passive sampling techniques might not provide the information necessary to consider time dependent effects in a systematic manner, depending on the approach chosen to model time dependent biological effects.

Thesis 4: Assessment of combination effects from sequential exposures is a challenging research field

When it comes to assessing combined effects from sequential there are three major issues to consider: (i) sufficient knowledge of the time variable exposure regime; (ii) adequate description of the concentration-time response relationship and (iii) a suitable link function between the time response and the mixture response functions.

Modelling of time-concentration-response relationships in order to obtain estimated effects/effect concentrations for individual components is the major challenge when aiming at combined effect predictions for sequential exposure. Most current modelling effort of time-response relationships base on some generalization of Haber’s rule (c x t = constant), typically in the form

\[ E = c \times t^k \]

With E=effect, c=concentration of a chemical, t=time of exposure, and k=compound specific coefficient (Rozman and Doull 2001). Thus the time-dependent biological effect is regarded as the product of time and concentration but with a compound specific exponent that prohibits extrapolations between compounds and organisms. To still derive generalisations various attempts have been made to identify patterns of response or extreme cases by devising models of time-response relationships focussing on the description of receptor chemical interaction (e.g. Legierse et al. 1999) or using equilibrium portioning theory for non-specifically acting chemicals (e.g. Mackay et al. 1992).

Currently, proposals for unifying both approaches (Lee et al. 2002) are under discussion. These approaches not only allow quantification of time-dependent responses by area under the curve, body burdens or similar calculations but also have greatly enhanced our understanding of the relevant processes in the pharmacokinetic and pharmacodynamic phase. For effect estimation in the context of ecological risk assessment, however, the major drawback is, that these models require not only independent estimates for uptake and elimination kinetics but also mode-of-action information that may not be available for many compounds ecological risk assessment has to deal with. Thus there is a need to at least additionally use and develop assumption-free descriptive models, for which an example is provided in figure 3.
Fig. 3: Predicted and observed combined effect of a binary mixture of atrazine and n-phenylnaphthylamine with the components present in timely sequence (application at 0 and 6 hours of the cell cycle) in an algal growth assay, based on a linked concentration-time- and mixture response model. Circles and crosses depict the expected effects for the individual compounds while lines represent the model prediction for the combined effect from sequential exposure and diamonds show the observations.

Recommendations & perspectives
For risk assessment, the following recommendations may be derived:

1. Extrapolation of expected combined effects based on dose response information of components of concern is possible using the reference models of concentration addition or independent action.

2. For time varying exposure and effects various mechanism-based models are available in the literature but consensus as to which to use in which cases is lacking and data requirement will not easily be met. Alternatively, scenarios for reasonable worst case estimations regarding time weighed average concentration and empirical extrapolations models might be used.

3. For the combined effects from exposure to chemical with non-chemical stressors non general recommendation can be extracted as yet.

References


Current research interests comprise mode-of-action analysis, mixture toxicity assessment, bioassay development for toxicant identification, and PKPD models.

Main projects deal with extrapolation techniques in risk assessment of chemicals, use of ecotoxicological tools for product development and site specific assessment of contamination impacts.
Higher Tier toxicity studies as the link between laboratory tests and ecosystem health assessment

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Background

‘Higher tier studies’ might not be a familiar concept in River Basin Risk Assessment. It is however a common concept in the risk assessment of agricultural chemicals, where a tiered approach is used to assess the potential environmental risk of a product before allowing it to the European market (EU, 2002). The first step (Tier 1) of this procedure is developed to determine the potential risk of a product with a minimum of data, such as toxicity data on algae, a crustacean and a fish and some information about the physical characteristics and use of the product. If the toxicity data in combination with the expected field concentrations indicate a potential environmental risk the substance will not be allowed on the market, unless it can be proven that the application under realistic field conditions will not have unacceptable impact on the ecosystem. This is where the higher tier studies come in need: non-standard studies tailored to give the answers that could lead to a better assessment of the environmental risk of the product. The types of studies can vary from ‘simply’ testing additional species under more or less controlled conditions towards the use of experimental ecosystems (micro- and mesocosms) which simulate an almost complete (aquatic) ecosystem. The results from these higher tier studies can overrule the conclusion based on the tier 1 data set. In principle this can go either way: the risk is less then expected e.g. to for instance the rapid degradation of the test substance under natural conditions or the risk can appear to be higher, for instance due to the fact that the tested ecosystem included more sensitive species than those used in the Tier 1 tests. The main point of this approach is that this procedure takes into account the fact that a laboratory test situation can significantly differ from what happens in the field.

In comparison with other forms of environmental risk assessment this procedure is rather sophisticated. The impact of a point source on a river basin for instance is still determined by using a dilution model to predict the potential environmental concentration (PEC) and to combine these values with the ‘predicted no effect concentrations’ (PNECs) for the individual substances as determined in standardised laboratory tests (Schobben & Scholten, 1993). State of the art is the use of species sensitivity distributions (SSD) to calculate a PNEC that is protective for 95% of the species (Posthuma et al., 2002), but still based on results from laboratory toxicity test performed under standard conditions.

Nowadays it is recognised that the potential risk of a contamination based on a simple PEC:PNEC ratio could well overestimate the actual risk, when the matter of bioavailability is not addressed. Therefore uptake routes and fate of the substances get more attention and recent exposure-models spend a lot of effort in describing the development of the internal tissue concentration in exposed organisms taking environmental conditions into account.

Thus the modern environmental risk assessment procedures apply sophisticated models to determine the exposure levels of contaminants as well as modern statistical techniques to determine the sensitivity of a community, while the data that is used to underpin this sensitivity is still derived from rather simplistic ecotoxicological test procedures. Therefore, we want to make a plea for the use of more complex toxicity studies in environmental risk assessment in order to bring the assessment of the ecosystem sensitivity to the same level. This will not only increase the relevance and applicability of environmental risk modelling, but might also give insight in the reason why in some situations the good ecological status is not reached, while the chemical status gives no reason for concern.

In addition, long term experiments under realistic field conditions may be needed to determine (validate) the field relevance of existing and new to be developed biomarkers.
Thesis 1: More complex experimentation is necessary to increase the understand-
ing of ecosystem sensitivity

Sensitive life stages are not always taken into account
A community is as sensitive as during its most sensitive period. It is recognised that the early life
stages of most organisms are the most sensitive for contamination (Hutchinton et al., 1989) However,
most effect concentrations that are used in ERA are derived from bioassays that are performed with
full developed organisms. In the search for short term cost effective bioassays some information can
get lost leading to underestimation of the risk.
Recent research with the flatfish sole (Solea solea) for instance showed that the most sensitive
moment in the development of this (and other?) fish species takes place after a standard test would
have been terminated. For dioxin like PCBs prolongation of the test would lead to a 40 times lower
NOEC, even when exposure took only placed during the egg stage (Foekema et al., in prep). Similar
results were found in prolonged tests with amphibians (Gutleb, 2006). Therefore effect data should not
be based on standardised tests only.

The environmental impact of mixtures can be hard to predict
The problem with complex mixtures of contaminants, as can be found in sediments and effluents, is
that chemical analysis will only reveal information about the substances that are analysed. Unknown
or unexpected substances will not be discovered. Therefore sensitive bioassays can add valuable
information about the (potential) environmental impact of a sample.
Besides taking effects of not analysed substances into account, a bioassay will also give information
about the effect of the whole mixture. The effect concentrations that are applied as NEC in
environmental risk assessment are in general based on laboratory test with single compounds.
Exceptions to this rule are formed by substances that cause non-specific (narcotising) effects (Wezel &
Opperhuizen, 1995) or substances with comparable modes of action as for instance dioxins and
coplanar PCBs (Berg et al., 1998). However, also combinations of other substances can have more
adverse effects then what would be expected by simply adding the effects of the individual compounds
(Nakayama et al., 2005). For this reason bioassays to assess the actual risk of environmental samples
should have a solid position in environmental health assessment.

Chemical stress impacts a species’ tolerance for temporal unfavourable environmental conditions
A species can survive in principle in any area that serves its needs as long as the environmental
conditions remain within the species specific tolerance range. In the centre of its geographical
distribution area the species finds its optimum conditions for survival, growth and reproduction. These
conditions allow the efficient use of resources/energy, and thus improve the competitive strength of
the species.
Experimental data show that the tolerance for unfavourable environmental conditions of an organism
can be significantly reduced by toxicant exposure. Blue mussels (Mytilus edulis) have a relatively
wide tolerance for low salinity, which allows the species to settle and survive in estuaries and
harbours. Healthy mussels are even capable of surviving longer periods of extreme low salinity
However, after being exposed to copper this tolerance is significantly reduced. Besides this example
our research showed that the tolerance for low (winter) temperatures of Spisula subtruncata was
reduced by chronic exposure to PAHs contaminated sediments, that the capability to survive periods
of anoxia of both Dreissena polymorpha and Mytilus edulis are reduced by exposure to heavy metals
and that copper exposure reduced the tolerance for high temperatures for the crustacean Neomysis
integer. This toxicant induced reduced tolerance for environmental conditions has also been found by
others (see for instance Heugens et al., 2003). Based on these data we conclude that chemical stress
has the potential to narrow down the ecological tolerance curve of populations. This could result in
changes in the distribution and abundance of species.
Thesis 2: Experimental ecosystem studies are needed to determine the applicability and ecological relevance of biomarkers

During the last decades various biomarkers have been developed (for instance VTG, Cyp1A, P450, etc.), mainly based on biochemical/physiological reaction of organisms that are induced (amongst other factors) by exposure to anthropogenic contaminants. Although these markers for sure tell us something about the individual’s physiology, the relation with the contaminant exposure is not always clear. In the controlled laboratory environment where (most?) biomarkers are discovered the relation with toxicant exposure is clear and reproducible, but in the wild things look often more complicated (Forbes et al., 2006).

Moreover, for the majority of biomarkers the ecological relevance is at least vague. Without this information a so called ‘effect-biomarker’ is, at its best, just another ‘exposure biomarker’. The natural variation often hampers to address these issues in the field. Experimental ecosystems are a suitable tool to address these issues, since they allow 1) a long term experimental set-up, 2) the controlled manipulation of the environmental and exposure conditions, 3) treatment replication and 4) appropriate reference conditions. This validation of the ecological relevance of exciting and new (‘-omics’) biomarkers is essential for the applicability of these tools in environmental health assessment.

Recommendations & perspectives

We believe that the reliability and applicability of environmental risk assessment procedures will be improved when more attention is given to the aspects that influence the sensitivity of the exposed organisms and ecosystems. In the same way that exposure assessment is being lifted to a higher level by including environmental conditions (bioavailability) into the procedure, the assessment of the sensitivity should be developed in the same way. Higher tier studies will be necessary to achieve the knowledge that is needed for the development and validation of sophisticated effect models and biomarkers that take the impact of (temporarily) environmental conditions into account and that can be translated into ecological relevant endpoints.

Wageningen IMARES is planning to make the first steps towards the development of such an effect model in 2008.

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How to link risk assessment to current understanding of community-level effects?

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Background

A good water quality is fundamental to a sustainable development of the human society. This societal need prompted the establishment of a legal framework for the protection of European aquatic systems. The WFD (2000/60/EC) builds a new legislative framework to guarantee a good status of European water resources within river basins as well as transitional and coastal zones until 2015. Toxic pressure is a relevant driving force of the deterioration of European surface waters (e.g. Brunsberg & Blomqvist, 2001) and reduced biodiversity of aquatic ecosystems and is therefore addressed in the WFD by the assessment of the chemical status and by the evaluation of aquatic community assemblies (ecological status).

While in pesticide assessment and regulation higher tier assessment, including community-level investigations, is considered, an important remaining question includes whether the regulatory chemical risk assessment methodology, mainly derived from standard toxicity testing based on single species test batteries is appropriate for deriving water column environmental quality standards (EQS) (Crane & Babute, 2007). Based on the assumptions that communities are more than the sum of populations (thesis 1) and that the protection of aquatic communities maintains ecosystem processes (Calow and Forbes, 2003), the consideration of community-level effects may have implications for risk assessment of chemicals in aquatic systems. It is the aim of this contribution to outline the significance and challenges of community-level effects for environmental risk assessment of chemicals.

Thesis 1: Communities are more than a sum of populations

Communities are defined as a group of interacting populations that overlap in space and time (Clements and Newman, 2002). Therefore, communities are characterised by distinct qualities which are beyond population level: patterns of the community structure which are described by species diversity and the dominance structure of an assembly, as well as factors regulating these patterns. These factors could be external (abiotic factors influencing the community, e.g. flow velocity changing the bioavailability of a toxicant) or intrinsic (species interaction and (multi-) trophic interaction, e.g. grazing). These community qualities are not represented in several standard toxicity testing procedures but may have implications for the sensitivity of a community to a toxicant and therefore the assessment of risks on communities in the environment.

Thesis 2: Understanding of community-level changes in aquatic systems needs understanding of species interaction

Current risk assessment of chemicals in aquatic systems is mainly based on toxicity testing using single species cultures. To extrapolate results from individual- or population-level endpoints to community-level effects in aquatic systems, assessment approaches were developed with the scope to incorporate community characteristics in risk assessment of chemicals: Communities are composed of several species differing in there normal operation range, therefore they differ in their sensitivity to toxicants. The diversity of sensitivities of exposed species is considered when applying the concept of species sensitivity distributions (SSD). From SSDs predicted no effect concentrations (PNECs) are calculated that should be protective for 95 % of species (Posthuma et al., 2002). In the environment, communities are mainly faced with sublethal exposure to toxicants over longer periods. Commonly applied methods in toxicity evaluation are based on acute toxicity testing. With respect to these differences in exposure times, acute to chronic ratios were calculated to extrapolate from acute exposure to chronic exposure scenarios.

An additional community characteristic with implications to effect assessment on community level is species interaction. However, species interaction was rarely included in risk assessment strategies, so
far, but may be of relevance when determining effect thresholds in communities. An example will be presented from a microcosm study using biofilms communities. These communities were exposed to a concentration series of an herbicide during community succession. Species interaction of microalgae was evaluated in terms of pollution-induced community tolerance (PICT). Comparison of the above mentioned approaches with the microcosm study using PICT revealed lowest effect thresholds for the community-level approach (McClellan et al., submitted) indicating that species interaction may be a relevant community characteristic determining community sensitivity to toxicants. Therefore, understanding species interaction may reduce uncertainty when extrapolating adverse effects of toxicants to ecosystems.

**Thesis 3: For retrospective risk assessment at multiple stressed sites causal links of effects of toxicants on community-level are needed**

Aquatic communities are subject to several stressors in aquatic systems. The determination of an insufficient ecological status, based on the biological quality elements (BQEs) is not directly attributed to a defined stressor quality. Therefore, a causal link of community-level effects to a stressor is essential for assessing risks on aquatic systems, appropriately, as well as for the allocation of restoration efforts. However, ranking systems of multiple stressors hardly exists (Håkanson, 1999) and risk assessment strategies in multiple stressed environments are still limited. Based on the PICT-concept a retrospective causal link of a group of chemicals or even a single toxicant to community-level effects in a multiple contaminated aquatic system is possible (Blanck, 2002; Schmitt-Jansen et al., in press), even though restricted to a selected group of chemicals and biological groups, at present. An example will be presented from the Elbe river basin. Sediment extracts from a heavily polluted site have been evaluated by effect-directed analysis and prometryn was identified as one of the keytoxicants (Brack et al., 1999). Communities, grown at the polluted river site showed higher tolerance to the selected toxicant in comparison to the reference sites (figure 1) providing a causal link of community changes to a selected stressor at this site of investigation.

![Concentration-response relationships of prometryn derived from biofilms grown at a reference site and a site contaminated with prometryn in the Elbe river basin.](image)

Fig. 1: Concentration-response relationships of prometryn derived from biofilms grown at a reference site and a site contaminated with prometryn in the Elbe river basin. Shifts in EC50-values indicate pollution-induced community tolerance (PICT). (Schmitt-Jansen et al., in press).
Recommendations & perspectives

To conclude from the above considerations, extrapolations derived from individual- or population-level endpoints might be misleading with respect to risk assessment of chemicals to communities. Including community-level endpoints in management schemes and integrated assessment strategies for retrospective as well as prospective assessment of chemicals might be more protective for ecosystems and improve reliability and certainty of risk management decisions for ecosystems. Causal links of community-level effects to specific stressors (and toxicants, respectively) are needed. A (limited) set of approaches exist, providing these links, e.g. by applying the concept of pollution-induced community tolerance, however, future efforts should focus on providing these links.

References


With a background in limnology and aquatic ecotoxicology, current research interests of Mechthild Schmitt-Jansen comprise biofilm community ecology, indirect effects of toxicants on community level and pollution-induced community tolerance applied in in-situ approaches as well as in model ecosystems.

Major involvements in projects are the EU-IP MODELKEY (subproject SITE) and the EU-Marie Curie RTN KEYBIOEFFECTS with focus on metabolite diversity in microalgal communities.
Estrogenic compounds in Spanish river basins - Implications for human and ecological risk assessment

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Background

Spanish river basins, as many other river basins all over the world, have been shown to contain estrogenic contaminants and their fauna to be affected by alarming estrogenic alterations such as feminization and hermaphroditism (Solé et al. 2000; Petrovic et al. 2002; Cespedes et al. 2005 and 2006). Studies integrating both chemical and biological analysis have pointed out estrogens and alkylphenols as the main responsible compounds for the estrogenic effects observed (Routledge et al. 1998; Petrovic et al. 2002; Sumpter et al. 2005). However, the range of compounds classes with suspected estrogenic properties is much wider; it includes many other extensively used chemicals, such phthalates or bisphenol A. In the last decades, many efforts have been dedicated to the study of these compounds; however, our knowledge about them is still very limited. Their sources, environmental fate, mode of action, estrogenic potency, routes of exposure, human and environmental impact, environmental occurrence, etc. are still largely ignored. In addition, many more compounds can still be discovered as estrogenic contaminants.

Thesis 1: Estrogenicity assays provide enough information to use them as first instance monitoring tools in routine control.

The estrogenic activity present in a river basin can be assessed by chemical and/or biological methods. Chemical methods are used to control specific substances and/or to characterise the nature (causative agents) of the estrogenicity observed. Biological assays, such as the estrogen receptor (ER)-CALUX, and the recombinant yeast assay (RYA), are used to measure the overall estrogenic activity (without discriminating the responsible substances) (García-Reyero et al. 2001, Céspedes et al. 2005). Some authors argue that estrogenicity assays provide enough information for routine control and risk assessment in river basins and that they should be included in the legislation as monitoring tools relegating the use of chemical methods to those situations where the bioassays are positive. One of the problems arising in this respect is the lack of standardized validated methods (Petrovic et al. 2004).

Thesis 2: Conventional (activated sludge) treatments do not completely remove estrogenic compounds from wastewater influents.

Due to their continuous introduction in the aquatic environment, estrogenic contaminants do not need to be persistent to reach environmental levels of concern. Sewage treatment plant (STP) effluents are considered the major source of estrogenic compounds in the aquatic environment. This is because most STPs only have primary and secondary (activated sludge) treatments and these treatments are not completely efficient at removing estrogenic compounds. Terciary treatments, such ozonation and activated carbon filtration, as well as other alternative methods, such as bioreactors, are more effective (Petrovic et al. 2003, Rodríguez-Mozaz et al. 2004). The question is whether the implementation of these advanced treatments in both urban and industrial STPs, and the economical inversion behind, is deemed justified and necessary.

Thesis 3: Estrogenic effects caused by environmental contaminants are well documented in aquatic organisms but not in humans.

There are enough evidences that estrogenic contaminants, such as estrogens and alkylphenolic compounds, induce adverse alterations, such as feminization and hermaphroditism, in aquatic organisms (e.g., fish). However, in humans, estrogenic effects and alterations in relation to estrogenic contaminants are not so clear and need further study (Kuster et al. 2007). Future research in this field of human risk assessment requires the close collaboration of environmentalists and physicians and the
performance of epidemiological studies to establish solid connections between the observed pathologies (e.g. decreased fertility) and the causative agents, sources and vias of exposure.

**Thesis 4: Restriction or prohibition of use has lead to decreased environmental levels of some estrogenic compounds.**

Restriction, prohibition of use or substitution of some estrogenic compounds by other less harmful chemicals has lead to gradually decreased environmental levels of the substances of concern. This has been, for instance, the case of nonylphenol ethoxylates. In the past, these compounds were used in both the domestic and the industrial field. At present, their use is restricted to the latter one and this has lead to a gradual decrease of their environmental levels (González et al. 2004). The application of this kind of measures, however, is not always possible. For instance, estrogens used in human medicine as contraceptives or in hormonal therapies (e.g. treatment of menopause) are difficult to substitute.

**Recommendations & perspectives**

Future research needs to continue investigating the environmental occurrence of estrogenic compounds, developing standardized estrogenicity bioassays, studying the removal efficiency of current and advanced wastewater treatment methods, searching for substitutive, less harmful chemicals, and performing epidemiological studies for human risk assessment. Recommended actions to diminish the presence and effects of estrogenic contaminants in river basins are the establishment of maximum allowable concentrations for compounds not yet regulated such as estrogens, the increment of the number of STPs, the incorporation of tertiary or more advanced water treatment processes in STPs, and the application of source-point, specific water treatments in industries.

**References**


Research lines: environmental analytical chemistry, priority (pesticides, phenols, phthalates, PAHs) and emerging (estrogens, progestogens, polar pesticides, antibiotics, drugs of abuse) organic contaminants, advanced analytical techniques (pressurised fluid extraction, on-line solid phase extraction, liquid chromatography-tandem mass spectrometry, biosensors, passive samplers).

Current projects: MODELKEY (GOCE-511237); AQUATERRA (GOCE-505428); CEMAGUA (CGL2007-64551/HID); SOSTAQUA (CENIT); EVITA (CTM2004-06265-C03-01).
Llobregat case study: How to elucidate the role of environmental factors and toxicants on biological community?

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Background

During the last decade, environmental policies in Europe have addressed the improvement of ecosystem function, with a particular focus on large geographical areas. With respect to freshwater systems, the Water Framework Directive requires that member states and the scientific community consider the protection and management of entire freshwater systems. To determine the ecological status of water bodies, the WFD has put forward an innovative concept of bio-assessment, namely not the impact of single pressures on individual biotic groups but deviation of the community from undisturbed conditions (Birk and Hering, 2006).

The assessment and conservation of biodiversity have been the focus of much attention in recent years. Trends and causes of species loss have been discussed at local, national and global levels and provide the bases for ecological risk assessment in areas strongly affected by human activity.

It is often difficult to establish relationships between the chemical characteristics of river water and biological community status. The European Community’s Modelkey project seeks to develop several tools to analyse data on the community, habitat, chemicals and toxicity in order to assess the impact of key pollutants on community structure and biodiversity. Moreover, this project addresses this analysis at basin level.

The Llobregat River is the Mediterranean basin examined by the project. The catchment is mainly calcareous with substantial salt deposits (chloride, sodium, potassium). The hydrological regime is characterized by high discharge fluctuations, which reflect the Mediterranean climate. The mean annual discharge of this watercourse is around 21 m³ s⁻¹, with very low summer values and flood peaks in spring and autumn as a result of heavy storms.

More than 5 million people live in the Llobregat basin. The headwaters of this river are characterized by abundant agricultural activities while middle and lower reaches have dense industrialized areas. Eutrophication is high in this river. In the study site, the mean concentration of soluble reactive phosphorous (SRP) is around 275 µg/L (minimum values of 16 µg/L and maximum of 580 µg/L) and mean nitrate concentrations around 7 mg/L (minimum of 1 mg/L and maximum of 22 mg/L). Numerous pollutants, including heavy metals, pesticides, PAH, etc., can be found in the river water.

In several sites of the lower part of the river the chemical analyses revealed the presence of emerging contaminants, some of them endocrine disrupter compounds (EDC’s): estrogens and progestogens and pharmaceuticals and alkylphenolic compounds (Garcia-Reyero et al., 2001, Petrovic at al., 2002).

Multivariate analysis link assemblage patterns to abiotic variables and help us to determine the importance of the organic pollutants stress on biological community (Clarke and Ainsworth 1993, Clarke, 1999).

Thesis 1: Interdisciplinary field sampling to assess the role of organic pollution on biological community

Hydrology, salt concentration, eutrophication, and toxic pollution affect the structure and diversity of the benthic community. To analyse the relationships between the community and water quality and to determine the weight of these stressors on species, we designed a simultaneous sampling protocol of chemical and biological parameters along the lower reach of the Llobregat River. The biological, chemical and toxicant parameters analysed are listed in the table 1.
Table 1: List of biological, chemical and toxicant parameters measured in the Llobregat River.

<table>
<thead>
<tr>
<th>Biological parameters</th>
<th>Physico-chemical parameters</th>
<th>Toxicant parameters</th>
</tr>
</thead>
<tbody>
<tr>
<td>diatom composition and density in the substrate</td>
<td>temperature</td>
<td>Metal concentration (Ba, Pb, Al, Mn, Zn)</td>
</tr>
<tr>
<td>macroinvertebrate density and biomass in the soft sediment</td>
<td>oxygen concentration</td>
<td>Pesticides (Bentazone, 2,4-D, MCPA, Mecoprop, Propanil,</td>
</tr>
<tr>
<td>bacterial density</td>
<td>pH</td>
<td>Fenitrothion, Isoproturon, Atrazine, Diuron, Deisopropylatrazine)</td>
</tr>
<tr>
<td>exoenzimatic activities</td>
<td>nutrient concentration (phosphate, total P, nitrate, nitrite, ammonium)</td>
<td></td>
</tr>
<tr>
<td>chlorophyll a concentration</td>
<td>anions and cations</td>
<td>Estrogenic compounds (Estrone, Estradiol, Estriol, EE, DES, E1, E2, E3)</td>
</tr>
<tr>
<td></td>
<td>TOC</td>
<td>Alkylphenolic compounds (LAS, CDEA, NP1, NP2, NP9, OP1EC)</td>
</tr>
<tr>
<td></td>
<td>suspended solids</td>
<td>Pharmaceuticals drugs (Analgesics and anti-inflammatories, lipid regulators and</td>
</tr>
<tr>
<td></td>
<td></td>
<td>cholesterol lowering statin drugs, psychiatric medication, antiulcer agent, histamine</td>
</tr>
<tr>
<td></td>
<td></td>
<td>H1 and H2 receptor antagonists, antibiotics, β-blockers)</td>
</tr>
</tbody>
</table>

Sampling campaigns were done in spring and autumn 2005. We examined seven sites along a pollution gradient in the main course of the river and in the tributary Anoia River. Chemical and biological results were contrasted with the routines BEST and LINKTREE in the PRIMER v. 6, statistical package (Plymouth Marine Laboratory, UK, 2006). This procedure was applied to find the best match between the multivariate among-sample patterns of an assemblage and that from environmental variables associated with those samples.

**Thesis 2: Evidences of emergent toxicant effects on community**

Preliminary results of the Llobregat basin analysis show that the abundance of macroinvertebrate communities is correlated with parameters related to organic pollution, mainly nitrates and phosphorus. Biomass is also correlated with temperature, thereby reflecting the seasonality in its values. Oligochaeta and *Chironomus* spp showed increases in the most polluted sites. Diatoms were correlated with nitrate but also with sulphates and sodium although these correlations were not significant.

Diatom abundance was not correlated with any organic toxicant; however, high concentrations of alkylphenolic compounds (mainly Las, NP1, NP2, OP1EC) and some pharmaceutical agents (indomethacin, erythromycin, propanolol and ibuprofen) were significantly correlated (Spearman’s coefficient around 0.5) with macroinvertebrate assemblage.

Further analysis is required to confirm these relationships. Two more sampling campaigns were done in 2006 and the results will be processed and compared with those for 2005.
Recommendations & perspectives

On the basis of our experience and these preliminary results, we propose the following suggestions be taken to improve the assessment and risk-based management of European river basins:

- Simultaneous samplings in time and space for chemical and biological parameters to identify the cause-effect relationships between them.
- General screenings in river basins to detect the presence of new compounds and to determine their potential toxicity. Future laboratory tests can help us to select those to be included in routine samplings and to be covered by legislation.
- Moreover, in most of the river basins benthic communities show a high degree of degradation (low number of species and high tolerant to pollution). It is therefore difficult to detect the effects of new contaminants, and discern whether these effects are additive or not to environmental factors (eutrophication, salt concentration, suspended solids, seasonality, etc.). Multivariate analysis may provide a useful tool for this analysis; however, complementary approaches are required for the risk-based management.

References


Isabel Muñoz Gracia

Ph.D. in Biology. Full professor at the Department of Ecology at the University of Barcelona. She has been involved in national and international projects focused on structure and function of stream invertebrate community, ecotoxicology and water biological quality. She has participated in 3 different EU programs within the 4th, 5th and 6th framework. She is now coordinator of a national research project titled “Biological key processes determining functioning of Mediterranean stream ecosystems” and the leader of the Llobregat river basin in the Modelkey EU project.
Impact of pollution on biodiversity: From basin to site scale in the Scheldt river basin

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Background

Assessment and conservation of biodiversity has been one of the important topics in recent years. Trends and causes of species loss have been discussed at local, national and global scales and as such highlighted by several authors (Linke & Norris, 2003; Kamppinen & Walls, 1999; Reyers & James, 1999). These studies are the basis for ecological risk assessment of areas with high human influence that puts ecosystems at risk. A bad ecological status and a reduced biodiversity of aquatic ecosystems can be caused by three main factors namely deterioration of the habitat quality, eutrophication or pollution. It is not clear yet whether it is possible to distinguish the relative impact of these three causes on a basin scale or at a site scale. However it would be useful if the monitoring data gathered within programs implemented in function of Water Framework Directive (WFD) could indicate the major disturbing causes. Within the EU-WFD assessment of the macro-invertebrate monitoring data is mainly using biotic indices, which are well developed but do not always take into account biodiversity, biomass and density. Within the MODELKEY project we analysed first of all the available monitoring data, which are not determined to species level, to see whether it was possible to distinguish quality of the several subbasins. Further an intensive survey of the macro-invertebrate communities at a number of selected sites, upstream and downstream of a pollution source, was done. Determination of all groups was done to species level, density and biomass were measured. These data were judged to see whether the results can give a better indication of the main disturbing factors at these sites.

Aims

The aim of this study is to see how much information the macro-invertebrate community can give us to understand the dominating stressors, with special emphasis on the impact of pollution.

Thesis 1: The required level of determination of macro-invertebrates depends on the type of monitoring

The available monitoring data of the Flemish Environment Agency contains data of macro-invertebrates where the determination level for a lot of the groups is only done to family level. However it is possible to distinguish the different subbasins within the Scheldt river basin based on these data. The highest number of families that is found within the subbasins varies between 20 and 28, however the highest numbers are only found in the Nete basin, which is considered as one of the cleanest subbasins. The other three subbasins are heavily disturbed. If we look at the list of families which are most common in the subbasins (Table 1), it is clear that in the heavily disturbed subbasins the communities are dominated by a limited number (5) of families, which are known to be tolerant groups, while in the Nete basin the communities are dominated by 9 families. Application of the SPEAR index (Lies & Von der Ohe, 2005) indicates that the lack of some of the families in the polluted subbasins might be due to pesticides.
Table 1: List of families contributing to 50% of similarity between samples (sites) in four subbasins of the Scheldt river basin. Families are listed in order of importance.

<table>
<thead>
<tr>
<th>Beneden-Scheldt</th>
<th>Boven-Scheldt</th>
<th>Dijle-Zenne</th>
<th>Nete</th>
</tr>
</thead>
<tbody>
<tr>
<td>Tubificidae</td>
<td>Tubificidae</td>
<td>Tubificidae</td>
<td>Tubificidae</td>
</tr>
<tr>
<td>Chironomidae</td>
<td>Chironomidae</td>
<td>Chironomidae</td>
<td>Chironomidae</td>
</tr>
<tr>
<td>Naididae</td>
<td>Glossiphoniidae</td>
<td>Glossiphoniidae</td>
<td>Asellidae</td>
</tr>
<tr>
<td>Glossiphoniidae</td>
<td>Asellidae</td>
<td>Asellidae</td>
<td>Lymnaeidae</td>
</tr>
<tr>
<td>Nematoda</td>
<td>Psychodidae</td>
<td>Erpobdellidae</td>
<td>Sphaeriidae</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>Naididae</td>
</tr>
</tbody>
</table>

Thesis 2: Biotic indices developed on the basis of organic load/oxygen needs and which do not take into account (relative) abundance are no good indicators for pollution

Assessment of macro-invertebrate data is often done with biotic indices that are based on the sensitivity of organisms to oxygen deficiency, such as the Biotic Sediment Index, the Belgian Biotic Index, the Multi Criteria Analysis and the Multi metric index. Taxonomic resolution used and thus the distinction between different sites is limited (Verdonschot, 2006). The analyses of data sampled upstream and downstream of a pollution source in the Grote Nete and in the Schijn shows clearly that there is no difference in classification when using these biotic indices (Table 2). Only the oligochaete index for sediments distinguishes a difference between the reference site and the polluted sites in the Grote Nete. This index takes into account relative abundance and it is even thought that based on the percentage of tubificidae without or without hair setae it is possible to have an indication whether toxicity is most likely due to metals or organic compounds (Prygiel et al, 2000).

Table 2: The classification of the macro-invertebrate community of reference and polluted sites in two streams of the Scheldt river basin using different biotic indices. Biotic sediment index (BSI), Belgian Biotic index (BBI), Multi Criteria Analysis (MCA), Multi metric index (MMI) and the Oligochaete index for sediments (IOBS) taking into account the percentage of tubificidae without hair setae (TUSP).

<table>
<thead>
<tr>
<th>Index</th>
<th>Schijn reference</th>
<th>Schijn polluted</th>
<th>Grote Nete reference</th>
<th>Grote Nete Polluted</th>
<th>Grote Nete Heavily polluted</th>
</tr>
</thead>
<tbody>
<tr>
<td>BSI</td>
<td>Low</td>
<td>low</td>
<td>low</td>
<td>Low</td>
<td>Low</td>
</tr>
<tr>
<td>BBI</td>
<td>Low</td>
<td>medium</td>
<td>low</td>
<td>low</td>
<td>low</td>
</tr>
<tr>
<td>MCA</td>
<td>medium</td>
<td>Medium</td>
<td>medium</td>
<td>medium</td>
<td>medium</td>
</tr>
<tr>
<td>MMI</td>
<td>low</td>
<td>Low</td>
<td>medium</td>
<td>medium</td>
<td></td>
</tr>
<tr>
<td>IOBS (+TUSP)</td>
<td>low</td>
<td>Severe</td>
<td>severe</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

Thesis 3: The impact of pollution is reflected clearly in the macro-invertebrate based indicators abundance, biomass and diversity

In freshwater systems is, contrary to transitional and coastal water systems, assessment of the macro-invertebrate communities mainly based on the presence or absence of taxa. In the transitional and coastal waters biomass and density are always considered, which gives a lot of extra information considering the health of the community and of the function within the food web. At the selected sites within this study, both situated in freshwater streams, biomass and density of the macro-invertebrates
was also measured. The results show a clear difference between the three sites in the Grote Nete (figure 1). The number of species that is responsible for 80 percent of the abundance and also the biomass is significantly lower at the heavily polluted site. The results also indicate that diversity is higher at the unimpacted site. This is confirmed when the Shannon-Wiener index is applied at the data. Both in the Grote Nete and in the Schijn diversity is significantly higher at the reference site compared to the polluted site, however the data show also that diversity at “reference” sites differs between different streams (figure 2).

Fig. 1: K-dominance curves of three sites along a pollution gradient in the Grote Nete. Left: calculated with abundance data, right: calculated with biomass data (in MS® Excel).

Fig. 2: Bootstrap estimates (2000 permutations) for the Shannon-Wiener index of diversity, calculated from data sampled in spring 2005. Vertical bars give the 95% confidence limits.
Recommendations & perspectives

Assessing the macro-invertebrate community based on a low taxonomic resolution and absence/presence is convenient for surveillance monitoring but certainly for investigative monitoring determination to species level, biomass and density can give valuable information, helping to detect the major stressors. Biomass and density data however are scarce and might be a topic for further research.

References


Eric de Deckere, an environmental scientist, is working mainly on Integrated Water Management with special emphasis on the relation between the biotic community and sediments and the impact of pollution. Major projects in which he is involved: MODELKEY, MANUDYN (macrophyte and nutrient dynamics in the Scheldt river basin), several Flemish projects related to the implementation of the WFD.
Risk assessment in European river basins - What can we learn from the Danube case study?

Jaroslav Slobodnik1 & Igor Liska2

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2International Commission for the Protection of the Danube River, Vienna, Austria

Background

Results of a chemical, biological and ecotoxicological survey on a large reach of the Danube River are presented and compared with the official risk analysis according to the EU WFD. The comparison necessitates further data collection and analysis to be able to classify the risk status.

Thesis 1: Chemical, biological and ecotoxicological evaluation of the Danube by the Aquaterra survey

The AQUATERRA Danube Survey (ADS) took place from 19 August to 6 September 2004. During this survey samples of sediments and suspended matter were taken from 30 sampling sites alongside the 1147 km stretch of the Danube from Klosterneuburg, Austria (rkm 1942) to Calafat, Romania (rkm 795). The samples were analyzed for basic physico-chemical parameters, organic micropollutants and heavy metals. Moreover, analyses of macrozoobenthos and ecotoxicological tests were performed.

As for nutrients, total nitrogen content in sediments was slightly increased downstream the Danube. Variations in total nitrogen content in sediment samples, correlated with dissolved oxygen in the water column and were associated with denitrification rate and high resuspension frequencies in respective sites. Total phosphorous content in sediments had a slightly increasing character from the upper to the lower section. Total Organic Carbon content in sediment followed a relative uniform profile along the monitored stretch of the Danube.

Among the organic micropollutants the most significant profiles were observed for polycyclic aromatic hydrocarbons. Most of the elevated summary concentrations of PAHs were detected in the section between rkm 1846 and 1560 (Fig 1). The concentration patterns for PCBs, organochlorinated pesticides, alkylphenols, DEHP and bisphenol A were rather random. Next to the target analyses for organic substances a GC-MS screening was carried out. Its results showed that the sediments and suspended matters contained typically around 120 and 65 detected compounds, respectively. A general trend of increase of the number of detected substances in the lower reach of the Danube (typically over 300 detected compounds after the Sava confluence) was observed. The GC-MS screening provided a lot of useful information on the design of a general pollution pattern of the Danube river, e.g., indicating significant pollution impacts from either left or right bank sources.

The analysis of heavy metals revealed a significant upward trend of concentrations of cadmium, lead and nickel in sediments downstream the Danube. For mercury, the concentration maxima in sediments were observed in the middle part of the monitored river stretch. For heavy metal concentrations in suspended particulate matter a clear increasing tendency downstream was recorded for all measurands.

The evaluation of the macroinvertebrate community was based on the results of the kick samples (Figure 1). The range of total number of macroinvertebrate taxa detected at the different 30 cross sections varies between 9 and 36. Altogether three decreasing and subsequent three increasing series of taxon numbers were observed. The first continuous decrease was recorded from Klosterneuburg to the last site of Gabčíkovo reservoir. A significant increase could be then recognised after the Gabčíkovo reservoir until the downstream section of Budapest where the highest taxon number was found (36). Generally it can be concluded that the reach between Szob and Budapest was the taxon-richest section of the investigated Danube. A similar behaviour was observed downstream Velika Morava and at Banatska Palanka/Bazias, both on the lower Danube having 30 and 26 taxa, respectively. Analysis of saprobic index revealed that the worst section was registered in the Serbian Danube between Novi Sad and the Iron Gate Reservoir.
Altogether five toxicity tests were used within the Aquaterra Danube study: growth inhibition tests of Lemna minor, algal growth inhibition test on Desmodesmus subspicatus, growth inhibition test with Sinapis alba, inhibition of the mobility of Daphnia magna and inhibition of the light emission of Vibrio fischeri. The results of the ecotoxicological studies showed that the upper and middle part of the monitored reach of the Danube River appeared to be the most responsive in its longitudinal profile. In this reach the toxic effects to 3 or 4 species were detected (Fig. 2). In other parts of the Danube River positive effects were indicated only for smaller amount of tested species.

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**Fig. 1:** Number of macroinvertebrate taxa along the Danube between 1942 and 795 rkm.

**Fig. 2:** Number of the sensitive test species for the left and right Danube stations.

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Thesis 2: ICPDR approach for the risk assessment on surface waters

The WFD requests from the Member States to carry out an assessment of the likelihood that water bodies will fail to meet the environmental quality objectives by 2015. Failure to achieve the objectives on surface waters may be the result from a very wide range of pressures, including point source discharges, diffuse source discharges, water abstractions, water flow regulation and morphological alterations. These and any other pressures that could affect the status of aquatic ecosystems must be considered in the analysis. The risk assessment is therefore based on information collected in the pressure and impact analysis. In the Danube River Basin the first risk assessment according to WFD was performed in the Article 5 report (ICPDR Roof Report 2004). In this analysis the following three risk classes were defined: "water body not at risk" (based on the pressure/impact analysis it is estimated, that the investigated water bodies will reach the objectives set out by the WFD); "water body possibly at risk" (this category of water bodies, are those for which not enough data is available); "water body at risk" (based on the performed pressure/impact analysis it is estimated, that these water bodies are “at risk” of failing to meet the objectives set out by the WFD). The pressures and their resulting impacts were disaggregated into the following risk categories: (i) Organic pollution, (ii) Hazardous substances, (iii) Nutrient pollution and (iv) Hydromorphological alterations.

Data on the risk assessment are available for the total length of the Danube (Fig. 3). The upper Danube, where chains of hydropower plants exist, is mainly impacted by hydromorphological alterations. Many of the water bodies in the upper Danube have also been provisionally identified as "heavily modified water bodies". The Middle Danube is classified as “possibly at risk” due to hazardous substances for the largest part. The Danube section shared by Slovakia and Hungary is classified in part as “at risk” and in part as “possibly at risk” due to hydromorphological alterations. The part of the Danube shared by Croatia, and Serbia is “possibly at risk” in all categories since not enough data is available for a sure assessment. The lower Danube is “at risk” due to nutrient pollution and hazardous substances, and in large parts due to hydromorphological alterations. It is, and “possibly at risk” due to organic pollution.

![Danube risk classification by risk categories](image)

Fig. 3: Risk classification of the Danube, disaggregated into risk categories. Each full band represents the assessment for one risk category (hydromorphological alterations, hazardous substances, nutrient pollution, organic pollution). Colours indicate the risk classes. * SK territory.
Recommendations & perspectives

Results of a chemical, biological and ecotoxicological survey on the Danube river provided downstream profiles for numerous quality elements as well as significant responses of selected test organisms. The analysis of data collected recognizes upward trends downstream the Danube River (heavy metals, total N), downward trends (total P) and local maxima at certain reaches (PAHs, ecotoxicity, macrozoobenthos analysis, saprobic index). However, the analysis of the information acquired did not enable to achieve one of the major goals of the Danube Aquaterra study – correlation of biological and toxicological results with those from chemical measurements as the results did not show any significant relations. Therefore, for a better understanding of the risks it is necessary further to study the inter-relations between the particular datasets. To extend the available information, a new complex dataset will be shortly available as the results of the second Joint Danube Survey (JDS2) organized in August and September 2007 by the ICPDR are under preparation.

In the official Danube risk classification in category hazardous substances the largest part of the middle Danube is classified as possibly at risk. Analyzing the results of chemical analyses and, especially ecotoxicological tests from the Aquaterra Danube Survey there is an indication of significant levels of toxic substances. This single information is naturally not sufficient for reclassification of the middle Danube reach but the attention should be given to the results of JDS2 as well as to the data from the national surveillance monitoring activities in 2007 – 2008.

References


Dr. Jaroslav Slobodnik has over 15 years experience in international/national-scale programmes on various aspects of integrated river basin management. Among his specialisations are monitoring of river basins, implementation of programmes on assessment of water quality (chemical and ecological status), design and implementation of environmental information and data management systems and environmental analytical chemistry. He is currently involved in EU FP projects AQUATERRA, NORMAN, E AQC-WISE and SOCOPSE. As a member of the advisory panel to the DG ENV he was involved in drafting institutional reform proposals of the DWD and its harmonisation with WFD.
Risks from contaminated sediments to ecosystem services of a river catchment: Improved approaches and lessons learned from the Elbe case study

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Background

Sediment quality in many cases has levelled off at a still unsatisfactory quality status. After industrial point sources and communal waste waters ceased being the main problem of pollution, for a number of chemicals it is now the contaminated sediment or contaminated soil itself which acts as a significant secondary source. Widely distributed over vast areas in a catchment it potentially threatens a variety of ecosystem services and functions along a river. Management in Europe hesitates to address these contaminated volumes for several reasons, some of which are partly due to the fact that it is unclear who is supposed to do what, as

• the contamination may come from another era and nobody can be held responsible today,
• the severeness of the risk coming from a specific area where measures could start is not clear due to the presence of a number of different potential sources for a contaminant in a river basin,
• the costs of dealing with the contamination can become immense.

We carried out an extensive study on the Elbe river using available data from a number of institutions and Federal States on sediment and SPM quality and loads in order to prioritize those sites which contribute most to the potential risk of ecosystem services. The hypotheses stem from the experiences with this study.

Thesis 1: Sustainable sediment management on river basins can only be successful if system understanding and the acceptance of a “joint catchment” idea succeeds over “protecting ones own turf” ambitions. We are not there yet.

The different perspectives of decision makers, whose interests can be e.g. harbour management, provision of ecosystem services, public health or recreational use, requires respect towards the importance of their values in a communication process. This and the knowledge of the implication of terms and 'blind spots' which can be different with different stakeholders asks for skilled facilitation of a decision making process. The Elbe river, for example, offers a number of ecosystem services, but also presents a variety of problems connected to contaminated sediment, which are in many cases politically sensitive.

Are key players still unaware of the problem of sediment management? Is there an unwillingness to deal with the issue or could the scientists have made a better job in communicating the problems?

What can scientists do to provide a sound basis for decision making and avoid being accused of taking sites? If they follow a mechanistic assessment scheme, the outcome of which surprises them as much as everybody else, did they get their values wrong? Do they deny responsibility or do they provide sound science? This presentation will not answer these questions but will demonstrate exemplarily the necessity to communicate the basis of scientific approaches and will point out the risk of misunderstanding.
Thesis 2: River Basin management of sediments requires different kinds of data than have been collected before and a re-evaluation of monitoring strategies and their objectives.

The Elbe Study on the risk from contaminated sediments to ecosystem functions showed that suspended matter needs to come into focus, when potential environmental impacts by contaminants are supposed to be estimated in a river basin. This also concerns ecotoxicological assessment. There is still – or again – a need for more (small), reliable standardized sediment contact assays (Cairns et al. 1992) to test the ecotoxicological effect of SPM which is potentially transported along the whole catchment area. There is, however, also a need for more reliable determination of particle bound contaminant load and – most of all – the need for attention towards this challenge from environmental decision makers.

But this still leaves the question of the objectives of river basin management. In the Elbe study we used different sediment guidelines that are protective of ecosystem services to indicate risk. However, the overall goal is a “healthy” environment. We have many tools to detect changes and shifts in communities, e.g. in their functional and structural diversity, in the adaptation of organisms, in their activity. And while we can say something about the quality of a specific site, what are the implications for the river ecosystem? When does a watershed stop being ‘healthy’? We can possibly differentiate a very good and a very bad status, but is it not the transitional state which we need to notice in order to initiate actions?

Many terms have been provided, which may together form a key to ‘early warning indicators’ (stability, resilience, ecosystem functioning, vulnerability) but the interpretation on river basin scale remains a challenge (Apitz 2006, Frederick & Cash 1996, Gunderson et al. 2002, Holling 2001, Peterson et al. 1998, Wall 2004).

With a background in marine science, it was originally the physiological effect of contaminated sediments on organisms which interested me most. Impact assessment based on ecotoxicological, environmental and chemical data, carried out addressing uncertainties and variability with the help of fuzzy logic are still one of my major fields of interest. My focus, though, has recently shifted towards river basin scale, as becomes apparent from the river basin scale studies on Rhine and Elbe and the SedNet activities in Europe that I have been involved in.
Challenges for river basin management

P. Gert-Jan de Maagd

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Background

As input for the RISKBASE/Modelkey - conference “Risk assessment in European Rivers Basins – State of the art and future challenges” (12-14 November 2007, Leipzig, Germany) the author was invited by the organisation to contribute on the subject of challenges in river basin management. Although the concept of integrated river basin management (IRBM) exists much longer, it was the European Water Framework Directive (WFD) [1] that made IRBM the paradigm of water management throughout Europe. Therefore, the discussion in this paper will focus on the WFD from the perspective of the river basin manager. In the recommendation and perspectives section special attention this will be broadened to IRBM in general.

Four steps1 in the selection of WFD measures are discerned in order to identify challenges for IRBM:

1. Getting a clear insight in the current chemical and ecological status of the waterbody/(sub)river basin;
2. Getting a clear insight in the main pressures that should be dealt with in order to reach good chemical and ecological status in waterbody/(sub)river basin;
3. Getting a clear insight in measure-effect relationships of possible measures to reach good chemical and ecological status in waterbody/(sub)river basin;
4. Decide whether synergy can be achieved by combining WFD measures with measures and goals of other policy themes (climate change, flooding, nature conservation, historical cultural values of the landscape).

These four steps cover a much broader field than merely risk assessment, which is limited to step 2. Nevertheless, the applicability of risk assessment increases for the river basin manager if it is connected to the acting perspective (step 3 & 4) of the river basin manager.

Thesis 1: Challenges for improving the insight in the chemical and ecological status include (1) integration of surface water monitoring, groundwater monitoring and monitoring of protected areas; (2) combining modelling and monitoring of biological quality elements

In 2005 EU Member States reported the so-called article 5 characterisation reports for the European River Basin to the European Commission. These reports include a preliminary risk assessment on which water bodies are at risk of not reaching the good chemical and ecological status in 2015. This assessment was partly based on the data of existing monitoring networks that were not specifically designed for the WFD. At the beginning of this year the EU Member States reported the WFD monitoring programmes that will yield a more complete insight in the current chemical and ecological status of the waterbody/(sub)river basin enabling river basin managers to focus their programme of measures that will form part of river basin management plans that should be finished by the end of 2009. From the contributions to the WFD monitoring conference in Lille, France, [2] it became clear that although the WFD monitoring programmes will certainly lead to a better picture of the chemical and ecological status of waterbodies/(sub) river basins a number of challenges will remain to exist. In many European countries there is a tradition of designing groundwater monitoring programmes and surface water monitoring programmes independent from each other. The WFD however requires that monitoring clarifies the interdependency of groundwater and surface water status. Therefore, groundwater and surface water specialists should work together on integrated groundwater and surface water monitoring programmes. In addition, further improvement is needed on the integration of groundwater and surface water monitoring and monitoring of protected areas, more specific habitat

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1 The reality of WFD implementation is more complex and includes additional steps like deriving and setting goals, status classification, public participation, cost-effectivity analysis and more.
and species protected areas. The main challenge lies in how to improve confidence and precision of the monitoring of biological quality elements. Although it is correctly stated by the EU-commission that the investment in monitoring can be extremely cost-effective as it can help taking well-informed decisions in the programme of measures [3], using monitoring, modelling, expert judgement and knowledge rules together applied in a weight-of-evidence based approach may be more cost-effective in determining the ecological status of water bodies.

**Thesis 2: The challenge is no longer in how to measure bioavailability but rather in how to apply the bioavailability concept within policy and river basin management**

Based on the chemical analytical data from the WFD monitoring programmes river basin managers have to determine the chemical status and part of the ecological status of the waterbody / (sub)river basin. To do so it is of importance to decide whether bioavailability should be taken into account in compliance checking against environmental quality standards. The proposed priority substances directive [4] allows the Member States to do so but for metals only. So far the scientific literature is focussed on how to measure bioavailability of both metals and organic substances. Numerous papers have been published on for instance measuring freely dissolved concentrations, free ion activity and fugacity. Recently for metals biotic ligand models are developed for different metal–species combinations. How to apply the bioavailability concept within river basins, however, is less straightforward. In the opinion of the author a number of issues should be addressed a.o.: (1) how to combine the bioavailability concept with the pollution prevention principle? (2) should the bioavailability concept only be applied in a curative environmental management (for instance, in prioritising sites of contaminated sediments for remediation) or should it be reflected in emission limits as well?; (3) to what extent does bioavailability change in time and position when a substance is transported down stream in a river basin?

**Thesis 3: More research effort is needed on ways to identify dominant pressures that hinder the achievement of good ecological status**

Once the chemical and ecological status of the waterbody / (sub)river basin are known and failing the objectives, the next step (step 2) is to identify the main pressures that are responsible for failing the objectives (DPSIR approach [5]). For the chemical status it is relatively easy. A failing chemical status is by definition reflected in the presence of priority substances in concentrations above the environmental quality standard. The river basin manager has to determine the main sources and select measures (step 3). For the ecological status, however, it is much more complicated. If the monitoring results of biological quality elements show that the ecological status is insufficient, it is often not straightforward what the main pressures are. This makes it often impossible for river basin managers to design a tailor-made programme of measures. Therefore, in the Netherlands recent work is done on applying WFD investigative monitoring in effort to determine the relative pressure of hydromorphological changes, pollution, eutrophication and other pressures [6]. Another promising approach is the application of diagnostic effect models as is done within the Modelkey-project. In this approach the relevance of pressures are presented in intelligible pie charts [7]. Nevertheless more discussion and research is needed in this field especially on the subject of calibrating the relative contribution of pressures to the ecological status.

**Thesis 4: Research should be focussed on improving the insight in measure -effect relationships**

After the determination of the dominant pressures the river basin manager has to select measures. An analysis of measures will contribute as an input to how the goals in the WFD can be cost-effectively implemented in river basin management plans [8]. In order to do so a quantitative insight is needed in measure-effect relationships. For emission reduction in point sources this is relatively easy, for diffuse sources it becomes a little harder, for measures to mitigate hydromorphological alterations it becomes highly complicated. Therefore, research focus should be on improving the knowledge on quantitative measure-effect relationships for hydromorphology.
Thesis 5: Integration of policies is essential for successful river basin management

Selection of measures should not only be based on insight in measure – effects relationships. River basin managers should check whether synergy could be achieved by combining WFD measures with measures and goals of other policy themes (i.e. climate change, flooding, nature conservation, recreation, historical cultural values of the landscape). An example is given in a recent paper by Wharton & Gilvaer on how to improve the ecological status whilst, at the same time, providing more sustainable flood management [9]. In general, more emphasis on the integration of policies is warranted for various reasons. First, money is to be saved by combining measures and goals. Second, the expected impact of climate change on flooding and the focus on the impact of hydromorphology for reaching the ecological goals makes water management more and more part of spatial planning for land is needed to reach the goals of the river basin manager. The river basin manager has to find new partners to combine the water objectives with that of others. Third, Member States have to implement various European (Environmental) Directives in an integrated way. An example is the implementation of the WFD besides the Birds and Habitat Directives. To meet this challenges national authorities may reinvent their role as facilitators of the integration of policies by regional authorities on subbasin- or waterbody level. Nowadays, the majority of environmental legislation is European shifting the role of national authorities from lawmakers to law implementators.

Recommendations & perspectives

In addition, to the research recommendations in the discussed theses above, it is recommended that the RISKBASE partners discuss the following paradox or contradiction: Although throughout Europe hydromorphological pressures and eutrophication are reported in the art 5 reports as being the main challenges for achieving good ecological status [9], the scientific community predominantly focuses research interests on fate and effects of micro pollutants.

When broadening the perspective from risk assessment for river basins to river basin management in general, other and possibly more pressing challenges may pop up. The World Wildlife Fund defined the following key elements for successful IRBM [10]: (1) a long-term vision for the river basin, agreed to by all the major stakeholders. (2) Integration of policies, decisions and costs across sectoral interests such as industry, agriculture, urban development, navigation, fisheries management and conservation, including through poverty reduction strategies. (3) Strategic decision-making at the river basin scale, which guides actions at sub-basin or local levels. (4) Effective timing, taking advantage of opportunities as they arise while working within a strategic framework. (5) Active participation by all relevant stakeholders in well-informed and transparent planning and decision-making. (6) Adequate investment by governments, the private sector, and civil society organisations in capacity for river basin planning and participation processes. (7) A solid foundation of knowledge of the river basin and the natural and socio-economic forces that influence it.

The author expects that the WFD, as such, is a great help in addressing most of these key elements (1,3,4,5,6) while EU funded research and the Common Implementation Strategy help to address key element 7. That leaves integration of policies the main challenge for the common years.

References


2 To be decided whether it is a paradox or a contradiction.
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Controlling pollution in river basins - Risk based approaches at different scales

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Background

The development of river basin management can be thought to have progressed through 3 broad phases [1]:

- Sanitisation provision phase 1850s to 1950s – where the emphasis was on clean water supplies and safe sewage disposal;
- Pollution control stage 1950s to 1990s – where the emphasis was on water quality improvement through the control of polluting discharges (point source pollution); and we are currently in the:
- Sustainable development phase – where the realisation that an holistic approach to environmental issues is necessary to meet sustainable development aspirations and address what is now the most significant challenge – diffuse pollution.


The WFD is the primary legislation that addresses the shift in the scale of environmental management with its River Basin Planning process requiring a broader and more integrated approach to environmental management. It has as its objectives the achievement of cleaner waters and healthier aquatic ecosystems, which are essentially surrogate measures for healthy river catchment/basins, and the environment itself. Such an “ecosystem-centred” approach requires environmental managers to develop a more holistic understanding of the land/water system and how aquatic life is influenced. As complete a systemic understanding as possible will be necessary before management solutions are adopted. The WFD will therefore be an important strategic driver not just for water but also for land management and land use change across Europe.

Thesis 2: A risk-based framework and associated tools are needed to allow effective management solutions to be identified and developed.

The WFD is the first risk-orientated environmental directive and invites risk-based approaches to be developed and used in its implementation. It uses the terminology “pressures and impacts” to focus on the key issues that are inhibiting the achievement of the quality measures and in turn focus river basin managers on those “solutions” that will have the most influence/benefit. Different pressures (pollution, abstraction, channel morphology, climate change, regional/global economics etc) affect different parts of the aquatic environment in different ways. If effective management solutions are to be developed in such a complex system, then a risk-based framework must be developed along with the appropriate tools which will enable it to be applied consistently.

A risk-based approach for land and water management is required and requires clarity about the receptors to be protected and the critical pressures acting upon them. Secondly, a conceptual understanding of how any particular river catchment/basin works and the ways in which pressures are promulgated through it at this scale is needed.
Thesis 3: The S-P-R paradigm, developed from the contaminated land technical area, provides a model that can be adapted and adopted for use at the river basin scale. The approach should be aimed at the 2nd and 3rd rounds of River Basin Plans.

The source–pathway–receptor paradigm, developed from the management of contaminated land at a site-specific scale, can be adapted and adopted to provide the framework needed at the larger scale. It could help deliver strategies for key pollutant linkages of more diffuse sources to be identified at the catchment scale, which present perhaps the greatest threat to the non-achievement of WFD goals. Management options (removing or minimising the source, intercepting the pathway or protecting/removing the receptor) can then be developed and the costs and benefits assessed (this is an important part of the Programme of Measures (PoMs) within each River Basin Plan). From such an analysis “win-wins” can also be determined which may deliver other benefits – e.g. changing from arable farmland to woodland to address nutrient leaching also reduces soil erosion, enhances biodiversity, leisure amenity, increases house prices etc.; focusing on one will benefit the other. The same approach can be used for any pressure, chemical or physical.

Competent authorities in Member States are currently developing PoMs for the 1st round of River Basin Planning, but there is little expectation of significant improvements in ecosystem quality by the set date of 2015. The 2nd and 3rd rounds of River Basin Plans will be where more radical (integrated and risk-based) approaches will be adopted. This fits with the timescale for research to impact on policy.

Thesis 4: The risk-based approach could be developed further from dealing with existing pressures to anticipating future ones and protecting the aquatic environment sustainably.

The above framework will be useful in a reactive way for dealing with historical and existing problems, but future risk-based land management approaches should be proactive. A goal of a new Risk-Based Land Management (RBLM) approach should be to ensure that land use planning incorporates environmental issues through a land zoning approach. This exists to some extent already - e.g. groundwater protection zones in public supply catchments, developed in some countries to include the whole of the groundwater resource. We should develop this approach to match land use to the vulnerability of the soil/geology/water/ecosystem. GIS is a powerful tool that allows us to develop the necessary visual interface (for engaging with stakeholder communities) in conjunction with integrated catchment scale models.

Thesis 5: We will need to train a new breed of catchment managers who can think in a different way.

If we are to address the huge uncertainties in understanding and develop options to combat both the historical legacy and the future adaption to climate change, suites of risk-based tools, and the people who can use them, will be both needed. A new breed of “catchment manager” will be required to manage those pressures that affect the health of our aquatic ecosystems the most. These people will need a more broad but integrated understanding of our environment at a scale at which pressures have not previously been managed. At present few such people exist and they will need to be trained.

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Bob Harris worked for the Environment Agency of England and Wales and its predecessors for 35 years largely in the area of groundwater and contaminated land. Latterly he was Head of Ecosystems Science also managing the Integrated Catchment Science Programme. This programme provided the scientific underpinning for policies and regulation, relating to the implementation of the WFD. Bob took early retirement in 2007 and is now part-time Strategy Director of the Catchment Science Centre at the University of Sheffield (www.sheffield.ac.uk/csc). He remains involved in a number of European initiatives including RISKBASE.
Resilience: A useful concept for risk-based river-basin management?

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Background

The concept Resilience is defined as: “the amount of change a system can undergo (its capacity to absorb disturbance) and remain within the same regime – essentially retaining the same function, structure and feedbacks” (Walker & Salt, 2006). The concept can be applied to various fields such as ecology, sociology and economics. Especially in ecology and economics, the concept has received a lot of attention recently, e.g. with regard to ecosystem change. Starting from this area, the last years has seen a lot of active thinking in this field, involving reflection on theory, application to various ecosystems and application to other systems (e.g. society) and combination of these various aspects (for examples, see www.resalliance.org). The concept is closely linked to similar concepts such as sustainability and biodiversity.

Within recent developments in river management the concept seems to be potentially very useful. On the other hand, within the scientific community the concept is partly new and unknown, and often met with sceptis: what can we practically do with this abstract concept? Here we want to assess what can be the use of the concept in risk-based river-basin management. We will pay attention both to the scientific side as well as to the practical application in river basin management.

Thesis 1: Resilience is a useful concept for risk-based river-basin management

Our main thesis is a confirmative answer to the question posed in the title. We will support this in the following subsections.

Thesis 1a: Resilience offers a concrete direction for a system-oriented approach of river basin management

It has become clear that river basins have to be approached as a whole: all parts of it are interrelated. This, however, makes things very complex: instead of focusing on a single sub-issue, one has to pay attention to many different processes of different nature to understand phenomena. To regain focus, one needs a direction, and resilience offers a very powerful one, especially in risk-based water management. Resilience collects various aspects of the system, which all are related to risks posed by stress imposed in the system.

An example of the way Resilience offers a unifying direction, can be seen in the European research project AquaTerra. In this, numerous aspects of the fate and effects of pollutants in European river basins were investigated. In fact, the research done is so diverse, that it might be hard to find a red thread. Resilience however, unifies most of the research carried out, and at the same time offers a provocative open end for further research questions following from it. Interestingly, the system-oriented approach of Resilience links very well to the move in the same direction with the adoption of the Water Framework Directive.

Thesis 1b: Resilience offers the potential of flexibility instead of rigidity in dealing with river basins

It was already stated before that river basins must be regarded as complex systems. Regulations however are more simplistic: think of the use of simple, general norms for chemicals in water bodies. Although it has become clear to many people that such a simplistic approach cannot be supported by scientific knowledge, it is difficult to find a good framework, from which we can subtract how to deal with the complexity, and at the same time come to clear regulations. Resilience can offer a framework for this. For instance, for lakes the combined effects of eutrophication, fish and water depth on the turbidity of the water system can be analysed well using resilience (Scheffer, 2001). This example
clearly demonstrates the relevance for water management and offers a concrete, flexible approach to deal with the system complexity.

**Thesis 1c: Adaptive management offers a way to continuously cope with changes**
A concept closely related to Resilience is Adaptive management. Instead of fixing an end-goal and the management strategy to reach it, Adaptive management implies a continuous re-evaluation of the strategy based on the results achieved. This offers a valuable approach both to deal with complexity and the limited predictability of the system, as well as to deal with limited knowledge about the system.

**Thesis 1d: The concept Resilience offers a way to link the biophysical as well as the societal system**
The water management practice is a whole of both natural and societal aspects. For proper management, these should be linked. Resilience offers a way to do this. Many loops and patterns involved in resilience transgress the borders of the natural and societal system. Examples of this imply the way communities are related to the natural system, for instance for their use of the system for food or other goods. Another example: very rigid regulations in water management can be regarded as a system with low resilience.

**Recommendations & perspectives**
The concept Resilience has some very attractive sides for application in water management practice. However, the exact role for science and water management has be made more clear, because there are still a lot of misconceptions and unclarities. Some recommendations:

1. In discussion with water managers, the best application in practice should be found out. This implies that the theoretical framework should be translated to concrete application: what practical endpoints do we have to assess resilience, how can resilience based management be given a practical form, etc.
2. For natural scientists, resilience can be an abstract, vague concept. The meaningfulness of it has to be demonstrated more clearly. Firstly, the concept should be validated with clear examples from science (which we try to do in this presentation). Secondly, the way that work which is already done fits into the Resilience framework and the way it can, as such, link to questions from practical management side, should be demonstrated more clearly.

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Managing European sediments: Can we expand our ecological risk assessment paradigms?

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Background

Whether it is to inform a contaminated sediment site assessment, a dredged material disposal application, or the management of floodplains to accommodate sea level rise, there is a need for scientists to consider the relationship between sectoral tasks and larger basin-scale sustainable development and ecosystem management goals. Depending upon the processes being addressed, different spatial and temporal scales must be understood. There is a need to explicitly link the science, management and policy of river basin management. To this end, we should be aware that we are not carrying out, for instance, sediment management, but the sediment-specific aspects of ecosystem management. We need to develop regional conceptual frameworks that identify, and provisionally quantify, these processes, to inform the prioritization of research, assessment, management and regulatory efforts. Even if all issues cannot be addressed in every task, research, assessment and management must be couched in larger conceptual models that address the interactions between benthic and pelagic ecosystems, as well as the processes that might be affecting them (and the services they provide) in the larger catchment (Apitz 2006a,b,c; Apitz et al., 2006a).

Thesis 1: Benthic community health and ecosystem services

The ecosystem approach which uses a scientific understanding of how various levels of biological organization and function interact with the environment, both natural and human-impacted is at the core of current and emerging European environmental policy and legislation. Expressing benthic ecological processes and resources in terms of the goods and services they provide (e.g. gas and climate regulation, including carbon cycling and storage; detoxification and purification of waste; and the storage, cycling and maintenance of nutrients) links our scientific understanding of the aquatic environment to socioeconomic factors which generally drive policy decisions (CBD COP 1998). Thus, the successful application of policy and management interventions (including the Marine Strategy, Habitats, Nitrates and Water Framework Directives) may succeed in reducing ecosystem degradation and enhancing the contributions of ecosystems to human wellbeing, but knowing when and how to intervene requires substantial understanding of both the ecological and the social systems involved. Ensuring productive watersheds that continue to provide the goods that society has come to depend upon requires maintenance of the sustaining services provided by healthy and functioning benthic communities, but these links are often ignored and are poorly understood. Developing and linking the underlying scientific understanding and effective river basin and coastal management strategies will be a significant challenge, but will be required if we are to manage river basins in support of these ecosystem services (COBO 2007; Apitz 2006a).

Thesis 2: Ecosystem status should be based upon function as well as structure

Most current indicators of benthic ecosystem health or status are based upon community structure which is highly dynamic and sensitive to a variety of natural and anthropogenic factors, including irreversible ones such as climate change and invasive species. A structure-focused criterion evaluates ecosystems in terms of “climax communities”; a disturbance can cause a shift from one stable state (e.g., seagrass to eutrophic) to another, and, possibly, return if the pressure is removed. Evaluating benthic “health” based upon differences in community structure provides little insight into function or causality, and defining recovery as a return to historical community structures may be doomed to failure. Ecosystem status is more meaningfully expressed as a set of multidimensional parameters of structure and function, with an understanding that changes in this complex system may move in various directions, to countless alternative states. Thus, an important measure of benthic community health should be an evaluation of selected aspects of community function, and their relationship to ecosystem services (Apitz 2006b; COBO 2007). Figure 1 illustrates a conceptual model developed to
frame these measures. For a given structural or functional parameter, resistance can be defined as the amount of a given pressure that can be applied without deterioration in status (as defined by a specific measure). As a pressure is removed, there is a lag in recovery, but given time, status may recover, though it may not return to original levels (Apitz 2006a; Elliott et al., 2007).

![Diagram](image)

Fig. 1: This conceptual framework recognizes that ecosystem “health” or status cannot be represented as a univariate index, but rather is represented by a complex set of structural and functional measures. Recovery of these characteristics after a perturbation may be partial or total, and it is a complex set of these parameters that affect the delivery of ecosystem services (Apitz 2006a; Elliott et al., 2007).

**Thesis 3: A need for studies to understand how ecosystem functions and services respond to single and mixed pressures**

An understanding of the interplay between various ecosystem parameters to an overall definition and management of ecosystem “health”, status, function, and services is required. Then, if restoration, remediation or recovery do not result in a return to reference conditions, communities can be evaluated over space or time in terms of their functional characteristics, and their ability to provide valued ecosystem services. Thus, it is important to evaluate how the functioning of communities already adapted to and/or impaired by various perturbations responds to further pressures. The Coastal Ocean Benthic Observatory (COBO; www.cobo.org.uk) programme was designed to study coastal benthic ecosystem health and function and their response to various perturbations using integrated, modular, in situ systems. There is a need for many such field studies examining the functional response of benthic and other communities to single and mixed pressures in various environments at multiple spatial scales and trophic levels. These studies will help reduce the uncertainty in predictions of how ecosystems will respond to multiple pressures as well as to various management scenarios (Apitz 2006a; Apitz et al., 2006b, 2007a)

**Thesis 4: Holistic, basin-scale management requires a flexible approach to complex risks**

It is clear that simple views of risk assessment cannot alone be used to inform the complex decisions society will have to make to holistically manage risk at the basin scale (Apitz et al 2007b). The task of managing risk at the river basin scale involves the identification, ranking, and management of many disparate (but often interacting) types of risks, hazards and vulnerabilities, from the assessment of risk of contaminated sediment to the prevention of flooding due to climate change. For relatively simple problems, models to define risk based upon severity (hazard) and probability (exposure) can be used. However, for more complex systemic risks, and for other categories of risk, a number of other tools must be used, including hazard analysis and identification of critical control points for safety;
catastrophe models and community vulnerability assessments for natural disasters; failure analysis and preliminary hazard assessment for complex systems; and system risk and vulnerability analysis, terrorism risk analysis and information security risk assessment for security issues. There is a need for clarity on what sort of risks are being managed, what sorts of decisions are being informed, and how. For various types of hazards or concerns, scenarios should be developed, and vulnerabilities and risks must be identified, characterized and ranked. This should be done with a clear understanding of how various natural and anthropogenic processes affect the objective of concern, and an evaluation of the importance and controllability of these processes (e.g., Figure 2). Once this has been done, for a potential risk, decisions should be based upon scenario probability, preventability, causality (human-caused or natural), time scale (gradual or sudden), and potential costs and risks. Prevention strategies and preparedness and response strategies (whether a scenario is unpreventable or if prevention fails) must be developed. Depending on the probability, vulnerabilities and costs of a given risk, emphases will be put on different measures (Apitz 2007; Figure 3).

Fig. 2: DPSIR-style framework which addresses the natural and anthropogenic processes causing sediment quantity inputs and outputs, their characteristics, impacts and potential response actions within the context of regional management objectives (from White et al., 2006).

Fig. 3: Action flowchart illustrating the various decisions or actions that must be considered to address a given risk. Clearly, whilst scientific input is necessary to develop management strategies for all these measures, how they are selected and prioritised is a policy decision, and thus there is a need for clear communication between scientists, stakeholders and decision makers in all aspects of the risk management process (Apitz 2007).

Recommendations & perspectives
We need to clearly define the role of sediments and their associated benthic ecosystems in sustainable river basin management. Even if all issues cannot be addressed in every task, research, assessment and management must be couched in larger conceptual models that address the interactions between surface and subsurface ecosystems, as well as the processes that might be affecting them (and the services they provide) in the larger catchment. In future, management in support of ecosystem health and sustainability may dominate sectoral management. This will result in the evaluation of complex, systemic risk issues at many scales. Whilst science can advise these decisions, how it will be carried
out is the choice of policy. Scientists must learn to communicate in terms of informing risk decisions. Complex risk decision making must have the continuous involvement of all parties, and should be an iterative and deliberative process. Although management decisions must be made, even when information is imperfect, adaptive management and thinking allows us to learn as we go. The challenges described above are not trivial, but otherwise there is a risk that we will continue to spend huge amounts of time and money without making any difference to overall ecosystem health or sustainability.

References


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The MODELKEY database - Towards an integrative European risk assessment

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Background

Freshwater is one of the most valuable resources on earth and its protection and conservation for future generations pose a major challenge for our society. Therefore, the European Commission enacted the Water Framework Directive (WFD) to manage the European River Basins (CEC 2000) in a sustainable way. This Directive implies, for the first time, that the protection of aquatic ecosystems as a whole is required from all Member States and across boarders. However, as aquatic ecosystems are subject to various pressures (Furse et al. 2006), this challenging task demands European water managers for an integrated assessment of all possibly available data to identify the responsible stressors and to select management options that will be most effective to improve water quality (von der Ohe et al. 2007). Moreover, this requires appropriate indicators that link the observed effects to certain anthropogenic stressors (Furse et al. 2006). As river basin catchments often cover several member states, their management demands for an integrative assessment approach to allow for a similar protection of the water resources. Since elevated levels of numerous chemicals are frequently detected in European surface waters, toxic stress from environmental pollutants may be one of the driving forces for an insufficient ecological status and reduced biodiversity of freshwater and marine ecosystems (Brack et al. 2005). Therefore, the Integrated Project MODELKEY aims to assist water managers with a generic risk assessment tool for European river basins. To enable the application of the new assessment tools, the existing monitoring data from three case study river basins were collated and standardized before implementing them into the MODELKEY database in order to allow for an integrative assessment of the available data within and among all basins.

Thesis 1: For an effective river basin management, an integrated ecological and chemical status monitoring is needed

Comprehensive monitoring programmes have been established to assess the actual status and to achieve and maintain at least a good ecological and chemical status of the European river basins. However, in many countries, ecological and chemical monitoring programmes developed independently and hence, have been implemented by different federal institutions. As the WFD does not require a comparative assessment of ecological and chemical status of a water body, the sampling spots of the former monitoring programmes were often maintained, as well as the different institutions. Therefore, sampling sites may be still located at different spots and the respective sampling campaigns, implementing the WFD, are often not synchronised. Hence, the levied data is often timely and spatially scattered which hampers an integrated assessment of those monitoring data and to establish potential cause-effect relationships, as shown for the characteristics of the macroinvertebrates community structure and pesticide exposure (Liess & von der Ohe 2005, Schäfer et al. 2007). Moreover, habitat characteristics, which would be useful to explain natural variability of the aquatic community structure commonly lack in such databases. However, especially the joint consideration of all information is crucial to eliminate confounding factors, like the co-occurrence of other natural stressors (e.g. floods or droughts), which mask the adverse effects of a particular stressor (Schäfer et al. 2007). Hence, it seems mandatory to combine the ecological and chemical monitoring programmes to identify responsible stressors for the deviation from good ecological status and to choose the appropriate management options.
Thesis 2: A meaningful chemical status should be based on comparable analytical measurements

The chemical status of a water body is usually based on measured concentrations of currently 33 priority pollutants (PP) and exceedance or not exceedance of respective threshold values (CEC 2000), referred to as environmental quality standards (EQS). However, the analytical methods used to determine chemical concentrations differ widely among member states. The measurement of chemicals in whole water samples, as for example in the Belgium part of the Scheldt river basin, are hampered by the fact that the more hydrophobic compounds are most likely adsorbed to the suspended particle fraction and may not be bioavailable. Hence, a respective exceedance of EQS may not necessarily indicate environmental harm. However, this method could be regarded as worst case scenario. Chemical measurements in filtered samples again, as for example in the Llobregat river basin, seem to be more realistic and could be supported by valuable information on hydrophobic toxicants, measured in the filtrate. The respective dissolved concentrations could be then estimated from its partitioning into the water phase. However, precise analytical measurements of environmental pollutant concentrations in differently taken and processed environmental samples are not useful for an integrative risk assessment on the river basin scale. Here, analytical measurements should be consistent to allow for the assessment of river basin-specific cause-effect relationships between ecological and chemical status. Moreover, the commonly applied spot samples are likely to miss the maximum concentrations of pollutants that are suspected to have the biggest effects on the community structure (Liess & Von der Ohe 2005). An alternative to water spot samples might be the use of event driven passive sampling. This would also allow for a cost-effective and more comprehensive picture of the potentially bioavailable exposure situation for aquatic organism.

Thesis 3: An integrative European Risk Assessment of environmental pollutants demands for a common nomenclature of chemicals

For the assignment of the chemical status of a water body, the concentrations of merely 33 PP and the respectively available EQS are considered. However, already now, the differing nomenclature of environmental pollutants hampers an integrative assessment among all member states of a river basin. The list of PP, however, was never intended to be fixed and will be eventually expanded to new compounds (CEC 2000). These new compounds are commonly referred to as emerging pollutants and will quickly extend the list - with river basin specific differences. Moreover, other routinely measured chemicals that assist the assignment of the ecological status also differ between river basins and even between river authorities of different Member states of the same river basin. For example, the catchment of the river Danube covers 17 member states that are all involved in the management of the river basin. The differing nomenclature in several member states, however, hampers the establishment of computer-based integrative assessment tools, such as GIS based transport models or Toxic Unit indicators. Even the commonly used names have often synonyms or country specific forms (e.g. benzene – Benzen (GER), Benzeen (BL) and benzè (ES) - to give a simple example). From a database point of view, these chemicals could not easily be identified as the same compound and would require elaborate translation steps before the actual assessment. Hence, simply the use of CAS numbers instead of country specific chemical names would be far more useful.

Thesis 4: Biological Quality Elements (BQE) specific toxicity indicators are needed to identify responsible stressors

Although mixtures of several pollutants are frequently detected in environmental samples, so far, EQS are only available for a few compounds. For example, the list of chemicals in the MODELKEY database that are currently measured to implement the WFD in the three river basins Elbe, Scheldt and Llobregat comprehend more than 500 compounds. However, for more than half of these compounds, no toxicity test results are available (EPA 2002) - for none of the four commonly used BQE (benthic diatoms, higher plants, macroinvertebrates and fish). Hence, the very precise measurement of a certain chemical may be meaningless without a respective effect measurement of that compound. However, QSAR-based effect predictions might be a promising tool to derive likely effect thresholds for those compounds (Von der Ohe et al. 2005). For the purpose of stressor identification, the thresholds should be furthermore BQE-specific, as the effects of different pollutants, such as insecticides or herbicides,
might be very different for single BQE. Hence, as the ecological status is based on the most affected BQE (CEC 2000), BQE-specific toxic units might be more indicative for a certain pollutant that might be responsible for the deviation of the good ecological status than simply the exceedance of one EQS. To derive respective toxic units, the measured concentrations of all compounds are scaled to inherent toxicity (acute LC50s) and added to an overall toxicity measure, based on the toxic unit approach of Peterson (1994). Hence, considering the toxicity of a mixture of several compounds that are all measured below their EQS may pose a severe environmental harm.

**Thesis 5: The biological indicators that are used to assess ecological status should be stressor specific**

With the establishment of the good ecological status, which has to be achieved until 2015, the WFD aims for the protection and preservation of whole aquatic ecosystems. Their management demands for an integrated assessment of many factors, as river basins are usually subject to several stressors (Furse et al. 2006) whose effects may be also altered by climate change. However, to detect deviations from the good ecological status and furthermore to identify the responsible stressors, specific indicators are needed (Furse et al. 2006, Von der Ohe et al. 2007) that often still lack. Moreover, only the integrated analysis of stressor-specific indicators will allow for the selection of effective and cost-efficient measures to finally achieve a good ecological status that could also be maintained. For the assessment of the actual status, the WFD requires at least four different BQE, and the implementation of combined monitoring programmes comprises the chance to assess the joint information that originates from the different trophic levels that are covered. In this context, stream-dwelling invertebrates have become the most commonly used organism group for the application of biological indicators, as the characteristic of their community structure represent a kind of “biological memory” for prevailing environmental conditions. For this group, quality metrics for several stressors already exist (see Figure 1) that allow for the establishment of river basin specific cause-effect relationships with environmental factors (e.g. toxicity of toxic pollutants) and to derive respective threshold values. The simultaneous observation of the exceedance of these thresholds in accordance to the respective value of quality metrics indicates potential effects of this factor. With respect to stress from environmental pollutants, the additional confirmation with bioassays may help to verify the bioavailability of the analysed toxicants as suggested by the TRIAD approach (Chapman 1990). Only the integrated information provides three lines of evidence to assess the ecosystem quality due to chemical contamination and may rule out other confounding factors. This concept could also be extrapolated to other stressors, such as organic pollution.

![Fig. 1: Example of the relation between the maximum Toxic Units (TU) for invertebrates and the percentage of invertebrate species at risk (SPEAR). Sites are differentiated on the presence of recovery sections upstream of the study sites (filled circles, linear regression, \( p \leq 0.01 \)) or absence of such sites (open circles; linear regression, \( p \leq 0.01 \)) at 34 monitoring sites in the river Llobregat. Confidence bands show the 95% confidence limit of the respective means.](image-url)
Recommendations & perspectives

For the responsible management of river basins, both an integrated and integrative risk assessment of all potential driving forces is needed in order to identify responsible stressors. To achieve the required prerequisites for this, only few adaptations to the present implementation procedure would be necessary. First of all, chemical and biological monitoring sites should be combined in order to allow for river basin specific cause-effect relationships between chemical and ecological status. Furthermore, the nomenclature of the analysed chemicals should be standardized (e.g. CAS numbers) to facilitate the application of European wide toxicity indicators. These again should be BQE-specific in order to use the information of trophic level specific differences in sensitivity to retrospectively identify potential compounds that are responsible for the deviation from the good ecological status. In contrast, the conventionally used EQS values allow for a prospective and conservative protection of the aquatic environment. For a basin wide risk assessment, the underlying measured chemical concentrations should be comparable for all sampling sites and comprise both worst case estimates from whole water samples as well as realistic bioavailable fractions to establish site specific cause-effect relationships. Besides new tools for the indication of toxic stress, specific biological and physico-chemical indicators for many other stressors are still needed. Respective tools that yield similar results in different eco-regions and therefore could be applied across Europe would be favourable. In general, a common way to process samples and store the levied data would be crucial for river basin wide and integrated risk assessment tools. This would allow to assign an ecological status in a more comprehensive way and to unravel potential confounding factors. Only then, effective management measures could be found that enhance the ecological status in a sustainable and cost-efficient way.

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Peter C. von der Ohe (www.ufz.de/index.php?en=14838) works at the UFZ as database manager of the MODELKEY project and in the WP4 coordination of RISKBASE. He developed the SPEcies At Risk (SPEAR) index to assess the impacts of pesticides on stream-dwelling invertebrate assemblages and investigated the positive effects of recovery potential for a better understanding of ecosystem functioning. For the assessment of risk from organic pollutants, he developed structural alerts to distinguish baseline from excess toxicity in macroinvertebrates that could be used to derive respective Toxic Units.
New diagnostic and predictive modelling tools for an advanced and integrated evaluation of chemical, ecological, and ecotoxicological monitoring data

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Background

The protection of aquatic communities has historically been evaluated via two main methods: 1) benchmarking against water quality criteria and 2) biomonitoring to determine ecological status. The first method assumes that the primary causative agent for adverse effects to aquatic communities is chemical presence and concentration. The second method for assessing water quality does not assume a particular cause for adverse effects, but measures the status. Comparisons of the status relative to acceptable ecological reference values provide measures of local impact. Unfortunately, linkages to factors that can be considered possible causes of impact often lack. The major obstacle appears to be lack of a theoretical framework to address a complex data set at all, but practically also the low statistical power when such a framework would be available. Where linkages have been made, they are often limited to qualitative weight-of-evidence methods that are often value-laden and contingent upon the levels of expertise and knowledge. The end results of these pragmatic evaluation methods are often inconsistent, resulting in unclear communication on the role of stressor-factors, and disparate management proposals for aquatic resources.

Hence, what is needed is a quantitative methodology that will enable appropriate assignment of causality relative to measured and/or predicted biological impacts, and that subsequently will assist a more consistent interpretation of causes if impacts, and more consistent management schemes. This should lead to better protection of aquatic communities, and to more effective programs of measures in case of impacts.

Aims

The aim of this contribution is to introduce and describe a set of approaches that link both evaluation methods in a novel framework. This is done according to a set of theses, working hypotheses for which specific data and analyses have been done recently.

Thesis 1: Exceedence of quality criteria is a clear observation in a legal sense, but highly unclear to predict local impacts

The MODELKEY BASIN database contains historical data on the physico-chemistry as well as the biota for multiple sites and years in three case study rivers: Elbe (Germany and Czech Republic), Llobregat (Spain) and Scheldt (Belgium). The present analysis is restricted to the Scheldt River in the period between 2000 and 2004, because the selected dataset is containing the most elaborate representation of toxicant data (Table 1).
Table 1: Tabular description of the Scheldt physico-chemical dataset (2000 – 2004).

<table>
<thead>
<tr>
<th>Numbers</th>
<th>Water samples</th>
<th>Sediment samples</th>
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</thead>
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<tr>
<td># Toxicant records</td>
<td>514554</td>
<td>80026</td>
</tr>
<tr>
<td># Site/Year combinations</td>
<td>3247</td>
<td>1145</td>
</tr>
<tr>
<td>Total # sites</td>
<td>972</td>
<td>991</td>
</tr>
<tr>
<td># Sites in 2000</td>
<td>517</td>
<td>256</td>
</tr>
<tr>
<td># Sites in 2001</td>
<td>554</td>
<td>280</td>
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<tr>
<td># Sites in 2002</td>
<td>671</td>
<td>218</td>
</tr>
<tr>
<td># Sites in 2003</td>
<td>731</td>
<td>216</td>
</tr>
<tr>
<td># Sites in 2004</td>
<td>774</td>
<td>175</td>
</tr>
<tr>
<td>Maximum # toxicants per site/year</td>
<td>285</td>
<td>117</td>
</tr>
<tr>
<td>Minimum # toxicants per site/year</td>
<td>8</td>
<td>1</td>
</tr>
</tbody>
</table>

According to the EU-WFD “one-out-all-out” principle, the annual average concentrations of the WFD priority pollutants (Lepper 2005) were checked against the Annual Average Environmental Quality Standard values (AA-EQS) for those toxicants (top rows of +-signs in Figure 1). Separately for water and sediment data, the average and maximum concentrations of toxicants per site and year are converted to a prediction of the loss of species attributable to whole mixtures (msPAF – multi substance Potentially Affected Fraction of species based on acute EC50 data). This conversion is done by sequentially applying the following models:

1. Bioavailability model for individual toxicants depending on substance properties and local environmental conditions (De Zwart et al. In press)
2. Species Sensitivity Distribution model for individual toxicants (De Zwart 2002, Posthuma et al. 2002)
3. Mixture toxicity model depending on the mode of action of individual toxicants (De Zwart and Posthuma 2005)

The variable msPAF-EC50, commonly called the acute toxic pressure, estimates the fraction of tested species that would be seriously affected when reared in a water body with the local mixture.

Figure 1 illustrates that for sediments the maximum loss of species (for the averaged concentration data) is predicted to exceed 10% at 45% of site/year combinations. For water the maximum loss of species is likely to exceed 10% at 20% of site/year combinations. Figure 1 also demonstrates that the predicted loss of species is far more diverse between site/year combinations when exposure is through toxicants present in the water phase, compared to toxicants contained in sediments. The most important conclusion from Figure 1 is that on-average quality standard compliance frequently coincides with a prediction of considerable impact on biodiversity, implying an underestimate of possible impact magnitudes when one would consider local, individual peak concentrations. Exceedence of EQS values per compound appears not to be a guideline towards quantifying local impacts of local mixtures. Note that the msPAF-values are based on a larger number of compounds than regulated. In other words: there are too few compounds regulated and EQS-base evaluation ignores mixture toxicity and the toxicity reducing aspect of bioavailability.
Fig. 1: Average and maximum predicted loss of species plotted against the proportion of site/year. The blue and red + signs indicate compliance and exceedence of AA-EQS values for WFD priority pollutants.

**Thesis 2: Predicted ecological impact of toxicant exposure can be observed in the field**

The results of testing Thesis 1 concerned model predictions of acute impact magnitudes, but did not show which impacts were really observed. Thus, this thesis concerns linkage of predicted to observed impacts.

For 489 of the above site/year combinations the MODELKEY BASIN database for water analysis also contains data on macroinvertebrate census. Plotting the observed loss of taxa (defined as (Max number of taxa - Observed number of taxa) / Max number of taxa) against the predicted loss of species approximately reveals a one-to-one relationship in the envelope of the scatter plot for the data rich part of Figure 2. High observed loss of taxa coinciding with low predicted loss of species from toxicant exposure (left upper side of the graph) is most probably caused by adverse conditions in other stress factors not related to toxicity. In other words, acute toxic pressure might relate linearly to observed loss of taxa, but the loss of taxa might be (much) higher due to other stress factors.

Fig. 2: Observed loss of taxa plotted against predicted loss of species. The red line indicates the envelope of the scatter plot. The grey area represents the part of the graph where the number of data points is too low for drawing the envelope of the scatter plot.
Thesis 3: The impact of toxicant exposure should be evaluated together with the impacts of all other stress factors

Outcomes of thesis 2 lead to the conclusion that an integrated stress assessment is needed. Linking a series of models allowed us to quantify the ecological impact in terms of loss of taxa and to attribute this loss to different stress factors. The results are mapped as simple Effect and Probable Cause pie charts (EPC pie-diagrams), with pie sizes corresponding to the magnitude of local impact (observed loss of taxa), and slice sizes to the relative probable contributions of different stress factors (Figure 3). The models used are described in detail in De Zwart et al. (2006):

1. Pie size: Loss of taxa deducted from site/year combinations with top-15 Average BMWP Score Per Taxon (ASPT > 5.5) (Hawkes 1997)
2. Generalized Linear Modelling (GLM): Relates abundance of individual taxa to the variation in multiple stress factors, including the acute mixture-toxicity parameter (Nutrients: Kjeldahl N - NH4+ - PO4 – Ptotal; Organic load: BOD – DO; Chemistry: Conductivity – pH – TSS; Toxicity: msPAF)
3. Misfit between observed and GLM-predicted taxa abundance quantifies unknown causes
4. Attribution of impact to other stress factors is accomplished by restricting negative contributions of individual stress factors in the GLM prediction to the taxa locally lost.

Fig. 3: Effect and Probable Cause (EPC) pie charts for 489 site/year combinations in the Scheldt River basin. Size of the pies is proportional to impact (i.e., large pie = large impact). Size of slices is relative to probable cause. Stressors grouped in 5 main types (colours) for ease of interpretation.

Figure 4 gives the unrelated distribution of the impact and stress factor attribution for all 489 site/year combinations. Both Figures 3 and 4 suggest a minimum loss of taxa as compared to the selected reference condition of close to 50%. This is most probably an aberration introduced by taking the taxa composition of the top-15 ASPT sites as a single imaginary reference. Availability of information on the divergence of species compositions of a set of regional reference sites, in dependence of local and undisturbed water quality characteristics, would allow for a more proper estimate of the magnitude of local impact. The high proportion of unexplained effects is understandable because other stressors known to affect aquatic ecosystems, such as habitat alteration, input of cooling water, shipping and fishing activities, were not included in our analyses. The analyses make clear that, despite the major role of other (sometimes unknown) stress factors, there is a significant contribution of local toxicant mixtures to taxa loss in the study area, which is highly variable between sites. This confirms the interpretation presented for Thesis 2.
Fig. 4: Unrelated distribution of impact and grouped stress factor attribution over 489 site/year combinations in the Scheldt River basin.

**Thesis 4: Neural network technique also demonstrates relationship between chemical stress and ecological status**

MODELKEY’s sub-project EFFECT also develops some artificial neural network (ANN) techniques. One of them is an ordination method allowing the classification of sites based on similarities between the biota communities (Self Organizing Map, Kohonen 1995). The community data for the Scheldt River basin are linked to the mixture toxicity predictor (msPAF). The study sites are clustered to indicate that chemical stress is indeed a factor determining ecological status (Figure 5).

Fig. 5: Scheldt River data on macroinvertebrates in water for the year 2004 clustered in a self organizing map. The clusters are plotted on the GIS map for the Scheldt basin. The distribution of the mixture toxicity predictor (msPAF) is presented in box plots for the different clusters.
The sites clustered in the clusters numbered 2 to 5 are characterized by relatively low toxic pressure. These sites more or less correspond to the sites characterized by low loss of taxa as identified in Figure 3.

**Recommendations & perspectives**

A major conclusion of the comparison of two classical methods (checking compliance to EQS per regulated compound, and looking at impacts) was that they yielded results suggesting different Programs of Measures: local impacts can be higher than expected based on EQS-exceedence alone (results Thesis 1), and are apparently caused by all stressors operating together (results Thesis 2). By defining one assemblage-level parameter to quantify the toxic pressure of mixtures, the acute toxic pressure msPAF, there was latitude to link and interpret data sets on water characteristics and the occurrence of macroinvertebrates, while increasing statistical power (results Thesis 3). By combining various (eco)toxicological and partly statistical models, impact magnitudes could be unravelled into probable causes (results thesis 3). This analysis suggested not only the likely causes of local impacts, but also provided confirmation of msPAF being a relative estimator of mixture impacts (results Thesis 3 and 4).

All results being preliminary as yet, further developments can be foreseen. By separation of the toxic stress prediction (msPAF) for groups of toxican ts originating from different human activities (agriculture, transport, types of industries) we can reveal the relationship between different pollution sources and chemical status. This is a relatively easy task, since msPAF is derived from the single-substance values of PAF (ssPAF), which can be grouped per compound-type into Mode-of-Action specific msPAF-estimates. Similarly, the EPC analyses can be improved by reducing the “Unknown” impacts, by introducing more potential stressors in the analyses, and by better defining reference conditions.

The availability of WFD monitoring data on reference conditions, habitat degradation and other potential stress factors would greatly enhance our ability to:

- Detect chemical stress (by mixtures, by groups of similarly acting compounds, and by individual compounds) as a factor causing impacts on the ecological status
- Quantify the relationship between chemical stress and ecological status
- Differentiate between the ecological impacts of different stress factors

thereby reducing the frequent misfit between the results of two classical evaluation approaches, and eventually resulting in more effective Programs of Measures to improve ecological status of water bodies.

**References**


As an ecologist and ecotoxicologist Dick de Zwart is mainly working on the diagnosis of ecosystem impairment by the application of diverse eco-epidemiological techniques.

Major projects include:
- RIVM program on R&D for ecological risk assessment
- MODELKEY (SP EFFECT and DSS development)
- UK Environment Agency – Eco-epidemiological analysis of water quality impairment
Supporting river basin management at different scales: 
A risk-based DPSIR framework

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Background

The main regulatory reference for the management of river basins in Europe is represented by the EU “Directive 2000/60/CE establishing a framework for Community action in the field of water policy”, known as EU Water Framework Directive (WFD; EC, 2000). It requires Member States to properly assess and manage water bodies in river basins in order to protect and improve water quality as well as to ensure sustainable use. In particular, the WFD calls for achieving a good ecological and chemical status across the Europe by the end of 2015.

In the light of the WFD requirements, a series of consecutive tasks are to be accomplished by water managers and decision-makers starting from pressures and impact analysis, environmental monitoring, chemical and ecological status classification, setting of environmental objectives, selection of management measures, and leading to the production of the River Basin Management Plan (RBMP) in 2009 (see the Management Planning Cycle in Figure 1).

In this context, the DPSIR (“Driving forces”, “Pressures”, “State”, “Impacts”, “Responses”) framework developed by the European Environment Agency (EEA, 1999) has been identified as instrumental in the implementation of the WFD since many of the tasks required by the Directive refer directly to the elements of the DPSIR framework (Rekolainen et al. 2003). In particular, Rekolainen et al. (2003) have proposed a modified framework for the implementation of the WFD, called DPCER, where the “State” and “Impacts” indicators are replaced by the “Chemical state” and the “Ecological state”, respectively. As shown in Figure 1, such framework specifically addresses the assessment phase of the Management Planning Cycle by identifying “Driving forces”, “Pressures”, “Chemical state” and “Ecological state”, while the production of the RBMP required in the management phase corresponds to the identification of the “Responses”.

![Fig. 1: Integration of WFD Management Planning Cycle and DPCER framework.](image-url)
A risk-based DPSIR framework

Considering both the DPSIR framework adopted by the European Environment Agency (EEA, 1999) and the DPCER scheme outlined by Rekolainen et al. (2003) for supporting the WFD implementation, a risk-based DPSIR framework is proposed for river basins assessment and management fulfilling each element by means of risk-based methodologies and tools.

The first phase of the risk-based DPSIR framework aims at identifying significant “Driving forces” (D), and related “Pressures” (P) causing potential impacts (hazard) on river basins with the final goal of estimating the risk that water bodies will fail to achieve the good ecological status required by 2015. To this end, both socio-economic information on trends in key economic drivers and environmental data on current quality status and vulnerability of water bodies are needed. In order to achieve this objective the increasingly adopted Regional Risk Assessment (RRA; Landis, 2005) approach could be applied since it is able to provide a relative ranking of areas, stressors and receptors along river basins by integrating two components, i.e. sources and stressors spatial distributions with vulnerability assessments.

The second phase of the risk-based DPSIR framework focuses on evaluating the “Status” (S) of water bodies. As pointed out by the DPCER scheme (Rekolainen et al., 2003), this phase, aiming to detect potential risk situations along river basins caused by priority substances or other pollutants measured in the water column, sediment or biota tissues of aquatic organisms, is mainly linked with the “Chemical status” definition. This screening like assessment at basin scale needs toxicological data quantifying toxic or carcinogenic effects caused by chemicals on generic aquatic species. On this regard, to derive risk-based Environmental Quality Standards (EQS), the probabilistic approach of Species Sensitivity Distributions (SSD; Posthuma et al., 2002), aggregating toxicological data referred to different organisms, is currently applied at international level. The comparison of measured or predicted chemical concentrations with appropriate EQS will allow, in this phase, the identification of substances potentially causing ecological risks.

In the third phase aiming at analysing “Impacts” (I) two main objectives are identified: to evaluate the ecological status for each water body identified along river basins as required by WFD as well as to investigate causal relationships at hot-spot scale, i.e. on those areas resulting of major concerns. In both cases, a number of different quality information (i.e biological, physico-chemical, chemical and hydromorphological data) have to be integrated and evaluated in order to achieve comprehensive and confident results. To this end, the Weight of Evidence approach (Burton et al., 2002) is regarded to be one of the most appropriate tools, since it allows to evaluate environmental impacts by integrating multiple lines of evidence, so that the likelihood of ecosystem impairment is higher if more assessment results suggest it.

The last phase of the DPSIR risk-based framework aims at identifying and selecting adequate “Responses” (R), i.e. technical measures, mitigation measures or policy instruments, for protecting or improving water quality of river basins in order to maintain or restore the good ecological status by 2015. This management phase needs decision-support tools guiding water managers in taking decisions on intervention alternatives, in assuring stakeholders involvement and participation, in communicating results in a transparent and simple way. To this end the development and application of a risk-Decision Support Systems (DSS) is widely recommended: it is able to interlink different assessment methodologies and support tools in a comprehensive structure making the decision process flexible, repeatable, changeable, traceable and transparent.

Within the MODELKEY project (DECIS subproject), the proposed risk-based DPSIR framework is implemented in the MODELKEY DSS according to two main assessment phases:

- the Preliminary Assessment phase aiming at analysing “Driving Forces” and “Pressures”;  
- the Integrated Assessment phase focusing on evaluating “Status” and “Impacts”, structured into a Screening step at basin scale and a Definitive step at hot-spot scale.

Both phases rely on Multi-Criteria Decision Analysis methods (Kiker et al., 2005) supporting the decision-making process of evaluation and selection of “Responses”.

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Recommendations & perspectives

The proposed risk-based DPSIR framework offers a structured solution to the complex assessment required by the WFD on water quality in the risk management perspective. In this context, there is a clear need of improving risk assessment methodologies by including spatial and temporal scale issues and quantitative risk estimation based on probabilistic assessment and uncertainty evaluation. Therefore, the development of specific methodologies and tools supporting stakeholders and decision makers in fulfilling the WFD requirements is particularly needed. However, in order to target the most recent scientific research outcomes to everyday needs of end-users, a close communication must be established between scientific and policy communities.

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Professor of Environmental Chemistry at the University Ca’ Foscari of Venice, Italy (http://venus.unive.it/eraunit/). His major research interests include a) analysis and environmental behaviour of micropollutants in waste waters, natural waters, biological media, sediment, and contaminated soils; b) development and application of risk assessment procedures to remediation of contaminated sites and ecosystems. He is involved in several European projects, as well as in the Euro-Mediterranean Centre for Climate Change (CMCC, www.cmcc.it).
List of Posters
## Topic 1: Ecosystem goods and services in risk-based management

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<td>¹Department of Systems Ecology and Sustainability, University of Bucharest, ²Faculty of Hydrotechnics, Technical University of Civil Engineering of Bucharest, ³National Research Development Institute for Environmental Protection</td>
<td>Romania</td>
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<td><a href="mailto:Josef.Settele@ufz.de">Josef.Settele@ufz.de</a></td>
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## Topic 2: Water regulation at risk under global change

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<td>Simulation of flood inundation areas under changing conditions</td>
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<td>Wölz J., Maier M., Dirk K., Fleig, M., Maier D., Mohrlock U., Lehmann B., Hillebrand G., Hollert H.</td>
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<td>Potential in-situ measures for risk reduction of contaminated sediments – Elbe river basin</td>
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<td>Jarkovský J.¹,²,³, Kubošová K.¹,², Némethová, D.¹,², Dušek L.¹,², Zahrádková S.³, Ráček J.¹,², Hodovský J.⁴</td>
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Poster Abstracts
Assessment of functions, services and resources in aquatic ecosystems as basis for the management of hydrographic systems

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Sustainable use and natural resources management are political priorities of the last decades. Water Framework Directive /WFD established the general framework for action within water resources policies. For a coherent implementation of this directive EU countries developed a common strategy for management, use, protection and restoration of surface and ground water resources of hydrographic basins and coastal area. Development of methodologies for classification and assessment of ecological state as well as of ecosystem services of aquatic ecosystems is one of the main components of WFD implementation.

The limitative effect of water resources on socio-economic systems and human welfare will be increased by the the sectoral and reductionist approaches used in the scientific knowledge development regarding aquatic systems until the beginning of this decade. Their management wears the mark of the same limits at conceptual, scientific and methodological level which drove to substantial changes on long term into their composition and structure (conversion of wetlands into agricultural fields; water courses regularization; dams; reduction or extinction of habitats; biological diversity loss) and their multifunctional character (function of production, regulation and control, support and informational) or diversity, quality and quantity of resources and services services (Norberg 1999, Pinay et al. 1990, Turner et al. 2001, de Groot 2002, Costanza and Farber 2002, Maltby et al. 1996, Vadineanu et al. 1992, 2003, 2004).

The growing demand for ecosystem services is compounded by increasingly serious degradation in the capability of ecosystems to provide these services. For instance fisheries, as depending of the ecosystem production function, are now in declining due to some multiple causes such as reduction of spawning and growing wetland habitats, interruption of river lateral and longitudinal connectivity, cultural eutrophication and overfishing. Flood detention and nutrient retention capacities, as important regulation functions, also declined. Size and distribution of many plant and animal populations as well as supporting capacity for the metabolism of the local associated socio-economic subsystems, as important supporting functions, have been severely affected. Finally, the recreation potential as well as the aesthetic and cultural values, as important information functions, have been also affected.

The problems approached are in fact, the scientific base for the sustainable management of the hydrographic basins and in the same time a preliminary step towards the economic assessment of the aquatic resources and services.

In order to develop a package of methods and proceedings for identification the functions and to estimate the fluxes of resources and services provided by lotic and lentic aquatic ecosystems (“water bodies”) with the purpose of transposing into practice the stipulations of EU directives (especially WFD) and of international conventions (especially Convention on Biological Diversity and Ramsar Convention) regarding conservation, reconstruction and sustainable use of aquatic ecosystems, activities were designed in four complementary plans: i) developing and perfection of necessary instruments for classification and estimation of ecological state on one hand and, on the other hand for identification and valuation of functions, resources and services fluxes provided by lotic and lentic systems; ii) using the developed methods and proceedings packages and the historical databases owned by partner institutions, in order to identify and estimate the functions, resources and services, inside the aquatic systems categories and as a function of their ecological management plans at catchment scale; and iv) facilitating the transfer toward end-users and immediately applications of the scientific product.

After a general characterization of the hydrographic basins at national level, two of them were selected as representative for implementation of the project, respectively Arges and Ialomita hydrographic basins, both including important socio-ecological complexes belonging to the European Network of
Long-term Ecological Research Sites (Neajlov basin site respectively Bucegi-Piatra Craiului site). A series of aquatic ecosystems were selected for implementation and testing of revised methodologies for identification of functions and estimation of resources and services fluxes.

The understanding of socio-economic system dynamics will allow also to identify the drivers and pressures of aquatic ecosystems services.

The complexity of problem required to correlate both multidisciplinary activities as well as a hierarchical approach, from aquatic system classification at the national level, establishment of the reference conditions for some of the identified representative systems and up to the identification of goods and services provided by aquatic ecosystems and continuous knowledge transfer to the users and potential users.
Environmental risk assessment for biodiversity and ecosystems: Results and perspectives of the large scale inter- and transdisciplinary research of the ALARM Project

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ALARM stands for “Assessing LArge-scale environmental Risks for biodiversity with tested Methods”. The consortium is co-ordinated by the Helmholtz-Centre for Environmental Research - UFZ. It is an Integrated Project (IP) within the 6th Framework Programme of the European Commission (EC) with more than 250 scientists, representing 67 institutions from 35 countries.

Based on a better understanding of terrestrial and freshwater biodiversity and ecosystem functioning, ALARM develops and tests methods and protocols for the assessment and forecast of large-scale environmental risks in order to minimise negative direct and indirect human impacts.

Research relates to ecosystem services and includes the relationship between society, economy and biodiversity. In particular, risks arising from climate change, environmental chemicals, biological invasions and pollinator loss in the context of current and future European land use patterns are assessed.

There have been an increasing number of case studies on the environmental risks subsequent to each of these impacts. These yield an improved understanding on how these act individually and affect living systems. However, knowledge as to how they act in concert is poor and ALARM is the first research initiative with the critical mass needed to deal with such aspects of combined impacts and their consequences.

KEYWORDS: risk assessment, biodiversity, global change, climate, land use, environmental chemicals, pollination, invasive species, socio-economy, scenarios
Simulation of flood inundation areas under changing conditions

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After the extreme flood event in 2002 the German Water Resources Act was enhanced through the paragraphs §31b and §31c for the localisation of flood risk areas throughout the country. However, there are still challenges concerning a sustainable definition of potential flood risk areas under changing conditions. Besides the experiences of the 2002 flood, possible spatiotemporal changes and the internal variability in the boundary conditions of extreme flood events should be considered in the determination of flood risk areas by law.

Especially urban areas require effective and differentiated approaches for the modelling of flood risk areas due to the accumulation of monetary values. The historical urbanisation in the last 200 years with a continue propagation into the natural floodplain areas changed and intensified the vulnerability and risk as well as the extent of the recent inundation areas.

Because of this fact the authors try to combine two different model approaches to consider possible land use changes, including their potential monetary losses, in the flood inundation and flood risk simulation. Using the hydrodynamic model TRIMR2D and the urban land use model RESMOB the impact of spatiotemporal effects of on-site land use changes are investigated.

The main objectives of the study are the improvement of the prediction of the potential inundation areas as well as a better estimation of future flood risk and vulnerability under changing conditions. Focussing on the availability of detailed data and modelled results there, the city of Leipzig is selected as investigation site.
Mitigation of contaminants in rural and semi-rural environments to protect surface water for drinking water supply - The Aquisafe-project

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Major reservoirs are a key element for public water supply in many countries. In Europe over 800 major reservoirs serve primarily this purpose. Eutrophication affects significant numbers of lakes and reservoirs, and is the well-known issue currently impacting drinking water supply reservoirs. In most cases, phosphorus is the principal cause of eutrophication, and therefore has been studied intensively. The presence of micro pollutants (e.g. pesticides, pharmaceutically active compounds - PhaCs) is not systematically monitored but some substances are very mobile and tend to resist degradation. Such contaminants have been detected in numerous surface water bodies (lakes, reservoirs and rivers). As agriculture is intensifying and land use is changing in many areas, the impact of diffuse pollution on water quality is expected to be more pervasive in the future.

The project Aquisafe proposes to investigate the topic in a multi-step approach which will include: i) an analysis of the nature, occurrence and risk of surface water contamination, ii) a modelling approach to quantify the contaminants origin, load and repartition to assess the effects of adapted controlled measures, and iii) the development, adaptation or optimisation of the design and operation of mitigation zones (riparian corridors and small scale wetlands) to reduce downstream loads of pollutants. Thus, Aquisafe is a first step to establish the state-of-the-knowledge on current existing solutions, identify emerging issues and assess the feasibility of using models for the evaluation of mitigation zones for contaminants removal.

Within the Aquisafe project it will expected: i) a recommendation on potential key substances to be targeted, also for further investigations, ii) an identification of drinking water source vulnerability to emerging contaminants using a coupled modelling approaches, and iii) an analysis of existing mitigation methods and scientific background for the construction of riparian corridors and/or constructed wetlands in order to mitigate trace contaminants entering the surface water.

KEYWORDS: trace contaminants, water supply, hydrological modelling, pesticides, water quality
Flood retention and drinking water supply - Preventing conflicts of interest

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Conflicts of interests are expected for virtually all major rivers in Germany. On the one hand, retention areas are designated to minimize the threats associated with extreme flood events. On the other hand, retention areas often overlap with riparian areas that act as drinking water reserve. Water suppliers are concerned that water flooding in retention areas eases the entry and transfer of contaminants into the riparian aquifer.

Within this Joint Research Project the dominant processes and mechanisms along the transport path from flood wave via retention area and groundwater to the waterworks are assessed. Field trials are accomplished in the planned retention area Bellenkopf/Rappenwört near Karlsruhe, Germany.

Suspended particulate matter (SPM), soil and ground water are sampled regularly to be chemically and biologically analyzed. Samples are assessed using acute and mechanism-specific biological tests representing major ecotoxicological endpoints. Previous testing indicated significantly increased Dixon-like and mutagenic activity by EROD assay and Ames Fluctuation assay assessing SPM and soil samples. Groundwater samples showed increased and strongly varying endocrine-like activity comparing sampling sites and times.

Finally, strategies to minimize mutual impairments of flood retention and drinking water supply will be devised and summarized to a guideline.
Extreme event effects on diversity in flooded grassland at the river Elbe

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The composition of vegetation in meadows is normally affected by use and additionally in floodplain areas by the water regime. The effect on abnormal disturbance such as the extreme flood event in the vegetation season in 2002 is poorly unknown.

In 1998/99, meadows with shorter and longer flooded areas and oxbow channels were investigated. Starting from this investigation the consequences of the extreme flood event for the vegetation were studied on the same plots like 1998/99 from 2003 to 2006. The aim of the study is to investigate:

1. What effects does an extreme flood event have on the plant species composition of floodplain grasslands?
2. How do different moisture grassland types respond to an extreme flood event?

For 1998 and 1999, the total number of species and the number of endangered species are very similar, whereas the numbers of both groups decreased in 2003. From 2004 to 2006, the gamma-diversity and the number of endangered species are comparable to the pre-flood time. Concerning the three different moistures grassland types- wet, intermediate and dry-, grasses and herbs show no significant differences between pre- and post-flood time in the oxbow channels. A few significant differences could be found for the herbs between 1998 and 2004, 2005, 2006 in the intermediate plots. The most significant differences were demonstrated for the dry plots: The dry plots in 2003 are characterised by a significant lower number of grasses and herbs compared to the pre-flood time and compared to 2004, 2005 and 2006.

It is assumed that floodplain meadows are very stable systems and regenerate within few years after disturbance. However, considering the recent discussion about climate change and therefore a possible higher frequency of extreme flood events, the future development of species composition and abundances may change significantly.
Research on Erosion Risk Assessment in Groyne Fields

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The erodibility of cohesive sediments and associated contaminants in natural rivers is difficult to predict due to complex interaction of sediment physical, chemical and biological properties, and hydraulic flow conditions. The spatial and temporal variability causes a large uncertainty. As there is no general analytical theory describing cohesive sediment resuspension, the quantitative determination of erosion is based on experimentally derived equation \( E = M \left( \tau_s / \tau_{cr,E} - 1 \right)^n \).

In the literature the magnitude of erosion parameters, defined in the erosion equation, varies widely depending on: (1) the properties of the sediment mixture density, grain size, TOC, KAK, water content, and (2) testing devices and methods used for parameter determination. Therefore, it was a challenging task to perform field experiments in order to measure the erosion parameters, and to assess their applicability in numerical modelling. This research was focussed on the River Elbe, which has been trained by groyne structures on both river sides.

Depth orientated critical shear stress for erosion \( \left( \tau_{cr,E} \right) \) of undisturbed sediment cores taken in three groyne fields of the River Elbe, was measured, using the SETEG-System. The results showed large variability of the critical erosion shear stress depending on: (1) location of a groyne field along the river course, (2) the sampling spot within a groyne field, (3) sediment depth, and (4) season of year. A wide range of sediment properties were determined, in order to quantify the properties affecting the erosion stability. A regression function based on 4 parameters describing the erosion behaviour was established.

Further, erosion rate was measured in order to calculate erosion coefficient \( (M) \). It was shown that two patterns for resuspension behaviour could be distinguished due to sediment properties in different sediment layers.

Due to high variability of measured parameters, it is very difficult to quantify the effective value of the erosion parameters, which represents the whole river reach and can be used in numerical calculations. Thus, numerical simulations were performed in order to explore the applicability of traditional approach of applying the mean measured parameter value in the numerical model. An innovative approach, presented in this research, was to determine the spatial distribution of the parameter by means of statistical analysis. The basic idea was to generate the random values of parameters based on distributions of the measured data. Generated values of the critical erosion shear stress were determined for each groyne field in the simulated domain of the Elbe, from Wittenberg to Magdeburg.

The influence of a critical erosion shear stress on total eroded volume for a river stretch of 112 km was shown for different values of the parameter: (1) mean value, (2) minimum value, (3) mean value plus standard deviation, and (4) randomly longitudinally distributed values. It was shown that for randomly distributed parameter the differences in bed elevation occurred depending on both the critical shear stress change and the groyne field size, which was not the case if constant value of parameter was used. This illustrates that the statistical realization allows to quantify the statistical properties of the output and hence, the assessment of the impact of input uncertainty on the output. That implies the fact that a deterministic calculation (constant mean value) does not allow an uncertainty assessment, which is necessary for environmental issues.
Macroinvertebrate community response upon degradation of the channelized stream

(Case study Upper Elbe, Czech Republic)

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Benthic macroinvertebrates are an important indicator of river health. However, their response upon water quality development downstream the pollution outlets considerably depends on the environmental habitat characteristics. Three successive sections, each of them providing three different mesohabitats (riffles, still water and riparian zone), were selected for evaluation of the impact of altered physical and chemical determinants of habitat quality on the Upper Elbe. In downstream direction, the sections were characterized as (a) unpolluted natural stream, (b) unpolluted channelized stream and (c) polluted channelized stream. Altogether, there were 135 taxa of benthic macroinvertebrates recorded in the Pardubice hotspot (between Němčice and Přelouč). Despite different level of stream bed and water quality degradation, micro- and mesohabitat determinants appeared to be the most important factor affecting the diversity of macrozoobenthos in riffles (substrate size structure) and in shoreline zones (macrophyte community composition). The diversity of macroinvertebrate communities increased downstream and was highest in the polluted and channelized stretch of the river between Valy and Přelouč. However, the quantitative and qualitative figures of macrozoobenthos in muddy substrates of still water zones upstream the Elbe weirs did not vary considerably. Saprobiological evaluation of the sites under study proved the best indices in riffle torrentile zones followed by riparian and still water ones. In downstream direction, the saprobiological indices decreased, thus indicating the worsening of water quality determinants.
Bioaccumulation of $^3$H-Ivermectin in *Lumbriculus variegatus* - Preliminary results

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A study on the bioaccumulation of the widely used veterinary parasiticide Ivermectin was performed with the benthic oligochaete *Lumbriculus variegatus*. Within the EU-Project ERAPharm (SSPI-CT-2003-511135), *L. variegatus* was exposed to artificial sediment spiked with $^3$H-ivermectin for 28 days. Sediment, worm and water samples were taken during the uptake phase and analysed for total radioactive residues. Samples of sediment and worms taken at the end of the uptake phase were analysed for percentage of radioactivity associated with the parent compound. Following the uptake phase, elimination of the accumulated $^3$H-residues from the worms was monitored. The uptake and elimination kinetics were determined by nonlinear regression analysis.

Based on the mean measured $^3$H-ivermectin concentrations in the sediment, the bioaccumulation factor (BAF) based on worm and sediment wet weight approached 4.9 at the end of the uptake phase. Approximately 94% of steady state of uptake was reached within the exposure period of 28 days. After a 10 day elimination phase, the worms had eliminated 61.5% ± 6.3% (SD) of the $^3$H-concentration at steady state, i.e., the non-eliminated residue (NER$_{10d}$) was 38.5%. Extrapolation of the elimination curve indicated that 90% of the steady state $^3$H-residues would be eliminated after 22 days.
Development of a European environmental Risk Assessment Toolkit (RAT) for biodiversity and ecosystems

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ALARM (Assessing LArge-scale environmental Risks for biodiversity with tested Methods), an Integrated Project funded under the European Union’s Framework Programme 6, aims to develop methods to predict risks to a range of aspects of biodiversity associated with the combined action of key drivers such as climate and land use change, environmental chemical pollution, biological invasion, and loss of pollinators. Such risk assessments should account for interactions between drivers, and reflect natural variability and qualitative and quantitative uncertainty in our knowledge. An important challenge in communicating biodiversity risk assessment is therefore how to present such a complex and uncertain picture across scientific disciplines, and to policy makers, politicians and the wider public. Here we describe the initial development of a European Risk Assessment Toolkit (RAT) by the ALARM consortium which aims to provide an interactive tool to allow users to explore uncertain multiple and varied biodiversity risk assessments generated at the European scale by the ALARM project. The aim is to provide access to individual risk assessments, but also allow sets of results to be grouped together according user defined criterion (e.g. all measures of aquatic environments), enabling end-users to build-up more complete pictures of areas of particular concern.

KEYWORDS: probabilistic risk assessment, impact modelling, large-scale risk assessment
Environmental risk assessment of nitrite emission peaks towards the Loire River

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In exceptional situations related to technical incidents in water treatment systems large amounts of nitrites may be released from industrial installations. The aim of this study is to evaluate whether a potential emission from installations on the Loire River could have an impact on the receiving aquatic ecosystem. For this purpose the degradation kinetics of nitrite were studied in batch systems containing original water samples from the Loire River aiming to mimic the real environmental conditions. The observed kinetics were integrated in a dispersion model and an expected worst-case nitrite concentration profile was modelled for the river system. For the estimation of a predicted no effect concentration (PNEC), a species sensitivity distribution (SSD) approach was chosen in order to integrate all available ecotoxicological data for the assessment. It can be shown that for the worst-case scenario (high nitrite emissions at low river flow rate), peak concentrations will remain below estimated acute PNEC values. Furthermore, due to oxidation to nitrate by microorganism activity, nitrite concentrations will be reduced to background concentrations before other potential emission points. It can thus be concluded that no unacceptable environmental hazard related to a possible occasional release of nitrite could be expected.
French research project on sediment quality: DIESE (Tools for Diagnostics of Sediments Ecotoxicity)

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The project named DIESE (Tools for Diagnostics of Sediment Ecotoxicity) will be funded by the French National Research Agency (ANR- Agence National de Recherche) from 2008 to 2011. The aim of this project is to develop a reliable graduated and cost-effective strategy for risk assessment of contaminated freshwater sediments by combining chemical and biological approaches. For the development of the assessment strategy, the response of a biotest battery using five entire organisms of different trophic levels will play a central role. It is considered as a reference for the judgment on sediment quality as it integrates different aspects of a potential risk: presence and bioavailability of toxic compound and possible mixture effects. Furthermore it is believed to be ecologically relevant as organisms of different levels of biological organisation are represented. Around this central element of assessment, different work packages are arranged which aim to:

- evaluate the ecological relevance of the biotest battery results by comparison to the state of the biological community at the sites where the sediments were sampled,
- apply rapid microbiotests in order of elucidate their pertinence for integration as a first step of an integrated risk assessment strategy or as a possible surrogate for whole organism tests,
- identify toxicants responsible for observed effects in the biotest battery by using toxicity identification evaluation approaches (TIE),
- develop tools to assess the bioavailability of toxicants in sediments and their bioaccumulation in different organisms,

The developed assessment strategy will be applied and validated in different case studies.
Ecological and health risk caused by heavy metals in the vicinity of the metallurgical waste disposals at the example Bukowno (Poland) and Mansfeld (Germany)

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Mining and smelting processes of Zn-Pb ores at the Bukowno (Poland) and of Cu at the Mansfeld (Germany) cause considerable contamination of water-soils systems with heavy metals and arsenic. The previous studies showed high concentration of heavy metals and arsenic in the soils at the area of Zn-Pb mining and smelting industries near Bukowno and Olkusz and at the area of Cu industry near Mansfeld. The flotation wastes, and the flue dust wastes, respectively have been the main sources of metal pollution. Metal concentrations were investigated in the water and bottom sediments of the Rivers – Baba and Roznos in the surrounding of deposited flotation wastes in Bukowno and in the River Glume in the vicinity of the flue dust disposal in Mansfeld, as well as in soils in both regions. Upon the determined concentration of heavy metals the Ecological Risk Assessment (ERA) and the Human Health Risk Assessment (HHRA) were assessed. In both regions for all environmental components as well as for all investigated heavy metals the assessed PEC/PNEC >1. That means, that environmental risk caused by heavy metals exists and it should provoke a remediation project. In case of HHRA analysed exposure pathways were: surface water ingestion, soil ingestion, soil dermal contact, outdoor air inhalation. The HHRA was determined separately for carcinogenic substances - in this case As, and for non-cancerogenic substances - here rest of analysed heavy metals. Estimated HHRA for As in Mansfeld is at the level of $10^{-3}$, and in Bukowno at the level of $10^{-4}$, whereas the acceptable risk value is equal to $10^{-6}$. For heavy metals, defined here as non-cancerogenic substances, the highest risk was assessed in case of Cd and Tl in Bukowno and for Cd, Sb, Tl, Zn in Mansfeld. The HHRA caused by Pb was not assessed in this study due to the lack of values of the reference dose RfD needed for calculations. Calculated cumulative Hazard Index HI is equal to 0,9 for Bukowno and 9,1 for Mansfeld, which means, that the risk for human health in the vicinity of industrial waste site in Bukowno is low and in Mansfeld is medium.

Application of risk assessment analysis for contaminated sites still seems to be difficult in practical use, mainly due to the lack of clear and comprehensive procedure. On the other hand, risk-analysis procedure can be very helpful in preparing and evaluating remediation projects.

KEYWORDS: heavy metals, ecological risk assessment, human health risk assessment, river water, river sediments, soils, industrial waste disposals.
Effects of Prometryn and Parathion-methyl on aquatic organisms of different trophic levels

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Within the subproject EFFECT of the EU-project MODELKEY (511237 GOCE), direct and indirect effects of the pesticides Prometryn and Parathion-methyl on aquatic organisms representing different trophic levels were analysed in an aquatic indoor microcosm. The species used for the microcosm are the flagellate Cryptomonas sp. as producer, the ciliates Urotricha furcata and Coleps spetai as consumer and an unidentified bacterial community. All test organisms were cultivated in modified WC-medium according to Guillard & Lorenzen (1972). Since for modelling purposes experimental data on both dynamic and static phases of growth are preferred, the nitrate content of the medium was reduced by the factor 5 in order to achieve growth limitation of the flagellates after about 1 week of exposure.

All tests were performed in Erlenmeyer flasks filled with 100-150 mL test medium at permanent light. After the performance of single-species tests with each test organism, multi-species tests with flagellates and one ciliate species were performed. Thereby, direct and indirect effects could be observed. In further studies such effects are investigated in extended multi-species test systems, i.e. the influence of the test compounds will be assessed when also other factors like competition for the food source takes place.

The overall goal of these studies is to generate effects data which can be used to model effects on simple aquatic ecosystems based on the dynamic energy budget (DEB) theory (Kooijman, 2000). These models are supposed to contribute to the development of a Decision Support System within the MODELKEY project, which intends to assist and support competent authorities by implementing the European Water Framework Directive.
Impact of Ivermectin on *Daphnia magna* and *Chironomus riparius* in a water-sediment system

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Ivermectin is one of the most used veterinary parasiticides since more than 20 years. Within the EU-Project ERAPharm (SSPI-CT-2003-511135), an aquatic multi-species test was performed to investigate effects of cattle dung contaminated with ivermectin on the aquatic organisms *Daphnia magna* and *Chironomus riparius*. To simulate realistic exposure conditions ivermectin was spiked into cattle dung at concentrations ranging from 11 to 1314 ng/g dung d.w. and placed on the sediments surface. 20 chironomid larvae and 9 daphnids of defined, mixed age were exposed for 51 days. On days 10, 24, 38 and 51 replicates of the test system were destructively sampled. Survival, growth and emergence of chironomids and abundance, growth and biomass of daphnids were evaluated. In order to simulate immigration, test vessels were re-inoculated with daphnids when extinction of the population occurred. Further, after all chironomids were hatched, a second batch of chironomid larvae was introduced into the vessels on day 27.

The most sensitive parameter for the daphnids was the abundance. After 10 days of exposure no daphnids survived at the two highest test concentration levels. At the highest ivermectin concentration of 1314 ng/g d.w. no daphnids survived following to reinoculation at day 11, 28 and 42. For the chironomids the most sensitive evaluated parameter was emergence.
A screening-level ecological risk based model to assess the impact of hazardous substances in river systems supported on artificial intelligence techniques

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A methodological model to assess the impact on aquatic organisms caused by the presence of hazardous substances in river water and sediments has been developed. The model incorporates a novel ranking and scoring system based on self-organizing maps (SOM), a special un-supervised artificial neural network, to account for the likely ecological hazards posed by the relative presence of toxic substances in freshwater. Weighting factors were extracted from persistence, bio-concentration, and toxicity to aquatic sensitive organisms from SOM. Due to the high imprecision and uncertainty present in screening-level risk assessment studies, fuzzy reasoning has been used to compute substance specific ecological risk potentials, which are calculated as a combination of the hazard to aquatic living organisms, and normalized concentrations (measures of relative deviations of the environmental concentrations to selected benchmarks). The proposed model has been applied to a case study. A comprehensive database with environmental concentrations for toxic substances collected in the Ebro river basin has been analyzed with the SOM-fuzzy framework. The list includes heavy metals, pesticides, and other persistent organic pollutants, integrated to a Geographic Information System (GIS). The ecological risk potentials calculated in this study have permitted to deal with trends at regional level and evaluate the long-term performance of pollution prevention and control strategies set out by environmental protection agencies. The proposed framework can support decision-makers in evaluation and classification of chemical pollution as required by the EU Water Framework Directive.

KEYWORDS: Fuzzy inference systems, Self organizing maps, Ecological risk potentials, Ebro river basin.
GIS-based risk assessment for surface water risk at different scale: The example of agricultural chemicals

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The EU Water Framework Directive (WFD) states that the management of surface water must be based on a site-specific assessment of water quality, that is dependent on land use. In most recognised risk assessment procedures for chemicals, the approach is based on the evaluation of chemical-physical and toxicological parameters, applied to more or less standardised scenarios where the territory, at different scale levels (local, regional, continental), is described without taking into account the spatial variability of data. The description of landscape variability, in particular, is important when assessing the risk posed by non-point source pollutants, such as pesticides, because the variability strongly affects both emissions (related to land use and crop distribution) and environmental distribution and fate of the pollutant (depending on many environmental characteristics). To develop a robust chemical management policy for aquatic ecosystems, the ecotoxicological risk must be strictly related to the local conditions and characteristics of the system, and, in agreement with the requirements of the WFD, should be based on three different components; exposure assessment, effect assessment and characterisation of exposed ecosystems. For site-specific assessment all three components require a careful description and furthermore for effects and exposed ecosystems an ecological characterization is required. To overcome these difficulties, Geographical Information Systems (GIS) provide a powerful tool to integrate geo-referenced and non geo-referenced data in a site-specific pesticide risk assessment. This research was financially supported by the European Union (EC, FP6: ALARM and NoMiracle projects).

Using biological traits to improve the risk assessment of pesticides in Europe

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To assess pesticide effects in the field, an indicator has to unmask pesticide effects from confounding factors and should be applicable in different biogeographical regions. Recently, the trait-based Species-At-Risk (SPEAR) indicator successfully discriminated pesticide stress from the effects of environmental covariables in central Germany. Our aim was to validate its applicability on the European scale. Therefore, field studies were conducted in France and Finland, two countries with contrasting pesticide use.

In France, pesticide input influenced taxonomic and functional endpoints of the aquatic ecosystem. By focusing on selected traits we were able to distinguish between acute and chronic effects of pesticides. Chronic effects were present under concentrations that were considered to be safe with regard to risk quotients. Leaf litter breakdown was significantly lower in streams that suffered from pesticide input. In contrast, no effects were observed in Finnish streams.

Furthermore, we show that the SPEAR-indicator is capable of discriminating reference sites from polluted sites across France, Finland and Germany. Hence, we emphasize that biological traits are a promising tool for predicting effects of pesticides on the European level.

Effects of increasing temperature on soil parameters and trace metal dynamics in floodplain soils

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The River Elbe is one of the largest streams in Central Europe. Because of intensive usage of the Elbe catchment during the last 50 years the river and its catchment are heavily encumbered by a plethora of pollutants (cp. Krüger et al. 2005). Even though the water quality has improved since 1989 the Elbe floodplains still act as great sink for contaminants (Miehlich 2000). According to the IPCC (2001) the average surface temperature will increase about 1.4-5.8 K during the next 100 years. That will also have effects of the water supply, which will significantly change the appearance of extreme events like floods and droughts (OcCC 2003, IPCC 2001). According to Wechsung (o.J.) it means for the Elbe River that in spite of regional decreasing rainfalls the feasibility of flood events like in August 2002 further increases. Therefore it can be assumed that the global change lead to long term changes of soil parameters, which can result in modifications of alluvial soil bonds and thus to trace metal dilution in soils. In three sites typical for floodplains (levee [49], depression [42] plateau [32]) the influence of increasing temperatures (25°C, 30°C, 35°C) are investigated using a microcosm system.

During the 25°C-experiment the soil parameters show no changes (Eh-values: [49]: 619mV, [32]: 602mV, [42]: 580mV, pH-values: [49]: 4.5, [32]: 5.0, [42]: 5.4). As expected, throughout the 30°C-experiment the soil parameters of the depression-soil-suspension change, after a conditioning time of 5 days, to redox potentials of about 60mV. Concerning the plateau-soil-suspension it was expected that the soil parameters also change significantly, but the reaction stays behind the anticipations; achieved redox potentials are only 450 mV. No reaction occurred in the levee-soil-suspension. Only during 35°C the different study sites show significant variations in their response. As estimated the depression-soil-suspension answered immediately without conditioning time. At the end of the experiment a weak anoxic redox potential is reached (-47mV), but in comparison to the 30°C-test a constant value is not achieved. Despite the hypothesis the redox potential in the levee-soil-suspension decreases much deeper (40mV) than that of the plateau-soil-suspension (425mV). As assumed, pH-values changed according to the redox potentials. Due to their total content in soil the depression-soil-suspension shows highest the levee-soil-suspension lowest contents of dissolved trace metals at the end of all three experiments. The content of the easy soluble metals increases with the duration of the test in all suspensions. Regarding the characteristics of the several trace metals the response to changes of soil parameters differs. So As-concentration increases in the depression-soil-suspension during the 30°C-experiment whereas it remains indifferent in the levee- and plateau-soil-suspension. Noticeable and unexpected are the content of the dissolved trace metals in the levee-soil-suspension during and at the end of the 30°C and 35°C-experiment which are partly equal or even higher than in the other suspensions.

References
Effects of ivermectin in fish caused by dietary exposure to contaminated benthic organisms

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Within the Marie Curie Research and Training Network KEYBIOEFFECTS (MRTN-CT-2006-035695) one of the objectives is to get more insight in the transfer of model compounds and effects caused by this transfer along a simplified food chain (i.e. from sediment to fish via sediment dwelling organisms).

To achieve this goal, in this study the transfer and effects of the veterinary parasiticide ivermectin from the benthic oligochaete Lumbriculus variegatus to the zebrafish Danio rerio (Cyprinidae) are investigated.

The study is performed in two steps. In the first step L. variegatus contaminated with ivermectin will be prepared by exposing synchronised worms of similar length for 3 days in a water-only system. The bioconcentration factor of ivermectin in worms has been measured in a 6 day water-only exposure test with radio-labelled ivermectin, which makes it possible to estimate the ivermectin burden in the worms after exposure.

In the second step juvenile zebrafish in the growing phase are exposed to the ivermectin via dietary uptake. The fish will be fed individually with alive contaminated worms for a period of 28 days. After 14 and 28 days growth and mortality of the fish will be determined and fish-tissue and worm samples will be analysed for their internal concentration of ivermectin. Additionally, immunohistological biomarkers will be evaluated by the project partner, University of Bern (H.Segner).

Results of the studies are intended to be used for modelling the flux of compounds through trophic levels and effects caused by this flux based on the Dynamic Energy Budget (DEB) theory (Kooijman, 2000).

Since the oral toxicity study has been planned to start at the end of September, not all final results can be presented during the conference. Therefore the poster will present the test set-up and preliminary results.
Toxicity data on contaminated sediments of Elbe, Scheldt and Llobregat river basins

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Sediments are the ultimate sink for the numerous chemicals originating from urban, agricultural and industrial activities. However, sediment-associated chemicals are not biologically inactive. Sediment toxicity tests bring the crucial data for evaluation the risks of environmental pollutants to sediment ecosystems. The aim of this study was to evaluate toxicity of selected sediments from three European river basins Elbe, Scheldt and Llobregat. Whole sediment toxicity tests were conducted with Oligochaeta worms (*Lumbriculus variegatus*), midge larvae (*Chironomus riparius*), nematodes (*Caenorhabditis elegans*), mud snails (*Potamopyrgus antipodarum*) and zebra fish embryos (*Danio rerio*) in laboratory circumstances. Suspected or known polluted sediments were in general more toxic than the reference sediments. Further the different test species showed a quite specific sensitivity to different sediments. Therefore the applied toxicity test battery turned out to be more meaningful than the single toxicity tests. The relationship between toxicity and measured contaminant concentrations of sediments will be included as the comparison will be completed.
Setting water quality standards for trace metals - From a metals industry perspective

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In the framework of metal risk assessments and the implementation of the EU Water Framework Directive there is increasing interest in Europe and elsewhere in the world in taking into account the natural background of metals and their bioavailability when assessing metal Water Quality Standards (WQS) against water monitoring results.

Metals are naturally present in freshwaters, some of them being essential trace elements such as Zn and Cu. Due to differences in geochemistry, important variations in metal background concentrations are observed at different sites and scales. Guidance is provided on how to estimate and integrate natural background concentrations in setting WQS and compliance checking.

Bioavailability is crucial for the understanding of environmental metal toxicity. Aquatic bioavailability models (Biotic Ligand Models, BLMs) assessing chronic toxicity for metals have been developed and validated for Zn, Cu and Ni in the framework of EU risk assessments under the Existing Substances Regulation (793/93/EEC). BLMs for other metals are under development (Ag, Al, Mo, Pb and Co). BLMs are used to calculate WQS for a specific monitoring site or river basin. Such models require monitoring of dissolved metal concentrations as well as bioavailability parameters such as pH, DOC and hardness.

The integration of metal background levels and bioavailability considerations in WQS settings as well as a tiered approach to standard setting is advised. A tiered approach to metal compliance has already been implemented by some Member States (UK, the Netherlands), and has proven to offer an ecologically relevant approach.
Application of the Mappe model for risk assessment of surface water pollution: Case study for lindane

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The MAPPE model (Multimedia Assessment of Pollutant Pathways in Europe), is a GIS-based strategy for screening level modeling of the fate and transport of chemicals between different media over large regions. Maps of concentration at continental scale on the different media and more specifically, in surface water, can be derived from algorithms based on data of emissions, and environmental removal or transfer rates. Based on these, results are presented as maps of concentrations for different media. These maps, combined with ecosystem sensitivity maps based on transfer rates coefficients, can provide risk maps for aquatic ecosystems. The main advantage of the Mappe model is the simplicity of calculations, and consequently it’s higher speed on processing, that provide good results compared to other models.

In this case study we analyzed the fate of lindane in surface water. Lindane is a pesticide formerly used for agricultural purposes and used only in pharmaceutical products since the mid-1990s, that’s included on the list of priority substances of the WFD. Based on this, two different scenarios of emissions were considered, and consequently two different maps of concentration in surface water were derived. This way Mappe model appears as a useful tool with implications for policy/decisions.
The First-order reliability method of predicting BTEX contaminants in groundwater

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In risk assessment of new and existing hazardous substances, it is current practice to characterize risk using a deterministic quotient of the exposure concentration, or the dose, and a no-effect level. A sense of uncertainty is tackled by introducing worst-case assumptions in the methodology. However, risks often are overestimated in this method and it is difficult to answer “how dangerous”. In addition, when assessing the environmental risks, there are many uncertainties, which the deterministic methods can not deal with. Probability methods can provide a powerful method to deal with these problems. Variability and uncertainty in the input parameters are then described by distributions, and the output is similarly presented as a probability distribution. The first order reliability method (FORM method) has been extensively applied to the probabilistic modelling of engineering structure systems and, also to groundwater transport modelling. By applying the FORM method, the failure probability and sensitivity factors of contaminants reaching the groundwater, can quantified. In this presentation, we assume a scenario of jet fuel leakage in an airport, percolation to the unsaturated zone, and further spreading in groundwater. By combining groundwater transport model, which includes biodegradation, and the FORM approach, we get the failure probability of BTEX (exceeding the standards of the water quality) in space and time. Sensitivity analysis presents the importance of almost all parameters influence groundwater flowing and identifies the main sources of uncertainty. The application of fuzzy ranking is also included in order to make the uncertainty as explicitly as possible for the decision-makers.
Potential in-situ measures for risk reduction of contaminated sediments - Elbe river basin

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Sediment management under the EU Water Framework Directive will need a wider scope with in situ technologies embedded in a modern system of risk assessment and communication on the river basin scale. It seems that in Europe there is less regulatory acceptance to risk-based (rather than mass-based or chemical threshold-based) decisions. In the view of the size of the problems in Europe (e.g., ConSoil 2000 [1]), the guidance to innovative remedial measures and the experience from successful problem solutions in the United States cannot be ignored [2].

The most recent study on the assessment of risks from particle-bound contaminants in the Elbe River basin [3] includes an overview on potential sediment remediation measures on the catchment scale. In the poster presentation, emphasis will be given to in-situ technologies such as capping and monitored natural recovery (MNR), including guidance, e.g., test criteria, scenario definition, assessment concepts and methodologies.

References:
Redistribution of organic pollutants in river sediments and alluvial soils related to major floods

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Relatively frequent occurrence of floods in recent years raises concern about the material damages but also an important issue of contamination of the affected areas. The effects of major floods on levels and spatial and temporal distribution and dynamics of contamination with hydrophobic organic pollutants were examined from the continuous set of data for floodplain soils and sediments from a model industrial area in the Czech Republic where 100-year flood occurred in 1997. Top-layer sediment and soil samples from fourteen sites each repeatedly sampled during the period from 1996 till 2005 were characterized and analyzed for PCBs, OCPs, HCB and PAHs. Pollutants concentrations and relative distribution as well as organic carbon content in both sediment and floodplain soils were significantly affected by the flooding. Floods resulted in a decrease of all studied contaminants in sediments and significant rise of the PAH pollution in the flooded soils. There was unique and highly conserved PAH pattern in soils regardless of the floods and greater changes in PAH pattern in sediments related to floods. This documents longer contamination memory and consistent contamination pattern in soils, whereas sediments showed more dynamic changes responding strongly to the actual situation. The relative distribution of individual PAHs reflected combustion generated PAH profile. PAH levels in the river sediments rose again at the sites with continuous sources several years after floods. The study documents that sediments have the potential to function as secondary source of contamination for the aquatic ecosystem but also for the floodplain soils and other flooded areas and that floods can serve as a vector of PAHs contamination from sediments to soils.
Compounds with specific modes of action in river sediment samples - Temporal and spatial variations also in relation to floods

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The contamination of river sediments is frequently assessed from single time sampling reflecting just the actual situation. However, the sediments comprise a dynamic system especially in areas with occurrence of floods. Two year study has been conducted to reveal the spatial and temporal variability in the concentrations of commonly studied contaminants, but namely of the chemicals with specific modes of action in river sediments, in a model study area where 10-50 year flood occurred in 2006. The overall potencies of the sediment extracts for cytotoxicity, dioxin-like, (anti)estrogenic and (anti)androgenic potency were assessed along with analysis of known organic pollutants and heavy metals. Battery of in vitro bioassays with recombinant yeast and human cell systems was used to evaluate the extracts of sediments from four consequent sampling representing two different seasons. These bioassays show an integrative measure and identify the sites with continuously increased potencies for specific toxicity and also the potential effect of relatively frequent floods on their occurrence. Samples from industrial places as well as areas without any obvious major sources induced significant responses in assays for the dioxin-like and hormone-like activity. Thus the disperse nonpoint regional sources seem to contribute significantly to the pollution of the studied area.

The data matrix showed interrelation between dioxin-like activity, antiandrogenicity and content of organic carbon, silt and concentration of PAHs and PCBs, which documents significance of abiotic factors in accumulation of pollutants with specific mode of action.

The project was supported by GACR 525/05/P160.
Long-term monitoring of fluvisols in Zlín region, Czech Republic

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Region Zlín located in southeastern Moravia in the river basin of Morava and its tributary Dřevnice belongs to the most human affected areas in the Czech Republic because of intensive industry, agriculture and high density of towns, villages and traffic. It lays along Dřevnice river from rural countryside at east to town agglomeration at west which makes a gradient for different pollution sources. Flood plains and inundation terraces often represent accumulation sites for contaminants and serve as long-term integrative indication of the whole area contamination. Fluvisols developed here are important natural ecosystems closely related to river dynamics and soil microbial communities are their key functional component. About 30 fluvisols have been monitored since 1997 in 13 sampling campaigns. Microbiological, chemical, and physico-chemical characteristics have been measured. In addition, large floods occurred in this area in 1997 which allowed evaluation of fluvisol changes and long-term development after the floods in comparison with the initial state. The aims of the research were to analyze temporal and spatial variability of parameters, to find relations between parameters, to study the effects of floods, and to evaluate indication power of microbial parameters. The results showed that most parameters are highly variable in time and space. However, significant changes in all parameters were observed after the floods and explained in relation to sediment changes. Five years later recovery near to the initial state was apparent on some localities. Microbial parameters were not fully confirmed as sensitive indicators. This research has been supported by project INCHEMBIOL MSM0021622412.
The evaluation of ecological status of surface waters in the ARROW project: Implementation of the WFD in the Czech Republic

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The project ARROW is focused on the implementation of the EU Water Framework Directive in the monitoring of surface waters of the Czech Republic. The project incorporates both WFD EU demands and complex solution of sampling, biological, analytical and ICT problems. The integral part of the project is also the definition and calibration of reference network of sites. The project is based on long term development of this field in the Czech Republic and is related to parallel project of Ministry of the Environment of the Czech Republic connected to monitoring programs.

The evaluation of ecological state of surface waters according to WFD EU depends on the methodology of analysis of biological communities. There are two main approaches with its own disadvantages and advantages which are thus combined in our system i) type specific approach with prediction of comprehensive indices in abiotic types of localities and ii) site specific predictive modeling of biological communities composition. The final evaluation of ecological state in this project is based on multi-metric model which combine results from indices evaluation of biological communities (type specific approach) and predictive modeling of biological communities composition.

The presented approach is compatible with the EU Water Framework Directive and is implemented in the national-wide information system supported with GIS-oriented analytic software. The results are provided in www based interface together with reporting and visualization capabilities.

The project is supported by the Ministry of Environment of the Czech Republic (project ARROW) and by project INCHEMBIOL (MSM0021622412).
Indicative capability of the fish community as an indicator of river degradation

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The evaluation of resident fish communities is an important component of the ecological status assessment in aquatic habitats. Despite significant water quality improvement in the Czech Republic in the last decade, several important pollution sources in the Elbe River basin remain (Elbe River near Pardubice and its tributary Bílina stream). Ecological characteristics of fish communities were obtained by boat electrofishing in the Elbe River (width 40m, depth 2m) in four inter-weir sections in July 2005 and 2006. Sixteen sites in the longitudinal profile of the Bílina stream (length of 84.2km) were sampled by wading electrofishing in June 2006 and 2007. Relatively high fish species richness (22 species) was registered in two seasons in selected sites of the Elbe River. However generalists (bleak, roach, chub) form a majority of the fish community in all four sections. There was no significant difference in fish species richness between polluted and unpolluted sections. Channelization and regulation of the study stretch of the Elbe River seems to be the most important determinant of fish community diversity. Fish community in the Bílina stream is strongly influenced by anthropogenic impact. Presence of reservoir changed the structure of fish community and point source pollution affected water quality up to complete fish absence. Restoration of fish community in the longitudinal profile is rather slow. Fish community was a good biological indicator of chemical pollution only in case of a considerable strong adverse effect of chemical contaminants corresponding to the river size.
The variable impact of sediments on the pollution status of floodplain soils

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The poster presents the RAMWASS project and sediment related problems for an agricultural management.

Within the EU Project Risk Assessment and Management of the Water-Sediment-Soil System a DSS for an integrated management of fluvial ecosystems will be developed considering social, technical, economical and ecological aspects. In the Elbe river test site (section of Lower-Saxony between Schnackenburg and Lauenburg) particularly economical and ecological aspects are affected by the pollution situation of soils due to sedimentation of contaminated material over decades.

In consideration of legally binding ordinance for grasslands, mercury is of main importance. In addition critical enrichment with regard to EU ordinances for food- and feeding stuffs were observed for dioxins. The guiding values for dioxin contamination of agricultural land (5 - 40 ng/kg I-TEQ) are exceeded at about 90 % of all investigated sites (mean: 473 ng/kg I-TEQ).

The agricultural management of affected areas aims for e.g. shortening of graizing times and restriction of graizing to relatively low and moderate polluted sites. However, regionalisation of concentrations for contaminants with a distinct load history like mercury or dioxins in the Elbe river catchment is troubled by several problems: The concentrations of sites with high and low flooding frequencies are levelled out due to variable spatial and temporal sedimentation processes, due to bioturbation and due to human impacts like ploughing and levelling the landscape. Even if there exists no correlation between mercury and dioxins, similarities of load history can be found allowing to draw conclusions from mercury distribution in soils for the estimation of dioxin concentrations.
Assessment of ecotoxicological risks and hazard factors of contaminated sediments from European freshwater ecosystems

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The present study aims to assess ecotoxicological risks of contaminated sediments from European freshwater ecosystems. In order to achieve this, in vitro acute toxicity, genotoxic and dioxin-like responses were determined. Micronucleus assay using erythrocytes of Leuciscus cephalus collected in the field was applied in order to evaluate the mutagenicity in situ.

Sediments samples were collected from rivers Bílina and Elbe (Czech Republic) and Llobregat (Spain). Samples were extracted by Soxhlet extraction and in vitro tests were performed with RTL-W1 cells.

All tested samples had significant cytotoxic and genotoxic effects in exposed cells. EROD activity was significantly induced by extracts exposure of Bílina and Elbe samples, responding to wide range of concentrations of these samples, while no EROD activity was induced in cells exposed to Llobregat extract.

Micronucleus assay revealed significant mutagenicity for fish erythrocytes from all investigated locations, evidencing ecological relevance of in vitro results. Hence, none of the investigated locations could serve as an uncontaminated reference site, and consequently, additional comparisons to fish exposed in the lab (negative control) should be carried out.

In order to identify unknown pollutants causing toxicity, effect-directed analysis are going to be performed.

This study was supported by a personal grant of the Landesgraduiertenförderung Baden Württemberg to Paula Suares Rocha and is associated to the EU project MODELKEY
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(Status: 29 Oct 2007)
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General Information
Conference Venue

Leipziger KUBUS  
- the UFZ conference venue -  
Permoserstrasse 15  
04318 Leipzig  
phone: +49-341-235-2413  
fax: +49-341-235-2782  
e-mail: kubus-info@ufz.de

How to get from Leipzig Central Station to Leipziger KUBUS:
The Tram station is in front of the Central Station. From platform 2, please take Tram 13 to "Taucha" or Tram 3 to "Sommerfeld" until stop "Permoser Straße/Torgauer Straße", which takes about 15 min. Then cross "Torgauer Straße" and walk about 5 min along "Permoserstraße" or change to bus 90 to "Paunsdorf Center" until the next stop ("Leonhard-Frank-Straße"). This stop is at the Leipziger KUBUS.

Ticket machines are at (most) stops and accept coins or banknotes. You could also buy tickets from the bus/tram drivers. "Zone 110" (1 zone) is valid for the whole city of Leipzig.
Conference Dinner

Tuesday, 13 Nov 2007 from 20:00 h at Moritzbastei "Schwalbennest" (Swallow's Nest)

In the heart of Leipzig, nestled behind the Gewandhaus Concert Hall and City Skyscraper, you find the Moritzbastei, the only remaining part of the ancient city fortifications. The Moritzbastei was commissioned by Elector Moritz of Saxony in the 16th century and later named after him. It has survived 400 tumultuous years and it was used in many different ways, only to end as part of the rubble of the Second World War.

The impressive vaults of the "MB", as locals lovingly call the building, offer space for cultural events like rock and jazz concerts, play performances, readings as well as cosy places to enjoy a glass of wine or else, for example in one of the comfortable bars like the "Schwalbennest".

How to get there from UFZ by tram:
From stop "Permoser Straße/Torgauer Straße", please take Tram 3 or 13 to "Knauckkleeberg" until stop "Hauptbahnhof". From there, you could either walk (about 5-10 min.) or change to Tram 4, 7 or 15, all from platform 4 (1 stop, "Augustusplatz").

Moritzbastei
Universitätssstrasse 9
04109 Leipzig
Access to W-LAN at Leipziger KUBUS

The following information provides you with access to W-LAN at Leipziger KUBUS:

Name of network (SSID): Gastnetz
Authentication: WPA / PSK
Encoding type: TKIP
Key: WLANLogin