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Risk-Informed Management of European River Basins

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Aims and Scope

Since 1980, *The Handbook of Environmental Chemistry* has provided sound and solid knowledge about environmental topics from a chemical perspective. Presenting a wide spectrum of viewpoints and approaches, the series now covers topics such as local and global changes of natural environment and climate; anthropogenic impact on the environment; water, air and soil pollution; remediation and waste characterization; environmental contaminants; biogeochemistry; geoecology; chemical reactions and processes; chemical and biological transformations as well as physical transport of chemicals in the environment; or environmental modeling. A particular focus of the series lies on methodological advances in environmental analytical chemistry.

Series Preface

With remarkable vision, Prof. Otto Hutzinger initiated *The Handbook of Environmental Chemistry* in 1980 and became the founding Editor-in-Chief. At that time, environmental chemistry was an emerging field, aiming at a complete description of the Earth's environment, encompassing the physical, chemical, biological, and geological transformations of chemical substances occurring on a local as well as a global scale. Environmental chemistry was intended to provide an account of the impact of man's activities on the natural environment by describing observed changes.

While a considerable amount of knowledge has been accumulated over the last three decades, as reflected in the more than 70 volumes of *The Handbook of Environmental Chemistry*, there are still many scientific and policy challenges ahead due to the complexity and interdisciplinary nature of the field. The series will therefore continue to provide compilations of current knowledge. Contributions are written by leading experts with practical experience in their fields. *The Handbook of Environmental Chemistry* grows with the increases in our scientific understanding, and provides a valuable source not only for scientists but also for environmental managers and decision-makers. Today, the series covers a broad range of environmental topics from a chemical perspective, including methodological advances in environmental analytical chemistry.

In recent years, there has been a growing tendency to include subject matter of societal relevance in the broad view of environmental chemistry. Topics include life cycle analysis, environmental management, sustainable development, and socio-economic, legal and even political problems, among others. While these topics are of great importance for the development and acceptance of *The Handbook of Environmental Chemistry*, the publisher and Editors-in-Chief have decided to keep the handbook essentially a source of information on "hard sciences" with a particular emphasis on chemistry, but also covering biology, geology, hydrology and engineering as applied to environmental sciences.

The volumes of the series are written at an advanced level, addressing the needs of both researchers and graduate students, as well as of people outside the field of "pure" chemistry, including those in industry, business, government, research establishments, and public interest groups. It would be very satisfying to see these volumes used as a basis for graduate courses in environmental chemistry. With its high standards of scientific quality and clarity, *The Handbook of*

Environmental Chemistry provides a solid basis from which scientists can share their knowledge on the different aspects of environmental problems, presenting a wide spectrum of viewpoints and approaches.

The Handbook of Environmental Chemistry is available both in print and online via www.springerlink.com/content/110354/. Articles are published online as soon as they have been approved for publication. Authors, Volume Editors and Editors-in-Chief are rewarded by the broad acceptance of *The Handbook of Environmental Chemistry* by the scientific community, from whom suggestions for new topics to the Editors-in-Chief are always very welcome.

Damià Barceló
Andrey G. Kostianoy
Editors-in-Chief

Foreword

It is my pleasure to present you this book 'Risk-Informed Management of European River Basins'. The book provides an in-depth, state-of-the-art description and review of all main topics addressed by the RISKBASE project. The book aims at scientists, consultants and practitioners concerned about river basins. However, the primary target audience includes the drafters, supporting scientists and implementers of the European Water Framework Directive (WFD) River Basin Management Plans (RBMPs). RISKBASE aims to get its key findings noted by—and ideally integrated within—revisions of these RBMPs.

RISKBASE was funded by EC DG-Research under the 6th RTD Framework Programme as a Coordination Action project. RISKBASE aimed to review and synthesize the outcome of EC RTD Framework Programme projects, and other major initiatives, in relation to integrated risk-assessment-based management of the (ground)water–sediment–soil system at the river basin scale.

Leading European scientists and representatives of major European stakeholder groups executed this review and synthesis. They concluded that because of the increasing pressures from both anthropogenic and natural causes on environmental systems, it is no longer effective or efficient to deal with one issue at a time, since solving a singular problem often causes damaging impacts on other environmental compartments or in other places. We must consider the consequences of our actions on all parts of the environment in an integrated way and configure these actions to cope with an uncertain future. These challenges demand a different approach in order to achieve actual improvement of the ecological quality of our river basins and thus sustain the goods and services they provide for the well-being of society. Risk-informed management is this new approach. It involves the integrated application of three key principles: be well informed, manage adaptively and take a participatory approach. These three guiding (key) principles to risk-informed management of river basins are further explained in this book.

In this context, I would like to take the opportunity to congratulate the RISKBASE community for the great efforts undertaken to produce this scientific book. I hope this effort will pay off by raising awareness among policymakers, water

managers and practitioners as to the importance of the three guiding (key) principles for a successful risk-informed management of river basins.

The main RISKBASE recommendation to river basin managers is to develop a network of more well-designed, coordinated and monitored ‘learning catchments’ and to apply in these catchments the three key principles to risk-informed management. It is, in my opinion, a worthwhile investment with high ‘dividends’ for both managers and scientists that will promote a much more profound understanding and insight of the functioning of our river systems and will consequently enhance the cost-effectiveness of the management measures. Further recommendations—also to policymakers, research funders and scientists—are described in the book as well.

I hope you will find the findings of RISKBASE rather inspiring in implementing risk-informed river basin management and more specifically in the revisions of RBMPs in line with the WFD, a great challenge in its own right for the years to come.

Brussels, Belgium

Christos Fragakis
European Commission,
Directorate General Research and Innovation

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Introduction: The Need for Risk-Informed River Basin Management

Jos Brils, Damià Barceló, Winfried Blum, Werner Brack, Bob Harris, Dietmar Müller-Grabherr, Philippe Négrel, Vala Ragnarsdottir, Wim Salomons, Adriaan Slob, Thomas Track, Joop Vegter, and Jan E. Vermaat

Abstract As the pressures from both anthropogenic and natural causes on environmental systems increase, it is no longer effective or efficient to deal with one issue at a time, since solving a singular problem often causes damaging impacts on other environmental compartments or in other places. We must consider the consequences of our actions on all parts of the environment in an integrated way and configure these actions to cope with an uncertain future. These challenges demand a different

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approach in order to achieve actual improvement of the ecological quality of our river basins and thus sustain the goods and services they provide for the well-being of society. Risk-informed management is this new approach. It involves the integrated application of three key principles: be well informed, manage adaptively and take a participatory approach. This chapter introduces this risk-informed management approach as it was developed in the European Commission funded project RISKBASE and provides an introduction to rest of this book where the key principles are explained and underpinned in detail.

Keywords Adaptive management • Ecosystem services • Participatory approaches • Risk • Risk-informed management • River basins • Social/ecological system • Water Framework Directive (WFD) • Water–sediment–soil system

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1 River Ecosystems at Risk

River basins throughout the world are under pressure from socio-economic activities. These affect the integrity of river (basin) ecosystems and deplete available natural resources, in particular soil, sediment, surface water and groundwater. Ecosystems

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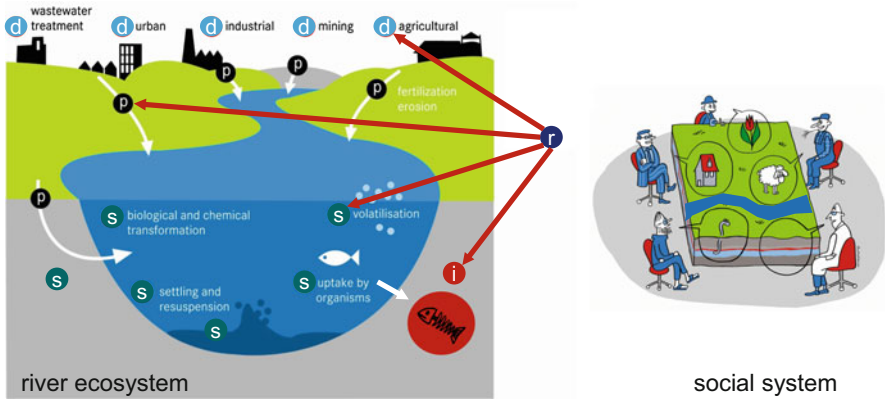


Fig. 1 The Drivers–Pressures–State–Impact–Response (DPSIR) framework

consist of complex and evolving interactions of living organisms and the abiotic environment in which they live, each making use of the production and waste products of others, creating dynamic cycles of nutrients and energy [1]. River basin ecosystems (short: river ecosystems) encompass river, lake, soil, sediment and groundwater ecosystems. The wide range of economic activities and the hydrological complexity of many river basins, both in terms of the functioning of the soil–sediment–water system and of the links between water quantity, quality and economic activities, result in multiple stress on aquatic ecosystems and make the integrated management of river basins both complex and highly challenging [2, 3] or even wicked [4]. A holistic and risk-informed approach is required considering several stressors at the same time [5, 6], including multiple contamination [7, 8] and taking into account uncertainty [9].

A useful conceptual framework for environmental processes and the links between human activities and their impact on the ecosystem functioning is provided by the Drivers–Pressures–State–Impact–Response (DPSIR) framework (Fig. 1). It treats the environmental management process as a feedback loop controlling a cycle consisting of five stages ([10]): Driving forces (D), Pressures (P), State (S), Impacts (I) and Responses (R). Economic activities (Driving forces) such as industry, agriculture and tourism lead to increasing pressures on the natural environment as these activities result in the use of natural resources and/or the release of emissions (accidental or controlled) of waste to (ground) water (e.g. [10]), soil (e.g. [11–13]) and sediment (e.g. [14, 15]).

The use of resources and/or the release of emissions will change the State of these environmental compartments both in quantity and/or quality. Sediment, water and soil resources are depleted (erosion) and/or they are loaded (contaminated) with hazardous substances originating from economic activities. Above a certain level of depletion and/or contamination, there will be significant further impacts (I) on the environment, e.g. loss of biodiversity; vulnerability to floods and landslides; decreased chemical and/or ecological water, soil or sediment quality; and/or a shortage of these resources. Several response (R) measures, implemented at any of the DPS or impact (I) phases,

could prevent this from happening or mitigate the impacts to a level deemed acceptable or tolerable by society. Moreover, the DPSIR framework also aims at bridging between science on one side and decision making and politics on the other side, bringing information from those establishing evidence and insight to those responsible for our common tomorrow.

In Europe, pollution from agriculture and hydraulic engineering (e.g. for navigation, water supply, hydroelectricity and flood control) are seen as the two main factors inhibiting the achievement of good ecological status of European river basins [16]. In addition, water is both an input to many industrial production processes and a sink for their pollutants and wastewater, while households also consume water and cause pollution with hazardous substances. Furthermore, other economic sectors such as navigation and hydroelectric power depend on minimum water levels for their functioning. Moreover, the river basin ecology is damaged by shortages of water and water pollution. Water prices are generally low and water-efficient technologies and practices are not yet fully implemented in many sectors. Additional factors such as population growth, economic growth and possible effects of climate change on river flow are expected to increase existing pressures on river basins [2]. In fact, global climate projections currently available anticipate crucial changes regarding extreme weather conditions, oceanic conditions and water regime of rivers. These changes will, in turn, severely modify basic riverine processes like currents and erosion, thus inducing important physical, geochemical and biological reactions.

2 River Basin Management-Related European Policy

Approaches to the management of water, water bodies and the wider environment across Europe were radically altered with the introduction of the European Water Framework Directive (WFD, [17]) and the subsidiary Groundwater Daughter Directive [18]. The aim of the WFD is to reach good chemical and good ecological status of all European waters. For the first time in the history of water management, the WFD implements integrated water resources management principles at a wide international scale (Europe) based on a common legal framework [19]. The WFD promotes the integrated management of water resources based on the natural geographical and hydrological unit of the river basin rather than administrative or political boundaries, whereas previous approaches would generally assess the chemical quality of a stretch of river or a water body such as a lake or an aquifer [20]. Consequently, in order to achieve ‘good ecological status’ and ‘good chemical status’ for all waters and develop strategies for achieving these goals through River Basin Management Plans (RBMPs), it is now necessary to assess the whole biophysical system, in particular surface water, groundwater, soil and sediment, in an integrated way [2].

In mirroring a continuing trend in environmental policy towards holistic planning approaches, a common European soil policy has been initiated by a Thematic Strategy on Soil Protection [21], which addresses the following four key pillars: (1) increasing public awareness, (2) closing knowledge gaps through research,

(3) integration of soil protection into other policies and (4) a framework legislation with protection and sustainable use of soil. Although the political debate on the draft Soil Framework Directive [22] has been blocked, soil protection must not necessarily be lagging behind water management in practice. It is important to note that most threats to soil identified in the Thematic Strategy on Soil Protection have strong relationships with the way water is managed. Therefore, when managing surface water, groundwater and sediment to achieve an ecologically satisfactory state, it is essential to manage and protect soil as well.

As climate change is anticipated to induce important physical, geochemical and biological reactions in river basins, adaptation towards these changes is one of the biggest challenges to date in river basin management. The WFD does not address climate change adaptation, but it is covered by the European Floods Directive [23]. Its aim is to reduce and manage the risks that floods pose to human health, the environment, cultural heritage and economic activity. The Directive required Member States to first carry out a preliminary assessment by 2011 to identify the river basins and associated coastal areas at risk of flooding. For such zones, they would then need to draw up flood risk maps by 2013 and establish flood risk management plans focused on prevention, protection and preparedness by 2015. The Directive applies to inland waters as well as all coastal waters across the whole territory of the EU. The Directive shall be carried out in coordination with the WFD, notably by flood risk management plans and RBMPs being coordinated, and through coordination of the public participation procedures in the preparation of these plans. Member States shall furthermore coordinate their flood risk management practices in shared river basins and shall, in solidarity, not undertake measures that would increase the flood risk in neighbouring countries. Member States shall take into consideration long-term developments, including climate change, as well as sustainable land use practices in the flood risk management cycle addressed in this Directive.¹

For a successful, joint implementation of the WFD and the Floods Directive and of future regulations on soil protection (land use), a holistic understanding is required of both river–sediment–soil–groundwater ecosystems and the social system (management, policy making, society and the economy) and how they all interrelate [24]. In addition, the future functioning of the system as a whole, including the impacts of changing climate conditions, land use practices and pollution needs to be understood and integrated in order to adequately establish RBMPs as well as groundwater quality control plans that will continue to be relevant throughout their lifespan (6 years in the case of RBMPs).

River Basin Management Plans (RBMPs) for most of Europe's river basins were submitted to the European Commission (EC) in Brussels in 2010 and 2011 and are now at an early stage of implementation.² The WFD is a strong stimulus in the European Union to effectively evaluate risks and impacts to deteriorated water

¹ See for more info: http://ec.europa.eu/environment/water/flood_risk/index.htm

² For detailed information see: http://ec.europa.eu/environment/water/water-framework/index_en.html

systems and identify appropriate remediation measures. However, it is clear that when the RBMPs are updated in 2015, the wide experience gained with the process of delivering the first RBMPs, *as well as new scientific knowledge, will have to be taken into account* [19].

The WFD conference organised in Lille in 2010 pointed out [19] that a lot of new, so far unused scientific knowledge is available—see also this book—to improve the effectiveness of selected measures or to inspire the introduction of new measures. It was universally agreed at the conference that scientific understanding is a very important basis for improving river basin management. It became clear that much of the presently available scientific knowledge has not yet found its way into the first RBMP. From the scientist's perspective, it could and should be used now to facilitate the implementation of the current version of the RBMPs and the drafting of the second generation of RBMPs. The science should be used to set priorities and define or refine the programme of measures in the RBMPs, rendering them more (cost-) effective. In fact, the WFD also requires this as, in annex IV it is stated as follows: *'The first update of the RBMP and all subsequent updates shall also include: . . . an assessment of the progress made towards the achievement of the environmental objectives, . . . and an explanation for any environmental objectives which have not been reached'* [17].

The most recent, for river basin management, important EU policy development to be mentioned is the adoption of a 'Blueprint to safeguard Europe's Water Resources' (short 'Blueprint', [25]). The Blueprint outlines actions that concentrate on better implementation of current water legislation, integration of water policy objectives into other policies and filling the gaps in particular as regards water quantity and efficiency. The objective is to ensure that a sufficient quantity of good quality water is available for people's needs, the economy and the environment throughout the EU. The Blueprint's time horizon is closely related to the EU's 2020 strategy [26] and, in particular, to the 2011 Resource Efficiency Roadmap [27], of which the Blueprint is the water milestone. However, the analysis underpinning the Blueprint covers a longer time span, up to 2050, and is expected to drive EU's water policy over the long term. In the Blueprint it is stated that knowledge gaps are one of the reasons for the currently insufficient levels of implementation and integration of European water (related) policy and that the knowledge base as well as governance should be improved.³

3 RISKBASE

In the last two decades, several European Commission (EC)-funded RTD Framework Programme (FP) projects (FP4–FP6) and other major research initiatives have addressed and promoted issues related to risk-informed management. Most of these initiatives focused on quality and management aspects of one specific compartment

³ See for more info: http://ec.europa.eu/environment/water/blueprint/index_en.htm

such as surface and groundwater (BRIDGE), soil and groundwater (INCORE, JOINT, WELCOME, CORONA, WATCH), sediment (SEDNET) or contaminated land (CARACAS, CLARINET, NICOLE). However, all these initiatives stressed the importance of an integrative approach for understanding and managing the multi-compartment ground- and surface–water–sediment–soil system at the river basin scale.

In order to improve the scientific basis for the integrated management of this soil–sediment–ground- and surface-water system, a number of EC RTD and Specific Support Action (SSA) initiatives have been funded, for example, HARMONICA in FP5, the Integrated Projects AQUATERRA and MODELKEY, the SSA EUROWET and the SSA SoilCritZone in FP6. Europe is also implementing the Environmental Technology Action Plan (ETAP) on the level of mitigation technologies, because Europe needs to increase investment in more innovative environmental protection methods while boosting competitiveness [28]. In relation to risk-informed management, the JOINT project concluded that *‘accompanying activities are necessary to help networking within the scientific and end-users communities, to evaluate the results and up-date the state of the art, and to focus on the main needs, strategies and future perspectives’*. Furthermore, *‘an integrated management approach is needed, which considers soil and waters as one system, interacting with other compartments of the environment and with the socioeconomic world through users and functions’* [29].

At the end of EC FP6, there was a clear need to bring all soil–water RTD efforts together through a CA as a stepping stone towards further policy development and implementation, whilst addressing the RTD topics highlighted in the FP7 programme. This CA was the FP6 project RISKBASE (acronym), full title: **Co-ordination Action on Risk-Based Management of River Basins**. The project started September 2006 and ended December 2009 (EC reference: GOCE 036938).

In RISKBASE, leading European scientists and representatives of major European stakeholder groups reviewed and synthesised the outcome of EC RTD Framework Programme projects, and other major initiatives, related to integrated, risk-based (at a latter stage adapted to risk-informed) management of the (ground-)water–sediment–soil system at the river basin scale. The synthesis led to the development of an integrated risk-informed management approach enabling the prevention and/or reduction of impacts caused by human activities on that system. RISKBASE aimed to deliver the following: (1) an overarching concept, generic approach and guiding principles to integrated, risk-informed management of river basins; (2) recommendations towards evolution and implementation of risk-informed management approaches in national and community policies and towards implementation in management; and (3) a proposal for the European research agenda related to risk-informed management.

To be able to achieve this, RISKBASE organised numerous workshops all along the project, which were dedicated to specific issues related to risk-informed management at the river basin scale. Major findings have been compiled in a special series of papers in the journal 'Integrated Environmental Assessment and Management' (IEAM).⁴ In addition, RISKBASE organised an annual General Assembly (GA) and made use of the already existing web-based information exchange system EUGRIS.⁵ The workshops and GA were open to all who were interested and willing to contribute to achieving the RISKBASE goals and objectives.

4 This Book: Guide to the Reader

4.1 The Book Structure: Four Sections (A–D)

The book is structured in three main (A–C, see Fig. 2) and one concluding section (D):

- Section A—Understanding River Ecosystems (three chapters)
- Section B—Anticipating Change (four chapters)
- Section C—Connecting to the Social System (five chapters)
- Section D—Synthesis and Recommendations (one chapter)

Section A starts addressing risk sources by presenting and discussing the soil-sediment-river connections and summarising of the pressures at the basin scale from their causes (natural and anthropogenic drivers) to their impacts on biophysical status [30]. Then [31], status and causal pathway assessment is described for the major stressors responsible for the deterioration of European water bodies. Stressor-specific abiotic and biotic indicators are presented that allow for a first indication of probable impacts, based on the assessment of available monitoring data. Subsequently, more advanced tools for site-specific confirmation of stressors at hand are discussed. Finally, local status assessments are put into the perspective of the risk for downstream stretches in order to be able to prioritise stressors and to select appropriate measures. Section A concludes [32] with providing a rationale for an extended monitoring programme, compared to the one required by the WFD, including a list of relevant parameters for an integrated a priori status assessment, based on single indicators. Furthermore, sophisticated tools are described that subsequently allow a statistical estimation of the magnitude of impairment and the likely relative importance of different stressors in a multiply stressed environment. The advantages and restrictions of these rather complicated analyses are discussed.

Section B starts [33] arguing that scenarios are well developed and suitable to explore the uncertain bandwidth of the future states of river basins, their rivers and occupant societies. It introduces a section of quantitative and qualitative scenario

⁴ See IEAM, 2009, Vol. 5, No 1, pp 2–126: <http://onlinelibrary.wiley.com/doi/10.1002/ieam.v5.1/issuetoc>

⁵ See <http://www.eugris.info>

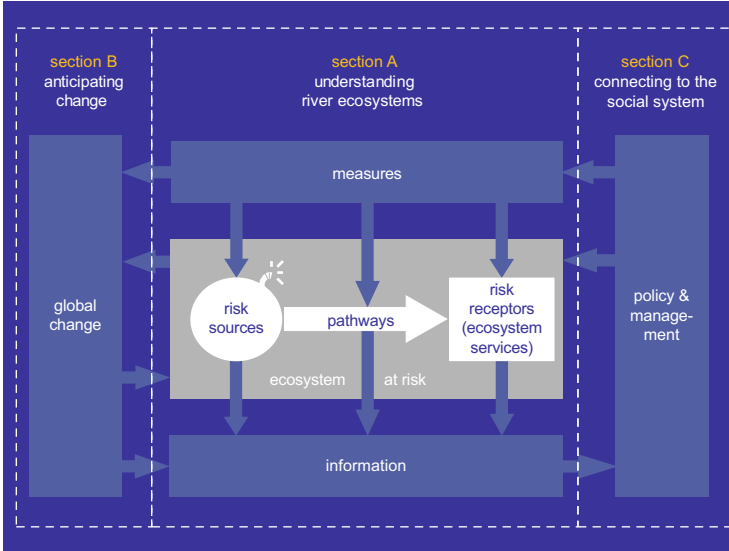


Fig. 2 Three main sections (A–C) as basis for the book structure

exercises and shows how scenario downscaling is carried out. Then it is described [34] how climate scenarios and socio-economic scenarios can be combined to assess future flood risks. Furthermore, several ways of dealing with uncertainty in a risk-informed approach are explained, and an example of drought risk assessment is described. The section continues [35] anticipating (via modelling) future land use changes and their hydrological impacts. This is done for ten different European river basins using the Intergovernmental Panel on Climate Change (IPCC) Special Report on Emission Scenarios (SRES). Section B concludes [36] by a qualitative outlining of downscaled articulations of the IPCC-SRES. This is done for a hypothetical Southern, Central and Northern European river basin and a time horizon set at 2030. The sensitivity of ecosystem state indicators is surveyed. It is assessed which drivers would be within the grasp of river basin management and a geographic comparison is made.

Section C starts [37] introducing the principles of integrated water resources management (IWRM), integrated river basin management, risk, risk governance (specifically the International Risk Governance Council's approach) and adaptive water management (AWM) and several cases are presented as illustration. Then it is described [38] how ecosystem services link to river basin management and how ecosystem services provide a coherent context to incorporate stakeholders and complex biophysical processes into a consistent, learning-based management scheme. The section continues by further describing [39] the process of involving stakeholders in river basin management and the important role therein of facilitative leadership. This is illustrated by three case studies. Connecting spatial planning to river basin management has great potential to improve that management [40] as it

involves the development and implementation of strategies and procedures to regulate land use and development in an attempt to manage and balance the numerous pressures placed upon land. The section concludes [41] by addressing the connection between science and policy and how this could be established through collaborative knowledge production. Also the boundaries to that are addressed and it is proposed to train people (knowledge brokers) in specific skills, like cooperation, who can facilitate the bridging between science and policy.

The book concludes with Section D ‘Synthesis and Recommendations’ containing one chapter [42]. That chapter aggregates all the findings of RISKBASE, i.e. synthesises all the information provided in the previous chapters, and a way forward (recommendations) is suggested for people concerned about river basins. Thus, it includes recommendations for river basin managers, for river basin management-related EU policy makers, for research funders as well as for scientists.

4.2 A Common Perspective as Basis for This Book

RISKBASE has its roots in the biophysical sciences domain. However, also in this domain, the several disciplines involved use of their own jargon/language and have diverging perspectives on river ecosystems and how they should best be managed. Thus, it was realised that to achieve its goal (see section above), it was essential to share common perspectives on:

- The primary target group: *science for whom?*
- The risk receptor: *risk to what?*
- The definition of risk sources: *risk of what?*
- A common language: *what language to use to enable communication within the RISKBASE community and with the target group?*
- And finally, a common perspective on risk-informed management: *what do we mean with risk-informed management?*

These common perspectives are explained hereafter and form the basis for this book.

4.2.1 Target Group

The primary target audience are the drafters and implementers of the WFD RBMPs. First RBMPs were published at the end of 2009, and so the RISKBASE findings and recommendations were not in time for this first generation of plans. This was realised already at the beginning of the project. Hence, RISKBASE aims to get its key findings noted for and hopefully also integrated in updates of the WFD RBMPs. Thus, it is anticipated that the first two ‘windows of opportunity’ for the RISKBASE messages will be the first and second update of the WFD RBMPs (see Fig. 3, original drawing courtesy of Damian Crilly, presented at RISKBASE Workshop in 2007, Vienna).

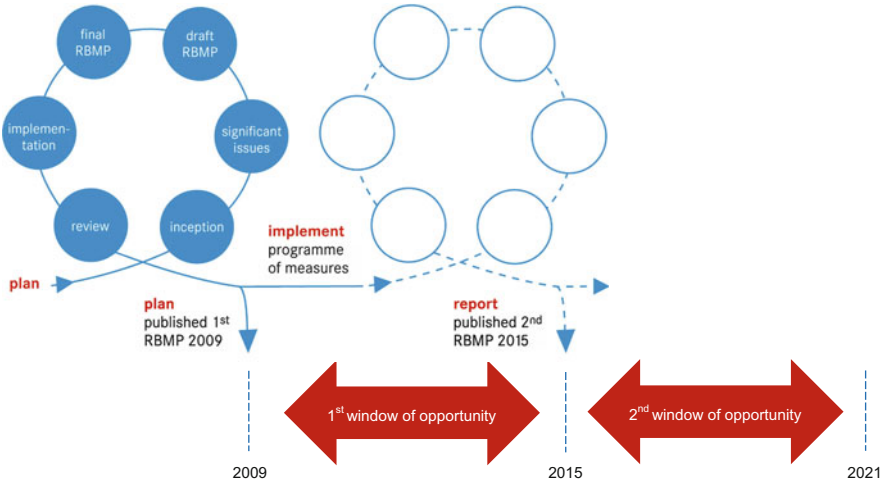


Fig. 3 Windows of opportunity for integration of the RISKBASE findings

4.2.2 Risk Receptor

Sustaining ecosystem services (see explanation further below under ‘common language’ and see [38], this volume) for human well-being is a main EU environmental policy objective [43]. However, RISKBASE selected ecosystem services provided by soil–sediment–(ground-)water ecosystems as its main risk receptor, because this concept fits closely with the highly dynamic nature of river basins (see Section A) and because it may provide a common language to facilitate stakeholder involvement (see this volume chapter by Brauman et al. [38], and chapter by van Maasackers et al. [39]). Thus, RISKBASE aimed to develop a risk-informed management approach enabling the sustainable use of these services.

4.2.3 Risk Sources

When talking about the protection of (risk) receptors—i.e. in the case of RISKBASE, the sustaining of ecosystem services—it is important to define the drivers and pressures that may pose a risk, i.e. the risk agent or risk source (see Fig. 2). In Section A (see chapter by Négrel et al. [30], and chapter by von der Ohe et al. [31]) the main sources and pathways of risk are described for European river basins.

4.2.4 Common Language

In the multidisciplinary and multi-stakeholder processes, the language that is being used is very important [44–46]. Slob et al. [44] investigated the hurdles for science policy interfacing and found that the capability to speak each other’s language is a very important factor for that (see [41], this volume). This observation is similar to

that of Totslandal et al. [47], who concluded that ‘each expert field, whether it is scientific or policy related, has its own set of jargon and expressions that may complicate communication’ [47]. Quevauviller et al. [48] also stressed the communication difficulties linked to different ‘jargon’ used in the different communities.

The words river basin, risk, ecosystem services, risk assessment, risk management and risk assessment-based (risk-informed) management mean different things to different people. RISKBASE has not tried to establish precise definitions of these words. However, consensus on a common perception of these words was developed. This consensus has been described by Brils et al. [3] and is summarised briefly below.

*River basin (or catchment)*⁶. RISKBASE follows the European WFD definition where ‘River basin means the area of land from which all surface runoff flows through a sequence of streams, rivers and, possibly, lakes into the sea at a single river mouth, estuary or delta’ [17]. Hence, RISKBASE perceives a river basin as a natural geographical and hydrological unit rather than looking at administrative or political boundaries. Although it may appear from Fig. 1 that there is a clear division between social systems and ecosystems, RISKBASE follows the opinion of the Resilience Alliance that such a clear division cannot be made and that in reality it is one, close-knit social/ecological system. Figure 1 combined with this Resilience Alliance viewpoint may result in a conceptual diagram as shown in Fig. 4. A lesson learned from contact with stakeholders participating in RISKBASE activities is that many stakeholders find the DPSIR framework still too complex, but seem to have no problems with the ‘simple’ Resilience Alliance social/ecological system presentation.

Risk (and Risk Terminology). Although risk has become a central concept in environmental policy and practice, this does not mean that it is easy to define it well (see also [37], this volume). The International Risk Governance Council [49] compiled in its ‘White Paper on Risk Governance’ an overview of risk terminology used by different organisations and/or found in different publications. In that white paper the IRGC uses the following term for risk: ‘An uncertain consequence of an event or an activity with respect to something that humans value (after [50]). Such consequences can be positive or negative, depending on the values that people associate with them’. Within RISKBASE risk is related to all agents or sources (see above and see this volume the chapter by Négrel et al. [30], and the chapter by von der Ohe et al. [31]) that may inhibit the achievement of river basin, the ecological quality objectives and, specifically, the sustainable use of ecosystem services.

Ecosystem Services. Societies (present and future generations) depend for their well-being on the goods and services provided by ecosystems ([51], see Fig. 4).

⁶Note: throughout this book—besides the term ‘river basin’—also the term ‘catchment’ is sometimes used. Both terms are regarded more or less synonym in this book. However, often a catchment is perceived smaller in scale than a river basin. A river basin or catchment can be subdivided in sub-basins or sub-catchments

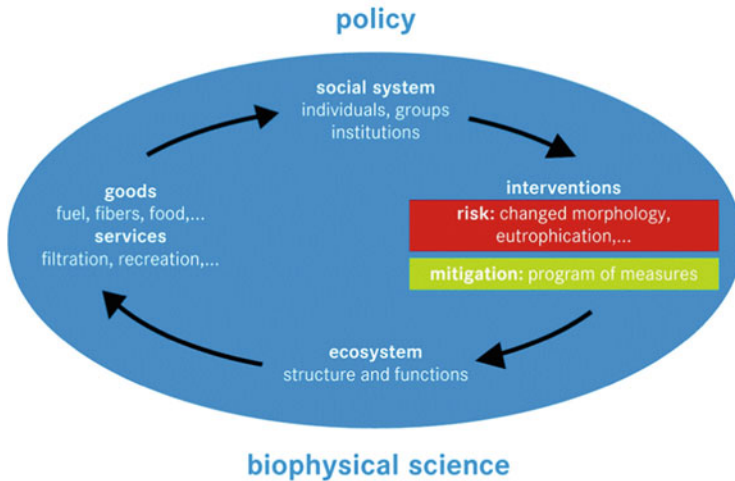


Fig. 4 The Resilience Alliance's (see: <http://www.resalliance.org>) social/ecological system from a RISKBASE perspective

Such goods comprise inter alia, (drinking) water, food, fuel, medicines and building materials. Services are the benefits that society obtains from ecosystems; for instance, provisional services (e.g. biodiversity, fishery, fertile soils for agriculture, water supply and protection against natural hazards), regenerative services (e.g. cycling of nutrients) and cleansing services (degradation of pollution) that nature provides. Also the well-being that nature imparts through enjoyment and recreation is such a service. Finally, biodiversity is seen as metaphor for the health of ecosystems [43, 51] and is thus an important measure for the effective functioning of natural processes (see further at the chapter by Brauman et al. [38], this volume).

Risk Assessment. Risk assessment (see [31], this volume) generally starts with a respective risk agent or source. It identifies both the potential damage scenarios and their probabilities and then models the potential consequences over time and space. From the early 1980s, there has been an ongoing debate in most developed countries between those who assess risk scientifically and technologically on the one hand and social scientists and psychologists on the other hand about the measurability or predictability of risk. The debate can, to a large extent, be characterised by two contrasting points of view: the scientific approach (formal risk assessment) versus the risks as perceived by individuals or the general public (intuitive risk assessment). The need for the better integration of these objective and subjective components is becoming increasingly recognised in almost all fields of risk assessment.

Risk Management. Risk management (see [37], this volume) relates to the policy-driven decisions about risks, which depend on specific time and economic conditions. In more general terms, risk management refers to the creation and evaluation of options for initiating or changing human activities or (natural and artificial) structures

with the objective to increase the net benefit to human society and prevent harm to humans and what they value. Managing of uncertainty—scientific, technical and subjective needs of society—is inherent part of risk management. Risk management is a permanent process for every society to organise anthropogenic activities and living with nature [52]. Thus, it is generally a major task to get a better understanding on how a social system understands ecosystems and reacts by different means of risk management, preventive approaches and policies. To analyse nature (see Section A) as an ecological system, or ecosystem, it is a prerequisite to understand political cultures, institutional arrangements, stakeholder involvement, risk perception, risk communication and how to translate societal values into science-related risk management systems (see Section C).

4.3 Risk-Informed Management

Risk-informed management of European river basins is the title of this book. But what was the common perspective in RISKBASE on that? RISKBASE concluded [53] that as the pressures from both anthropogenic and natural causes on environmental systems increase, it is no longer effective or efficient to deal with one issue at a time, since solving a singular problem often causes damaging impacts on other environmental compartments or in other places. We must consider the consequences of our actions on all parts of the environment in an integrated way and configure these actions to cope with an uncertain future. These challenges demand a different approach in order to achieve actual improvement of the ecological quality of our river basins and thus sustain the goods and services they provide for the well-being of society (Fig 4).

Risk-informed management is this new approach. It involves the integrated application of three key principles: be well informed, manage adaptively and take a participatory approach (Fig. 5).

A brief explanation of these principles and—in between brackets—guidance to where you can read much more on this in the book (see also Fig. 2):

1. *Be well informed:* A sound, evidence-based understanding of the functioning of the soil–sediment–water ecosystem (Section A) and of its interaction with the social system (Section A, B and above all Section C) is the basis to river basin management.
2. *Manage adaptively:* We have to learn by doing (Section C) as social/ecological systems are complex and dynamic and can respond in non-linear and unexpected ways. Furthermore, we are uncertain of the direction and magnitude of global (policy, climate, economy, land use) changes. Fortunately we can use scenarios (Section B) to chart several plausible trajectories of change leading to different societies inhabiting landscapes that are not as they look now.
3. *Take a participatory approach:* The involvement of stakeholders (Section C) will improve management, e.g. because they may bring in local knowledge.

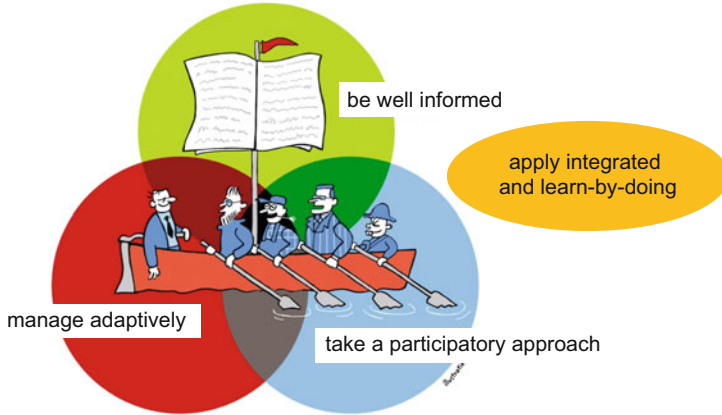


Fig. 5 Three guiding (key) principles to risk-informed management

This requires a common language. The developing ecosystem services approach (Section C) may provide that language.

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Section A
Understanding River Ecosystems

Soil–Sediment–River Connections: Catchment Processes Delivering Pressures to River Catchments

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Abstract This chapter presents and discusses the soil–sediment–river connections and summarises the pressures at the basin scale from their causes (natural and anthropogenic drivers) to their consequences (impacts on biophysical status). Nine important pressures on river basins are evidenced with respect to their temporal and spatial scale of occurrence and their impact on the river basin at the basin scale and concerns: erosion, sealing, compaction, hydromorphological changes, salinisation, contamination, changes in water quantity, acidification and reduction of soil organic matter. Each pressure can affect the biophysical status, and the simultaneous presence of pressures can have cumulative or compensatory impacts on biophysical status through propagation. Eight biophysical statuses were identified (concentration of chemicals, trophic status, biota status, buffering capacity, salinity, suspended matter and sediment, water level, morphology and pedology), and the pressures are described in this chapter in the sense of impacts on these biophysical status.

Keywords Anthropocene system • Biophysical status • Ecosystem • Natural system • Pressures • Sediments • Soils

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1 Introduction

Water covers three quarters of the earth surface and is omnipresent, in rivers, groundwater, lakes, seas and oceans. The amount of water on the planet does not change but continues to transform, through the various stages of steam, liquid water and ice, to perpetuate the ongoing cycle of water (Fig. 1).

Water is a ubiquitous reactant, occurring in all internal and external geodynamical cycles. Most of the geological phenomena involve water, whatever period or scale. The Earth's surface can be considered as a physical interface between the lithosphere, on the one hand, made of solid rock and soils encompassing all elements of the relief of the continents and the hydrosphere and atmosphere, on the other hand, that consist of gases and particles. Fluid-rock interaction mechanisms in the upper crust are important in earth sciences to understand the phenomena existing and decipher past events [1, 2]. Indeed, the fluids are present at all levels of the crust and play a fundamental role in physical and chemical processes that affect the Earth's crust, being an essential component of chemical reactions. Through their composition, they influence the nature of rock or sediment in which they are present and/or circulating. They are the main carrier of chemical elements mobility, of the mobility enabling chemical reactions and the transfer of materials at different scales.

On one hand, the water cycle governs the exchange between the major terrestrial reservoirs, the atmosphere, the hydrosphere and the geosphere (Fig. 1). This is the main process governing the formation of soils and sediment mobility. Two entities of the environment are jointly organising the flow of water, the hydrographic network and aquifers [3]. The first one is more or less dense and branched and can disperse or confine the waters with a variable carrying capacity according to the fluvial system (alluvial fan, braided channels, meandering, braided, delta plain, etc.). The second one is formed by the porous formations, open fractures and karst systems, and geological heterogeneities constrain the volumes and the flow of groundwater.

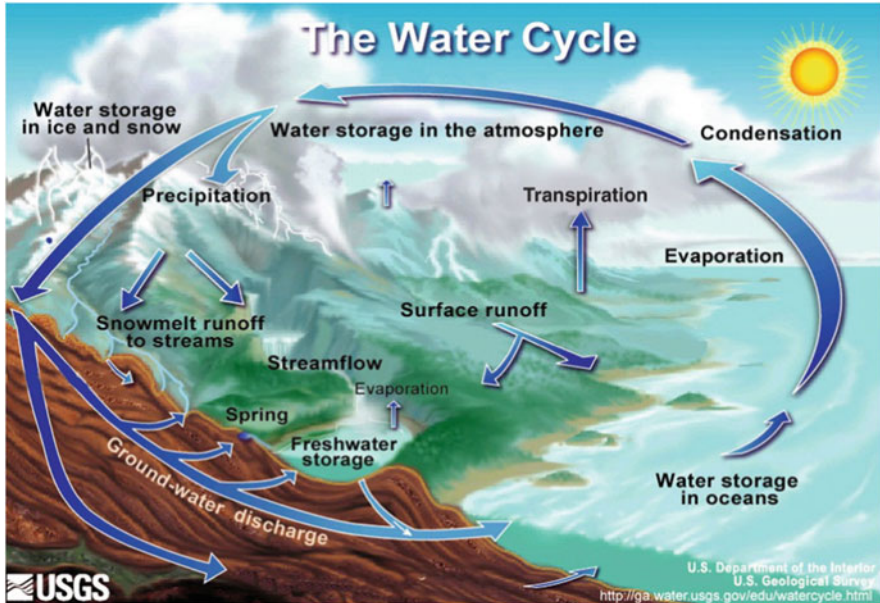


Fig. 1 Representation of the water cycle (Figure used by permission of USGS) (See: at <http://ga.water.usgs.gov/edu/watercycle.html>)

On the other hand, economic activities over the world induce pressure at the river catchment scale. These pressures may affect the chemical and ecological status along the water cycle (from rivers to groundwaters) and may impact the soil–sediment–water resources [4].

This chapter aims at providing a synthesis of soil–sediment–water connections at the catchment scale with regard to pressures that are delivered either by the natural (eco)system or the anthropogenic one. More particularly, this chapter is dedicated to assess and characterise causes of degradation of the river basin at large scale, processes involved and consequences on biophysical status. The selected focus in this chapter originates from a large panel experts consultation through workshops (Orléans, Leipzig, Berlin) managed during the course of the RISKBASE project.

2 The Natural System (Ecosystem)

2.1 *Continental Weathering and Erosion: The Soil Formation*

Chemical and mechanical destruction of rocks is the primary source of soil materials and solid load transported by rivers. This affects all rock types, igneous,¹

¹ Granite or basalt rock type formed through the cooling and solidification of magma or lava

metamorphic² and sedimentary.³ Rock fragments are produced by the mechanical action of the erosion processes, e.g. through natural physical agents like rocks cracking due to temperature changes, particularly important in rocks composed of different minerals with different expansion coefficients, or other processes like abrasion, wetting and drying, freezing and action of living organisms and plant roots. Primary minerals and rocks are split into fragments due to physical weathering, and the new environmental conditions, like the increase in surface area, favour chemical weathering (Fig. 2).

Water is the basis of chemical weathering of rocks thanks to the dipole behaviour of the water molecule due to its electrical charge. This property is due to the asymmetric covalent bond that links the hydrogen and oxygen atoms. The polar nature of the water molecule allows the establishment of hydrogen bonds between molecules in tetrahedral groups then inducing properties of the water structure such as the character of the solvent [6].

Chemical weathering involves the alteration of the chemical and mineralogical composition of the material being weathered. Different processes induce chemical weathering like hydrolysis, hydration, oxidation, reduction and carbonation.

Hydrolysis involves the destruction of the molecular structure (e.g. the primary minerals) and transformation into other molecular structures (e.g. the secondary minerals such as clays). Water molecules at the mineral surface dissociate into H^+ and OH^- and the mobile H^+ ions (H_3O^+) penetrate the crystal lattice, creating a charge imbalance, which causes cations (Ca^{2+} , Mg^{2+} , K^+ , Na^+) to diffuse out. When hydration occurs, i.e. the incorporation of water molecules to low-hydrated minerals (attachment of H^+ and OH^- ions to a reacted compound) increasing the volume of the mineral thus promotes the destruction of the rock. This is due to the property of ions to hydrate when H_2O is present and dissociate. Carbonation occurs when rain combines with carbon dioxide or an organic acid to form a weak carbonic acid which produces the solubilisation of limestone, dolomite and evaporite rocks under the action of CO_2 dissolved in water. Oxidation–reduction concerns minerals that contain Fe^{2+} (and Mn^{2+}). If there are oxidising environmental conditions, the Fe^{2+} is oxidised to Fe^{3+} that precipitates as an insoluble oxyhydroxide, usually either ferrihydrite or the stable mineral goethite. Mn^{2+} is oxidised to Mn^{3+} or Mn^{4+} partly inside the minerals, resulting in a positive charge balanced either by a loss of some oxidised iron and manganese ions or some cations dissociated from the mineral.

Climate is the most important factor affecting chemical weathering processes by controlling the rate of weathering that takes place through regulation of moisture and temperature.

Soil formation involves processes that develop layers or horizons in the soil profile. Five major factors influence soil formation, namely, the parent material, the climate, living organisms (especially native vegetation), the topography and the time. Parent material is the material from which a soil forms and generally consists of

² Arise from the transformation of existing rock types through metamorphism (initial rock subjected to high temperature and pressure causing profound physical and/or chemical change)

³ Formed by the processes of compaction and cementation of sediment over a long period of time

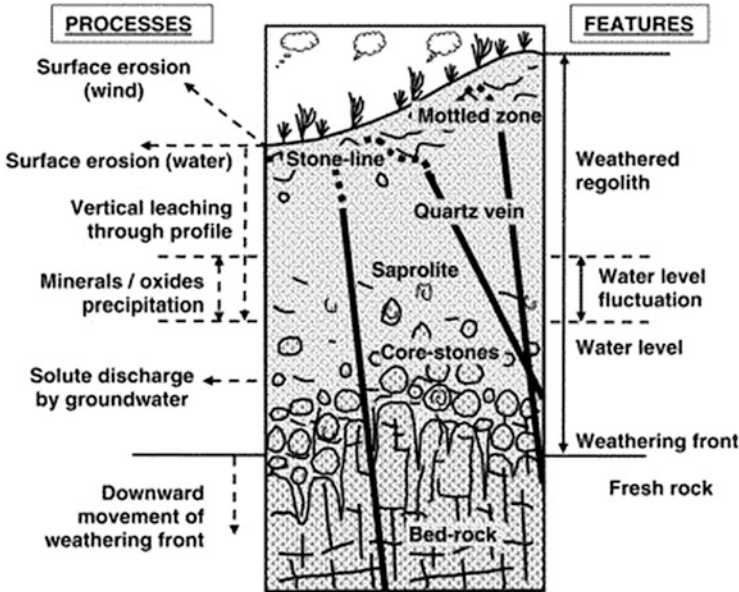


Fig. 2 Schematic sketch of the representative weathered profiles over Precambrian basement of South-West Nigeria, modified after Tijani et al. [5]

unconsolidated and more or less chemically weathered mineral or organic material. Resulting parent material is alluvium, lacustrine or marine if water is the mode of deposition. Water and ice result in glacio-fluvial, glacio-lacustrine or glacio-marine; ice results in till; wind results in eolian; and gravity results in colluvium.

Soils are named and classified on the basis of physical and chemical properties in their horizons (layers). Texture class is one of the first things to be determined for a soil examination. It is related to weathering and parent material as illustrated in the Fig. 3. The differences in horizons may be due to the differences in texture of their respective parent materials. Soil taxonomy identifies more than ten soil orders; more than 60 suborders are recognised at the next level of classification, about 300 great groups and more than 2,400 subgroups. They are labelled using a short-hand notation of letters and numbers [7]. They are described and classified by their colour, size, texture, structure, consistency, root quantity, pH, voids and boundary characteristics and if they have nodules or concretions [8].

The Critical Zone concept, defined as the Earth’s outer layer from vegetation canopy to the soil and groundwater that sustains human life, is a recent way of networking scientists investigating processes within the Critical Zone in the same large field site case studies (Critical Zone Exploration Network (CZEN) is a community of people and a network of field sites⁴). The Critical Zone encompasses the sequence of individual

⁴ See <http://www.czen.org/>

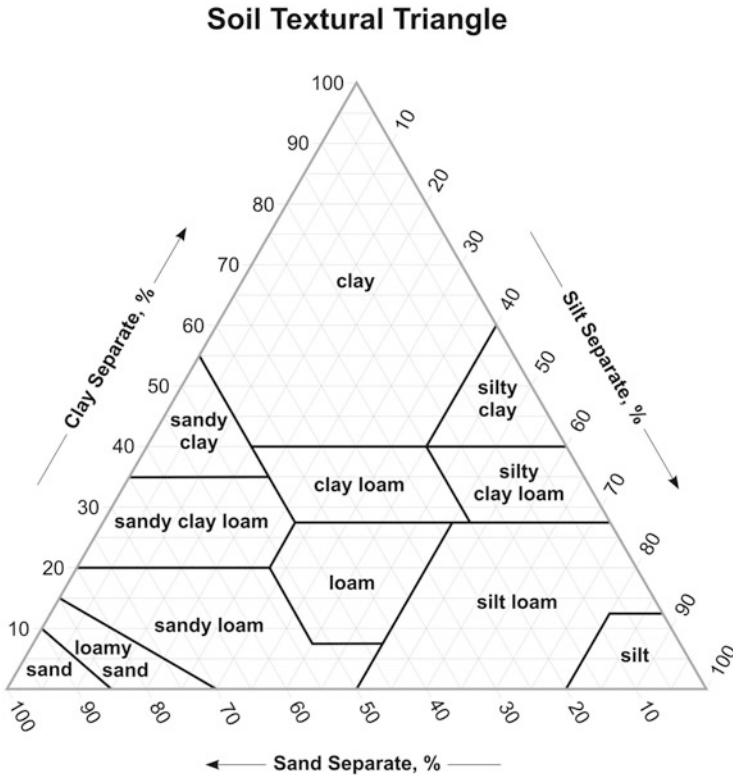


Fig. 3 Soil texture triangle classification, from USDA (free of permission) (See Soil Survey Staff, Natural Resources Conservation Service, United States Department of Agriculture. Soil Series Classification Database. Available online at <http://soils.usda.gov/technical/classification/scfile/index.html>. Accessed [15/09/2012].)

natural systems like plant communities, soil, weathering mantle, regolith, vadose zone and groundwater layer and represents, therefore, an extremely complex super system. The SoilCritZone⁵ project (FP6) aimed to bring together the fragmented European soil research community so that we can better understand what policy needs to be developed for the sustainability of European soils in the twenty-first century, in line with the aims of the European Soil Strategy. SoilCritZone aimed to mobilise the scientific and engineering community to develop a European research and innovation strategy on soil science. The SoilCritZone project concludes that the key research priorities identified (soil degradation, weathering, biodiversity, life cycle of soils) would benefit from well-instrumented field soil observatories enabling long-term studies in Europe. These long-term soil observatories need to be based on the main soil types of Europe;

⁵ See <http://sustainability.gly.bris.ac.uk/soilcritzone/>

soil types are influenced by the underlying geology, geography, climatology and tectonics of the area [9].

2.2 The Solid Transportation Over Continent

Rivers and aquifers play a central role in the natural system (ecosystem), from upstream to downstream; the river is the place of erosion, transit and deposition; and in this system, soil erosion is a natural process (Fig. 4). In the upstream part of the rivers, the natural load of a river appears as the product of the interaction, either chemical or mechanical, between runoff and rocks. For chemical erosion, this product is in dissolved form [6], while for mechanical erosion, it is in particulate form [10]. In a fluvial system and its watershed, the water flow is a continuum from precipitation to the ocean through runoff, evapotranspiration, infiltration, runoff into rivers, unsaturated zone and aquifer systems (Fig. 4).

At the earth surface, there are numerous streams flowing off the mountains and large rivers meandering in plains [11]. Each river basin is a functional group in which different reservoirs (e.g. rocks, water, vegetation, etc.) and interfaces are intimately connected. The geomorphological characteristics change gradually from the river source to its mouth, depending on the slope, the relative topography and the base level. In the most upstream part of the river system, erosion processes dominate, producing the majority of sediment load of the river system. The streams converge toward the transition zone between relief and flood plain. At this level the flow diverges and the transport capacity decreases and the coarser sediments accumulate to build up an alluvial fan. Beyond the alluvial fan, the coarser load (gravels, sand) has been deposited, and currents flow in the direction of the steepest slope. Toward downstream, the slope is decreasing, the currents lose their energy and then the finer particles (silts) are deposited. As the base of the channels will be less erosive, braided channels will gradually evolve to sinuous and meandering channels. In the river flood plains, silt and sand are deposited with some mud, creating fertile plains.

During high floods period, the currents coming out of channels, being more erosive, disperse the particles previously deposited in the flood plain. The solid products of chemical weathering and mechanical erosion transported by rivers join the ocean through the transition zone between continental and marine environments [12], i.e. estuaries and deltas. Although intermediates exist between the two types of mouth, usually when the river has a dominant influence, a delta is formed; if the marine influence is dominant, the river mouth has the morphology of an estuary. In estuaries, under the influence of tides and waves, fine particles are washed out and sand flats are left behind. When the sea level drops, these become workable soils (i.e. exploitable for agriculture) but poor in nutrients.

Considering the Europe's sediment budget, Owens and Batalla [13] gave a preliminary view to enhance the extent and budgets of European sediment, from erosion to delivery to oceans and seas, including partial deposition and remobilisation in catchments. The budget they gave was based on general estimates of soil erosion, sediment yield and sediment storage on river channel and floodplain environments

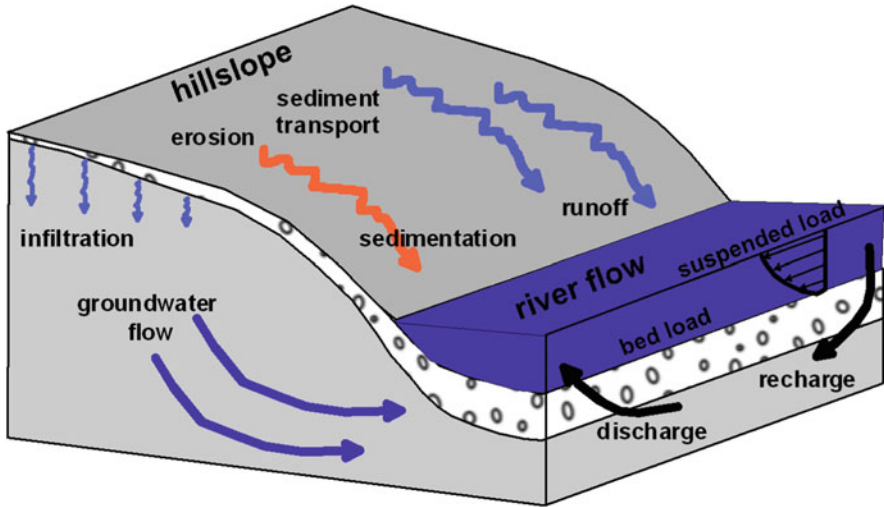


Fig. 4 Conceptual model of surface-subsurface water, sediment and flow bed transport

and in reservoirs, on one hand, and estimates of the sediment discharge to oceans and seas, on the other hand. The authors considered only the suspended sediment load (e.g. the particulate matter carried out in the water column) and did not take in consideration the bed load (e.g. coarse-grained sediment transported by saltation onto the river channel bottom).

The most recent erosion rates representing the sediment production of the European catchments were estimated by Owens and Batalla [13]. From humid environments in the northern and mid-southern Europe to semiarid regions in the southern Europe, they gave an estimate of around ca. $1,800 \times 10^6$ tonnes year⁻¹. Additionally, they estimated the total amount of sediment transported by rivers toward the lowermost land areas (deltas and estuaries, harbours, etc.) of around ca. 714×10^6 tonnes year⁻¹. Among the amount being eroded from rocks and soils and delivered to rivers, around 30 % would be deposited on floodplains, 20 % would be deposited in reservoirs and 10 % would be extracted for aggregate, leading to 40 % reaching the lowermost deposition zones and the coastal zone.

2.3 The Anthropocene System

It is well recognised that the human imprint on the environment, starting since the 1800s, has now become so large and active that it impacts the functioning of the Earth system. The term anthropocene, introduced about a decade ago [14], has only recently become widely, but informally, used in the global change research community [15, 16]. The anthropocene corresponds to the current epoch in which humans and our societies have become a global geophysical force. The development

of the anthropocene began around 1800 with the onset of industrialisation as analyses of air trapped in polar ice have shown an increase in the concentrations of carbon dioxide and methane. Since introduced, anthropocene was used in many studies of the environment, in the sediment investigations [17, 18], carbon and climate [19], land cover change [20] or rather more political reflections [21].

In the anthropocene, economic activities (drivers) lead to inputs, which are called pressures, to the ecosystem (see Fig. 1 in [22], this volume). The ecosystem adapts, resists, modifies and transforms these inputs into outputs, which correspond to stressors and lead to modification in the level of ecological service from the river basin [23] which is the ecosystem response. Thus, this leads to a segmentation of the ecosystem in a physical system on one hand and the aquatic life *sensu stricto* on the other hand, both being linked through stressors. Stressors are the results of socioeconomic activities as they are experienced by aquatic and terrestrial life and they reflect the unnatural inputs to the system that disturb the controlling factors, structure and processes in the ecosystem (see also Fig. 2 in [22], this volume). An example given by Negus et al. [24] demonstrates that a change in land use (indirect pressure/human activity) results in an increase in sediment load entering a stream (direct pressure), which affects stream turbidity (stressor) which in turn may lead to a changed ecological condition (ecosystem response).

A reasonable working hypothesis would be that good chemical status and good ecological status of water is a necessary condition for good performance of ecological services in a river basin. This implies that the assessment of status and impacts in aquatic systems in the river basin is relevant not only for the protection of aquatic life but also for wider issues concerning ecological services in the river basin. The health of the aquatic system is an important indicator of the health of the whole river basin system. The inventory of the drivers causing pressures and of the consequences of pressures on biophysical status showed that most drivers were correlated to most pressures and pressures had multiple impacts on biophysical status, reflecting the complexity of the system and its interconnections.

2.4 Relevant Pressures at the Basin Scale

Pressures or causes of degradation are defined as factors/phenomena which can reduce the resilience of the system, e.g. its resilience to cope with climate change, and whose occurrence or propagation has significant consequences on the biophysical status at large scale in the river basin. Two main types of drivers causing pressures on the biophysical status can be distinguished: the anthropogenic drivers, which are mainly associated to human land use and activities (agriculture, industrialisation, river basin network activities and other activities such as tourism), and the natural drivers, which include geological and climatic changes, inducing extreme events such as droughts, floods and natural hazards (Fig. 5). Pressures caused by natural drivers may be increased by human activities, but with regard to the ecosystem, it does not matter whether a pressure is man made, natural but intensified by human activities or just natural.

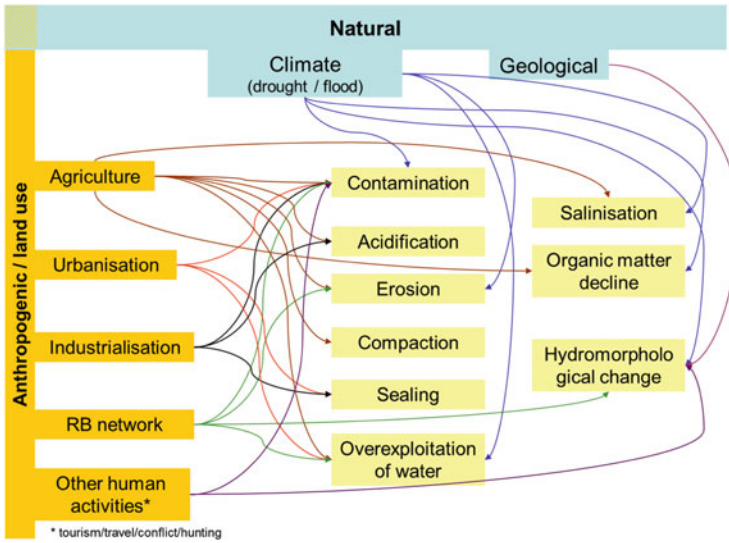


Fig. 5 Relevant pressures controlled by natural and anthropogenic drivers at the river basin (RB) scale and illustration of the complexity of the relations between drivers (natural and anthropogenic) and pressures

It is also important to consider the complexity of the links between human activities, natural changes and the pressures listed above as illustrated in Fig. 5. The pressures discussed in this chapter were selected with respect to their temporal and spatial scale of occurrence and their impact on the river basin at the basin scale. The pressures definition originates from a large panel experts consultation through two workshops organised in RISKBASE according to two main themes, i.e. “understanding processes and measurements of both soil/sediment and water compartments” and “monitoring and remediation soil/sediment and water compartments”. The pressures in the river basins thus identified may reflect the main concerns for the management of river basins, and these pressures are as follows: erosion, sealing, compaction, hydromorphological changes, salinisation, contamination, changes in water quantity, acidification and reduction of soil organic matter.

It is thus important to estimate that the links between human activities, natural changes and the pressures are complex and never can be all embracing. Figure 5 gives an illustration of such complexity. One pressure can have a direct effect on biophysical status, but several pressures can also interact to result in cumulative or compensatory effects on the biophysical status. The simultaneous presence of several pressures can originate from two different phenomena: (1) they appear individually in the basin due to specific human activities or (2) the presence of one pressure entailed to the presence of another or several other pressures.

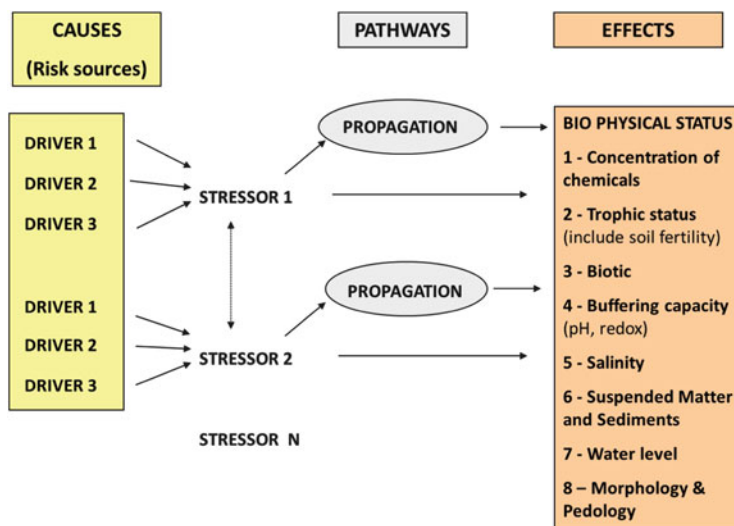


Fig. 6 Drivers and stressors (causes) via propagation (pathways) linked to their effect (notable as postulated changes) on the biophysical status

2.5 Pressures and Biophysical Status

Pressures, as causes of degradation, are directly related to specific drivers, pathways and biophysical status as summarised in Fig. 6. The pressures (e.g. erosion, sealing, compaction, hydromorphological changes, salinisation, contamination, changes in water quantity, acidification and reduction of soil organic matter) may have significant consequences on the biophysical status at large scale in the river basin (see Fig. 6). The propagation is defined as the potential for pressures to propagate at large scale through vector media, such as water, through dissolved and solid loads of surface water, groundwater and air. Each pressure affects the biophysical status and the simultaneous occurrence of different pressures, and their propagation can have cumulative or compensatory impacts on biophysical status.

Biophysical status has to be described, in terms of measurable characteristics, i.e. indicators. The description must follow two ways: it should be linked to the pressures and it should be relevant for the assessment of impacts on aquatic life. As stated earlier, the optimal description of biophysical status may vary as a function of the geological and ecological conditions in river basins (see Fig. 5). Thus, eight indicators may be used for the characterisation of the biophysical status:

- The chemicals contents: contents of inorganic and organic compounds in water, soil and sediment compartments.
- The trophic status: overall level of biological productivity in water and fertility in the soil. It refers to the concentrations of key nutrients (phosphorus, nitrogen, etc.) and planktonic algae (or phytoplankton).

- The biota status: all the interacting organisms living together in a specific habitat (biodiversity).
- The buffering capacity: ability of water or soil to resist and adapt to chemical property changes.
- The salinity: quantity of salt content in a soil (sediment), quantity of dissolved salts in water. Mainly refers to sodium chloride, but other salts occur.
- The suspended particulate matter and sediment: load that is carried for a considerable period of time in suspension. This particulate matter that can be transported by fluid flow and may be deposited as a layer of particles in form of bottom sediment.
- The water level: variation of level (or hydraulic potential) of water in groundwater (e.g. aquifers) or surface water.
- The morphology (a) and pedology (b): (a) the nature and history of the landforms on the surface and of the processes that create them and (b) the nature, properties, formation, distribution and function of soils.

Modification of the biophysical status is either a direct consequence of the pressure on a water body or an indirect consequence of the pressure due to its propagation at large scale in the system. Therefore, for each pressure, its occurrence at large scale and its potential propagation in the system are assessed. Consequences are also assessed with respect to biophysical status (and indicators) but do not give information on whether the consequence has a positive or a negative effect on the biophysical status, which was considered to be too dependent on the environmental settings. For example, erosion may lead to remobilisation of contamination in polluted areas with increasing contamination level downstream or may lead to buffering/reducing pollution acting as a trap for contamination present in water and/or dilution of contaminated sediments with clean eroded soil and sediments.

2.5.1 Erosion

Erosion is a process that can potentially affect all the river basin area through various erosion processes (e.g. hillslope erosion, channel bank erosion, landslide, gully and sediment dynamics at landscape scale). Chemical erosion is also a main feature especially on catchments which are heavily transformed by human activities such as reservoirs and dams (e.g. of the Ebro River basin accommodating more than 170 dams). The geological settings as well as river management control the chemistry of erosion (see above). Different causes of erosion processes can be incriminated, either natural (storm magnitude and frequency, topography, wildfires, etc.) or anthropogenic (sensitive land-use types, specific soil management regime and depopulation of mountain areas, etc.). The causes vary significantly with catchment type, relief, land use, soil type, etc. [25]. It is noteworthy that historical land management has changed the vulnerability and the intensity of the phenomenon [26]. Human settlements and the implementation of intensive agriculture have led to a shift from natural geological erosion to anthropogenic, accelerated erosion [27, 28].

An example can be given regarding the Ebro River basin. The use of a large set of chemical data covering the last 2 decades (1981–2004) allowed Négrel et al. [29] to estimate the export fluxes for dissolved and solid matter loads on the catchment scale of the Ebro River basin over a long time period of more than 20 years. They defined mean annual dissolved fluxes between $1,600 \times 10^3$ tonnes year⁻¹ upstream and $5,900 \times 10^3$ tonnes year⁻¹ close to the river outlet, leading to specific export rates around 70–90 tonnes km⁻² year⁻¹.

As more than 97 % of the Ebro drainage basin is affected by the presence of dams, the retention of sediment in dams and the decrease in water discharge have produced a drastic decline in the supply of river suspended matter: from $20\text{--}25 \times 10^6$ tonnes year⁻¹ in 1880 to 2.2×10^6 tonnes year⁻¹ in 1960 and 0.12×10^6 tonnes year⁻¹ in 1991 [30]. The long-term flux calculated by Négrel et al. [29] is around 0.1×10^6 tonnes year⁻¹, and the specific export rate is 1.2 tonnes km⁻² year⁻¹ near the outlet, close to that given by Guillén and Palanques [30] and Palanques et al. [31]. The specific export rate for suspended matter in the Ebro can be compared to other rivers in Western Europe. Manickam et al. [32] and Négrel and Grosbois [33] reported 8 tonnes km⁻² year⁻¹ for the Loire Basin. Meybeck and Ragu [12] reported 9 tonnes km⁻² year⁻¹ for the Seine River Basin and Roy et al. [34] reported 6 tonnes km⁻² year⁻¹ for the Seine at Paris. Higher values are given for the Rhine River Basin (15 tonnes km⁻² year⁻¹, [12]) and the Garonne River Basin (20 tonnes km⁻² year⁻¹, [35]). Geographically close to the Ebro Basin, Rovira et al. [36] reported a specific export rate for suspended matter of around 8.5 tonnes km⁻² year⁻¹ in the Tordera Basin (NE of the Ebro). Further north in France, Serrat et al. [37] reported between 40 and 103 and 40 tonnes km⁻² year⁻¹, respectively, for the Agly and Têt rivers. This places the Ebro River artificially with very low suspended matter fluxes and the associated weathering and erosion processes. The lack of relationship between suspended matter and discharge also supports the fact that dams play an important role in controlling suspended matter fluxes and regulating water flow. This is particularly of primary importance for the mouth of the river and its evolution. The evolution of the mouth of this river during the last 2,000 years, from an estuary to a delta, was interpreted as a process accelerated by the land management and the human impacts [13, 38].

In the European context, propagation of erosion is often significant at the basin scale. A recent assessment of the rate and spatial variation of soil erosion in Europe concluded that total sheet and rill erosion in Europe as covered by the CORINE database⁶ is estimated to be ca. 5.5×10^8 tonnes over a total surface area of ca. 4.46×10^8 ha. Thus, the average sheet and rill erosion rate for Europe is estimated to be slightly over 1 tonnes/ha/year, but this average rate may vary between 0.2 and 3.2 tonnes/ha/year on a per country basis. Mean rates per land-use class show very important variations, the value for arable land being around 3.6 tonnes/ha/year. The study also highlighted high spatial variability in soil erosion

⁶ See <http://www.eea.europa.eu/data-and-maps/data/corine-land-cover-2000-clc2000-seamless-vector-database-4>

rates, 70 % of the erosion being produced by 15 % of the area. It is certain that within these 15 % of the area, soil erosion and its associated consequences represent a major environmental issue [39]. Apart from the local degradation of the soil functions, erosion processes result in an increase in sediment transport as particulate matter and associated chemicals as dissolved load, which are transferred throughout the river basin. The consequences are hence very diverse from the loss of fertile topsoil, organic matter and soil biodiversity [40] to the occurrence of muddy floods [41], the siltation of reservoirs [42, 43], the pollution of continental and estuarial sediments and water bodies [44, 45] and eventually the modification of the landscape topography. Erosion is therefore a degradation that can potentially affect all the river basin area. In Europe, the main vector for particles (soil, sediment and associated chemicals) is surface water, including overland flow; notwithstanding in localised sandy areas, wind erosion can be significant.

However, erosion is not only a negative process as sediments are an important vital constituent of river basin development and functioning, moreover considering earth cycles. This occurred over the geologic times when large erosive periods and lithospheric dynamic shaped the landscape. The removal by erosion of large amounts of rock from a particular region, and its deposition elsewhere, impacts the load on the lower crust, generally causing tectonic or isostatic uplift. Therefore, the sediment behaviour concerns the erosion processes, transit and sediment deposition that punctuated the land-ocean continuum. Once going through the barrier between continent and ocean (e.g. estuaries or deltas), suspended solids and bed loads transported by rivers combine with particles only of marine origin and, through sedimentary and diagenetic processes, form sedimentary rocks. The sediment cycle continues through the passage into the oceanic trenches and subduction zones giving a large and continuous cycle.

2.5.2 Sealing

Sealing is generally described as the covering of soil by impermeable structures like buildings and infrastructure. Soil sealing is a complex phenomenon with various causes including the loss of structure due to the impact of rain or soil labouring, the dispersion of colloids and the compaction [46, 47]. Natural [48–50] and anthropogenic [51] sealing can be distinguished. The ever-increasing urbanisation in Europe will have effects at the river basin [52], especially on the water regime.

Like compaction (see Sect. 2.5.3), soil sealing increases surface runoff and prevents infiltration but does not propagate as such, although secondary effects may propagate to larger scales. Secondary effects may be the enhanced spreading of urban and agricultural pollution, and thus, sealing is now to be considered as an important emerging problem [53]. In situ consequences of sealing are dramatic and affect largely ecosystems. In a European context, propagation of the interactions of sealing on the other pressures (e.g. erosion) is often significant at the basin scale. Soil sealing reduces infiltration and increases runoff and its transport capacity. The water regime of the underlying soil is severely altered by sealing [54, 55]. In addition to the general decrease in soil moisture content, there is a

lowering of water tables in urban areas. Erosion is also significantly modified by soil sealing [56, 57]. The increase of impervious surfaces can have consequences also on the quality of water, due to the reduced filtering capacity of the soil [58, 59]. It also has been reported that pH and salinity may be affected by sealing [60]. The negative effects of sealing can also induce loss of plant production and natural habitats [51] and generally results in biodiversity loss [61].

2.5.3 Compaction

Compaction is mainly due to anthropogenic process and affects the first soil horizons. It is defined as: “the process by which the soil grains are rearranged to decrease void space and bring them into closer contact with one another, thereby increasing the bulk density” [62]. Soil compaction is a particularly important issue in the broad field of soil management as one of the causal agents of soil erosion, nutrient depletion and pollution [63] even if it does not propagate at catchment scale. Compaction also affects the mineralisation of soil organic carbon and nitrogen [64] as well as the concentration of carbon dioxide in the soil [65]. Soil compaction therefore adversely affects soil physical fertility [66], particularly storage and supply of water and nutrients, through increasing soil bulk density, decreasing porosity, increasing soil strength and decreasing soil water infiltration and water-holding capacity [67]. The main incriminated reason that exacerbates the compaction process is the increased size and use of machinery, particularly in remodelled landscape [68]. Secondary reasons include working the soil at the wrong soil water content [67], timber harvesting [69, 70], industrial activities such as the extraction of minerals [71] and the installation of underground pipelines [72]. Increasing soil porosity or decreasing bulk density reduces or eliminates soil compaction. Compaction does not propagate as there is no vector through the river system, and the amount of compacted soil must be very large before effects, such as reduced replenishment of aquifers, become significant at large scale. However, it should be noted that tramline wheel tracks are the principal culprits, often leading to overland flow both within field and off field; furthermore, the flow may redistribute sediment, nutrients and pesticides within the field and beyond [63], increasing connectivity at the field scale.

In the European context, the propagation of the interactions of compaction on the other pressures (e.g. erosion) is often significant at the basin scale. Soil compaction is estimated to be responsible for the degradation of an area of 33 million ha in Europe [73]. The extent of compacted soils may decrease if the shift from conventional agriculture toward simplified cultural techniques is pursued. It should however be noted that compaction, either due to natural (rainfall, plant roots, foot traffic of man, or animal) or artificial (mechanical operations) causes, is not reversible at a human time scale, and a very recent review by Nawaz et al. [74] summarises the advances in understanding, quantification and prediction of the effects of soil compaction. They highlighted the results of the experimental studies that confirm some parameters increase (soil strength, bulk density, volumetric water contents, field capacity) and

the decrease of others (total porosity, soil aeration, water infiltration rate, saturated hydraulic conductivity) in relation with soil compaction.

2.5.4 Hydromorphological Changes

Issues of water quality have generally dominated river research. However, also a better understanding of the ecological effects of a wide range of changes in physical habitat is needed and thus management and conservation being more crucial [75]. Rivers are increasingly exploited, regulated or otherwise modified through flood-defence engineering, impoundment, river restoration, climate change and the spread of alien species [76]. The EU Water Framework Directive [77] risk assessments, carried out in 2005, showed that hydromorphological pressures and impacts are one of the most important risk factors of failing to achieve WFD objectives. Hydromorphological integrity is central to conservation since it provides the template upon which all other ecological structures and functions are built [78]. Hydromorphological elements are hydrological regime, quantity and dynamics of water flows, connection to groundwater bodies, river continuity, morphological conditions, river depth and width variations, structure and substrate of the riverbed and structure of the riparian zone [79].

There exists a feedback between hydromorphology and ecology as they influence each other; the hydromorphology of the river will define the type of habitats that can settle which in turn may affect the sediment stability and release [80, 81]. The river hydromorphology has also a direct impact on the physical connectivity, which has not always been identified as a major issue in river basin management, though it plays an important role. Physical connectivity embraces horizontal and vertical connectivity and is scale dependent [82]. It depends on environmental and weather conditions (e.g. intensity of storm events).

Hydromorphological change is not a threat that propagates. The extent of the issue depends on infiltration capacity/land uses in the catchment. In a European context, propagation of the interactions of hydromorphological change on most of the other pressures is often significant at the basin scale and is a major issue. This is the main focus of the EC FP7 REFORM project (see further Sect. 2.7 in [83], this volume, also for more information on hydromorphological degradation).

2.5.5 Salinisation

Salinisation is the process that leads to an excessive increase of soluble salts in the soil solution compared to the natural levels generally observed. Salt-affected soils are defined as soils having salts in the solution phase and/or sodium ions on the cation exchange sites exceeding the specified limits. Major cations in these soils are sodium, calcium, magnesium and to a lesser extent potassium. The major anions are chloride, sulphate, bicarbonate, carbonate and nitrate. These soils are generally divided into three broad categories: saline, sodic and saline-sodic according to their electrical conductivity and sodium adsorption ratio [84].

Salinisation concern excess salts hinder the growth of crops by limiting their ability to take up water. Salinisation may occur naturally or because of conditions resulting from management practices. Soil salinity is mainly controlled by geological factors (presence of evaporates, e.g. soluble salts—such as sulphates of sodium, calcium and magnesium in the soil) and climatic factors (drought, low annual rainfall, high rate of evaporation). However, any change in the hydrologic cycle, such as results from draining or flooding areas (irrigation in intensive agriculture) changing the shape of the land's surface or increasing or reducing vegetative growth, can affect soil salinity [85, 86]. In the European Union, about 1 million hectares of soils are salt affected, mainly in the Mediterranean countries. Salinisation is also a major cause of desertification [87]. In Spain 3 % of the 3.5 million hectares of irrigated land are severely affected, reducing markedly its agricultural potential, while another 15 % are under serious risk [88, 89].

The water compartment is also affected by salinisation, and the main cause is the salt intrusion in coastal groundwater often occurring when coastal aquifers are over-exploited because of the demands of growing urbanisation, industry and agriculture. Over-exploitation of aquifers can also lead to leakage from deep salty aquifers. The phenomenon also occurs when the aquifer is near the surface, as the water table limits the depth to which salts are leached and moreover evaporation carries the salts back to the upper soil level. Climate change induces rise of the sea level that will accelerate the salinisation of coastal aquifers. Mine industries and urbanisation are also responsible of salinisation. While salinisation originating from mining decreases, salinisation in coastal areas drastically increases due to tourism extension and climate change. Artificial estuary deepening for navigation purposes greatly affects the estuary hydro-dynamics especially with regard to salt–water intrusion [90, 91].

Salinisation thus affects soil, surface- and groundwater quality. This threat can be of significance at the river basin scale, depending on the affected surface area and on the main lithologies of the catchment and the land use. Salinisation is not a threat that propagates through the whole river system with respect to the soil compartment. However, mineral dissolution with the release of chemical elements in solution (e.g. salts) and the propagation of elevated level of salts from the soil to the groundwater and the surface water can occur by the means of vectors such as infiltration water or drainage water network like on the Ebro River basin [29, 89]. Salts are also deposited by dust and precipitation. In dry regions, salts may accumulate leading to naturally saline soils. Salty waters can then affect large part of the river basin, both surface- and groundwater compartments. The groundwater compartment thus affected by salinisation can propagate through the natural water cycle up to the surface waters.

Salinisation, as a pressure, may thus affect several biophysical statuses. First, the concentration of chemicals and the trophic status will be impacted because of the change in the basic natural chemistry of the water (surface water and groundwater). Salinity increases till an improper water quality for users (drinking-water supply, irrigation, etc.) is reached. Salinisation affects the biological status with damages on plant growth leading to growing salt-tolerant crops. Ultimately, when crops are too strongly affected by the amount of salts, soil erosion will increase, affecting the suspended matter and sediments status.

In a European context, propagation of salinisation per se can be significant at the basin scale for specific areas: coastal zone, mining areas, specific lithologies like evaporates and dry areas. In Europe, Romanian, Hungarian and Greek soils are mainly affected; Spain and Italy also present areas under this specific pressure (European Commission—Joint Research Centre Institute for Environment and Sustainability⁷).

2.5.6 Contamination

Contamination is the introduction of hazardous substances into the soil and water system due to emissions and discharges originating from various sources often linked to human activities (see [83], this volume). Generally a distinction is made between point sources (direct inputs into the system at specific points) and diffuse sources (indirect inputs, for instance, atmospheric deposition of pollutants emitted to air or fertilisers inputs in agricultural areas).

The impact of anthropogenic contamination on the ecosystem can stem from multiple diffuse and point sources including mining wastes [92], spills, pesticide use and wastewater emissions with associated chemicals (e.g. pharmaceuticals, [93]). However, also high natural background concentrations may affect the system as shown for arsenic in groundwaters [94, 95] or selenium (Se) in different environments [96]. Investigations of environmental problems due to contamination generally aim at deciphering between natural and man-made origin of contaminants [97], but the way land is used may often influence the fate of contamination and its impacts. An interesting example is selenium (Se) contamination in the Lower Arkansas River, where the Se concentrations are linked with the NO₃ ones, revealing that the influence of the later, sourced and transported by fertilisation and irrigation, promotes the dissolution of Se from geologic and soil material [98].

Contaminated sites, the historical legacies like mine tailings dam failures [99], metal-contaminated sediments [100, 101], former industrial and military sites, are often addressed as a separate area of policy in most industrialised countries. Other contamination of special interest is diffuse pollution with persistent organic pollutants (POPs, [102]), from rural or industrial sites [103] but also in urban areas, that may form large impacted zones. Finally, as recently evidenced, emerging compounds like veterinary medicines and endocrine-disrupting chemicals (natural estrogens, pharmaceutical estrogens or industrial chemicals) are contaminants of increasing concern [104].

Main components in surficial Earth system, like N, P and S, have similarities with the contaminants issue but are often considered separately under the headings acidification for nitrogen (N) and sulphur (S) and eutrophication for N and phosphate (P). Phosphate and nitrogen levels in many soils are a legacy of historical fertilisation. Levels are “saturated” in many agricultural areas. The different causes of degradation

⁷ <http://eusoils.jrc.ec.europa.eu/library/themes/salinization/>

for contamination by such elements are the current and past fertiliser application [105], both mineral and organic; the natural organic matter mineralisation that can be affected by tillage management; the acidification by NO_x that has increased atmospheric N inputs; and finally the natural background because some areas have high phosphate levels (anoxic zones in sediments of marine origin, volcanic soils). In groundwater the reduction of nitrates due to the oxidation of pyrite can enhance the amounts of nickel and arsenic as trace elements in pyrites and subsequently in the surface water. Specifically for nutrients N and P, as they are very strong regulators of ecological quality, impacts are important in freshwaters (for P) and in marine systems (for N).

Contamination affects both the soil and the water (groundwater and surface water) compartment, and contaminants propagate differently at the basin scale according to their physico-chemical properties. They are transported adsorbed and absorbed by sediments, suspended matter and organic matter and/or transported in the dissolved phase [106]. Selected contaminants can be temporarily stored in soils and released according to local changes, e.g. modification of redox conditions. Contaminants (e.g. pesticides and their metabolites) and nutrients load stored in groundwater can cause problems for drinking-water supply in wells/springs and for recreation in mainly groundwater fed lakes and rivers even decades after the application of the contaminants or fertilisers [107–109].

The biophysical status can be affected by the concentration of chemicals through change in the basic natural chemistry of the water and increase the concentration of specific chemical elements (e.g. metals). The trophic status may also fluctuate as nutrient loads change, and this may lead to an increase in risk to human health (degradation of drinking-water quality, cumulative effects in the food chain) and risk to farm animals.

Numerous studies on the sources and pathways and consequences of the selected types of contaminants were conducted, especially through European Commission funded research projects like AquaTerra [110]. The large inputs from mining activities and from industrial megasites have been considered [111] for characterising the water flowpaths or to help environmental megasite managers in establishing an appropriate management approach for their megasites like in the WELCOME project [112] or AquaTerra [110]. Other examples are the contamination of the Danube and several of its tributaries by organic contaminants (phthalates like DEHP, PAH, tributyltin, nonylphenol) and heavy metals, including copper and nickel [113] or recent research on the links between natural background concentrations of contaminants and their concentrations in groundwater [114].

In a catchment context, contamination is significant at a large scale especially for diffuse pollution caused by urbanisation (increasing of sealing, wastewater), tourism and agriculture. Point source pollution shall currently be under local regulations (source control), and such contamination of soil, sediment, surface and groundwater is significant at the basin scale in specific context (megasites, region with industrial and mining histories). Historic contamination in sediments can be mobilised under extreme events due to extreme weather conditions, oceanographic conditions and water regime of rivers in relation with global climate changes. These changes may impact and modify drastically riverine processes like currents and erosion, thus

inducing important physical, geochemical and biological reactions. Floods and storm flows will likely lead to remobilisation of sediment and soil associated, historic contamination as well as to high-energy mass flows (eroded soil and remobilised sediment), and both will induce impact on river ecosystems [115].

Through comprehensive sampling and monitoring campaigns in diverse European river basins, using newly developed leading-edge approaches and methods, AquaTerra could show that many man-made chemical compounds can be found today ubiquitously, in all compartments, in air, in soil and sediments, in surface waters as well in groundwater.

2.5.7 Changes in Water Quantity

There are natural and anthropogenic drivers for changes in water quantity. The major natural driver is climate change causing region-specific reduction or increase of water quantity [116]. The spatial changes of water availability under climatic scenarios are generally evaluated on the basis of projections of general circulation models,⁸ giving the variation worldwide of the precipitation [117].

Anthropogenic drivers for changes in water quantity include the decrease of infiltration at recharge areas or intensive use of the water resource mainly in coastal, industrial and urban areas. Concerning anthropogenic drivers, the nationwide and local anthropogenic impact on the water quantity is much more considered by states or local authorities and may have high economical and societal costs evaluated by various UN programmes (e.g. World Water Assessment Program⁹). The water crisis is reflected well, at both local and national consideration, in Australia as south and north parts of the continent are affected by wet/dry seasonal climatic conditions. In particular, increasing climate variability in the south together with declining rainfall resulted in reduced surface- and groundwater supplies and increasing competition for water from irrigators, urban/domestic, industrial and mining users [118].

In Europe, the propagation of changes in water quantity can be significant at the basin scale. As water is itself a vector through surface- and groundwater, the water quantity change is implying variations of groundwater level and surface-water runoff. Notwithstanding, floods remain one of the major natural disasters at the earth surface. At the opposite, drought or water deficits have also major impact at the basin scale. Water-level changes are the more important impact of water quantity changes at the basin scale. Furthermore, various authors showed that large rainfall/runoff events are responsible for high proportions of erosion [119–122]. Concerning floods, rising of groundwater levels leads to large and persistent floods while surface-water runoff may be responsible for fast, intense and destructive flood ([123] and see several websites¹⁰). One dramatic example of a groundwater-induced flood is the

⁸ See <http://www.ipcc.ch/> (IPCC (Intergovernmental Panel of Climate Change), available at <http://www.ipcc.ch/>); IPCC (2009) Climate change and Water. IPCC reports, 7 Dec 2009, p 200

⁹ See <http://www.unesco.org/new/en/natural-sciences/environment/water/wwap/> (World Water Assessment Program. Available at <http://www.unesco.org/water/wwap/>)

¹⁰ See <http://www.acqwa.ch/>; <http://www.crue-eranet.net/>; <http://www.floodsite.net/>

repetitive and uncommonly severe floods which France was subject to in 2001. Early in the year, catchments in the north and west of the country were flooded for several weeks, and areas in eastern France were affected as early as December 2000. After the lowest discharge in August 1999, strong and continuous river-discharge increase was observed up to the damaging flood event that took place in the basin in April 2001, due to immense groundwater storage in the chalk aquifer [124, 125].

Mobilisation of contaminants and change in the basic natural chemistry of the water are also important effects of water quantity changes. Salinity and concentration of chemicals are diluted or concentrated with water-level changes [126]. Possible changes in flows between surface water and groundwater and within the hydrogeological systems may also induce changes in water contamination due to leakages and leaching of previously unsaturated zones. In coastal areas, over-pumping of groundwater resources or decrease of the recharge may lead to salinisation by raise of the seawater intrusion.

The water quantity and most particularly surface-water increase may change the morphology of the basin. The impact of floods on channel morphology is highly variable in intensity and response type as it depends on various factors, and there is therefore no correlation between the flood magnitude and its morphological impact [127, 128]. The role of the extreme events has been recognised as significant in conditioning channel river form [129]; episodic behaviour of extreme events has lasting effects on the river system. Increase of surface-water level also impacts the valley morphology and channel incision [130]. Water quantity changes may induce local variations in hydraulic connectivity. Surface-water and groundwater interrelationships may then change due to severe drought or flood episodes [124, 131, 132]. Longer lasting severe droughts have a strong impact on the drinking, irrigation and cooling water supply as reported from the river Po basin during the hot summer 2003 [133]. Under unfavourable geological circumstances, long-lasting severe droughts can cause even regional land subsidence [134].

2.5.8 Acidification

Acidification can be caused by industrial activities through emissions of sulphur oxide and nitrogen oxide in the atmosphere. This may impact the river catchment [135], for instance, via acid mine drainage [136, 137]. It can be the consequence of land-use change or practices such as afforestation with conifers [138], ammonia deposition resulting in nitrification [139] and H^+ release and absence of tillage resulting in pH depletion in the top few cm of the soils. In addition, it can also be due to wind erosion through desertification (i.e. there is no more buffer effect of the vegetation and thus weathering increases), CO_2 concentrations and landfill leachate. Acidification has occurred in many regions of Europe and North America where slowly weathering bedrock(s) and base-poor soils are unable to neutralise acidifying deposition of sulphur (S) and nitrogen (N) and is a great challenge to be solved since the 1980s [140]. Nowadays, sulphur oxide and nitrogen deposition generally tends to decrease

enabling the catchment to recover from acidification even though this can lead to dissolved organic carbon dissolution [141].

Acidification causes local and regional degradation. The media affected by acidification are the water, sediment and soil profile. Effects can be propagated by air or surface-water transport: however, in a European context, propagation of acidity is not significant at the basin scale except in specific areas (e.g. Poland and east part of Germany¹¹).

The biophysical status can be affected by acidification through increasing availability of heavy metals, aluminium and nutrients. Acidification also influences the hardness of the water and therefore the degradation of specific plant communities (see [83], this volume). The trophic status may fluctuate as soil fertility decreases. Finally, the buffering capacity is a function of the soil fertility decrease and the variation (generally the decrease) of carbonate in rivers. Therefore, the buffer capacity as well as the increase of acidity in the river may have an impact on the ecosystem [142].

2.5.9 Soil Organic Matter Reduction

Organic matter is defined as any material that is part of or originated from living organisms. It includes soil organic matter, plant residue, mulch, compost and other materials. More specifically, soil organic matter (SOM) is the total amount of organic matter present in the soil. It can be divided into three general pools: living biomass of microorganisms, fresh and partially decomposed residues (the active fraction) and the well-decomposed and highly stable organic material. Surface litter is generally not included as part of SOM [143, 144].

SOM is an important part of soil physical, chemical and biological fertility. With respect to biological fertility, dissolved organic carbon (DOC), which is a fraction of organic matter, is known as being an important food reservoir for the ecosystem [145–147]. Moreover, the importance of SOM as a sink for carbon storage has been recognised in climate change and CO₂ debate [148].

Loss of SOM is usually related to the loss of topsoil through erosion. Organic matter is also lost by microbial oxidation, in which soil microorganisms use organic matter as a food source during their normal metabolism. Intensive agriculture or the shift from forest plantation to cash crops can lead to a decrease in SOM levels that will affect the ability of the soils to fulfil their different economic, cultural and ecological functions. Cultivation practices affecting SOM levels include crop management and residue management (removal of straw vs. incorporation), tillage management (aeration increases mineralisation, no till can increase SOM [149]), deforestation, water abstraction/drainage of peat soils (compaction plus mineralisation) and increased intentional burning and wild fires [150].

¹¹ See <http://www.eurolimpacs.ucl.ac.uk/>

Decline of SOM is more frequently observed in southern Europe than in northern Europe where the amount of organic matter in soils may even increase. The amount of soil where organic matter has declined below certain critical limits is poorly understood. However, at the river basin scale, organic matter decline in soil is only relevant if the decline takes place in large parts of the basin. Decline of organic matter does not have a mean to propagate in the river basin.

Reduction in SOM levels, as a pressure, may thus affect several biophysical statuses. It may interfere with the concentration of chemicals as the reduction of SOM may reduce the contaminant retention and thus increase the transfer and release. DOC is also an efficient vector of metals (like nickel), of organic pollutant transport and an oxygen carrier. In the same way, the soil buffering capacity is modified because of a decreasing filtering capacity, reduction of Cation Exchange Capacity (CEC) and reduction of carbon stocks. The reduction in SOM impacts the trophic status because of reduction of the soil fertility (need to increase the fertilisers input). The biotic status is also affected by the reduction and change in the biodiversity that is enhanced by the reduction of the soil capacity to hold water and nutrient when SOM decline. A further consequence is change in water level in the soil because the water-holding capacity decreases. Finally, morphology and pedology can be modified due to the decreasing buffering capacity and reduction of water-holding capacity when SOM decrease.

3 Summary and Recommendations

Water is a ubiquitous reactant, occurring in all internal and external geodynamical cycles in the earth. Chemical and mechanical destruction of rocks remains the primary source of soil materials and solid load transported by rivers. Climate is the most important factor affecting chemical weathering processes by controlling the rate of weathering. In the natural system (e.g. without any anthropogenic influence), rivers and aquifers play a central role, being the place of erosion, transit and deposition. The human imprint on the environment has now become so large and active that it impacts the functioning of the Earth system, allowing to create the “anthropocene” that corresponds to the current epoch in which humans and our societies have become a global geophysical force. In the anthropocene, our economic activities lead to pressures to the ecosystem which, as response, adapts, resists, modifies and transforms these pressures into stressors (i.e. the sources of risk).

Based on discussions in the RISKBASE workshops (see Sect. 1 in [151], this volume), we provided in this chapter a synthesis of soil–sediment–water connections at the catchment scale with regard to pressures that are delivered either by the natural system or the anthropogenic one. Thus, nine important pressures on river basins are evidenced: erosion, sealing, compaction, hydromorphological changes, salinisation, contamination, changes in water quantity, acidification and reduction of soil organic matter. Each pressure can affect the biophysical status, and the simultaneous presence of pressures can have cumulative or compensatory impacts on biophysical status through propagation. Eight of such biophysical statuses were identified: concentration

of chemicals, trophic status, biota status, buffering capacity, salinity, suspended matter and sediment, water level and morphology and pedology.

Furthermore, the discussions in the RISKBASE workshops resulted in some recommendations. The first recommendation is that the water and soil compartments should be investigated together, especially for contamination. Management of contamination of biophysical system must include interactive surface- and groundwater and soil management as they are interconnected. Thus, there is a need to provide consistent concepts and applicable tools to identify relevant stressors in multiply stressed environments and to better understand the interaction of different stressors (see also [83, 152], this volume). This will help water managers who are frequently confronted with a multiple stressor situation of toxic pressure, organic pollution, eutrophication, adverse hydromorphological conditions, pathogens and invasive species to develop effective programmes of measures for improving the status of aquatic systems. On the other hand, there is an urgent need to provide tools for the identification of emerging pollutants that pose a risk to aquatic ecosystems (see also [83], this volume). The very large and increasing amount of chemicals and environmental transformation products, the general tendency toward higher polarity and complexity of emerging compounds and the increasing relevance of compounds with highly specific modes of action (e.g. pharmaceuticals) demand more innovative approaches and techniques for isolation, structure elucidation and effects assessment. For this, the recent development of CSIA (compound-specific stable isotope analysis; see [153] and references therein) for organic contaminants provides valuable and sometimes unique information for deciphering their origin and studying their degradation processes in the environment.

Finally, as most management is based on knowledge acquired at a certain time, there is a need of prospective studies to meet the potential requirement of the system to adapt to futures changes (see also Section B). Protecting and managing surface and groundwater resources is a critical action for safeguarding drinking-water supplies. However, organic contaminants, which derive from industry, oil spills, improper disposal and/or leaking storage tanks, landfill leachates, household use, motor-vehicle emissions, agricultural fertilisers and pesticides, pose a threat to soil and freshwater resources. For this, the application of high-grade analytical tools (see also [83], this volume) offers a new potential way to distinguish the different sources within contamination plumes and even to estimate the different source inputs at the catchment scale. In addition, the possibility of such high-grade analytical tools for evaluating *in situ* transformation reactions is of fundamental importance for studying the fate of organic pollutants. This is particularly crucial for designing remediation strategies that rely on monitoring natural and/or engineered attenuation of organic contaminants in ecosystems.

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Abstract The European Water Framework Directive (WFD) requires a status assessment of all water bodies. If that status is deteriorated, the WFD urges the identification of its potential causes in order to be able to suggest appropriate management measures. The instrument of investigative monitoring allows for such identification, provided that appropriate tools are available to link the observed effects to causative stressors, while unravelling confounding factors. In this chapter, the state of the art of status and causal pathway assessment is described for the major stressors responsible for the deterioration of European water bodies, i.e. toxicity, acidification, salinisation, eutrophication and oxygen depletion, parasites and pathogens, invasive alien species, hydromorphological degradation, changing water levels as well as sediments and suspended matter. For each stressor, an extensive description of the potential effects on the ecological status is given. Secondly, stressor-specific abiotic and biotic indicators are described that allow for a first indication of probable causes, based on the assessment of available monitoring data. Subsequently, more advanced tools for site-specific confirmation of stressors at hand are discussed. Finally, the local status assessments are put into the perspective of the risk for downstream stretches in order to be able to prioritise stressors and to be able to select appropriate measures for mitigation of the risks resulting from these stressors.

Keywords Biotic metric • Causal pathway • Impact • Status assessment • Stressor

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1 Introduction

Freshwater, which supports many important and varied ecosystems [1], is one of the most valuable resources on earth. Its sustainable use is a great challenge for human societies: Many rivers, wetlands and lakes are facing the impact of a number of anthropogenic and natural stressors, including the effluents and changes from municipal, industrial, agricultural and extractive land use, and are subject to floods and droughts (see [2], this volume). It is obvious that conditions in the aquatic environment do not only depend on processes within the water body, but very much on land use as a cause of pollution, erosion, water quantity related problems and many others. Hence, an integrated management of the whole water–groundwater–sediment–soil system is critical to achieve and sustain good ecological and chemical status in rivers and lakes as required by the European Union’s (EU) Water Framework Directive (WFD) as a major objective for river basin management plans (RBMP) [3]. Aquatic systems respond sensitively to human activities in the whole catchment (see [4], this volume) and even to global change (see Section B). The assessment of stressors resulting from human activities is the focus of this chapter, discussing the state-of-the-art and promising scientific developments that contribute to the understanding of river basins as integrated systems in order to support successful river basin management. An integrated status and causal pathway assessment will help to identify the predominant stressors of such systems and to select effective measures. This chapter will provide an overview of relevant stressors together with specific metrics to identify and assess major impacts on freshwater ecosystems. Diagnostic tools for a more comprehensive status assessment with regard to multiple stressors are provided in [5], this volume.

Freshwater ecosystems are often subject to the impacts of multiple stressors: i.e. due to erosion, sealing, compaction, hydromorphological changes, salinisation, eutrophication, toxic chemicals, changes in water quantity, acidification, invasive alien species and changes in sediment quantity and quality [1, 6]. To manage these ecosystems in order to achieve good status and to safeguard ecological services in a sustainable way, there is a need to understand how these ecosystems function and how

this functioning is affected by various stressors caused by socioeconomic activities (see [4], this volume). Although the focus of this chapter is on stressors caused by anthropogenic activities, it should be noted that change and thus stress are natural parts of ecosystems and that even completely unaltered ecosystems will experience stress and changes in ecosystem services. It should also be noted that some forms of aquatic life can also be considered stressors: Invasive alien species or pathogens are examples. Current knowledge about the relationships between the status and the well-being of aquatic communities on the one hand and the potential or actual performance of ecological services on the other hand is rather limited. However, a reasonable working hypothesis is that good chemical status and good ecological status of water are necessary preconditions for good performance of ecological services. This working hypothesis implies that achieving good ecological status is mandatory not only for the protection of aquatic life but also for wider issues concerning ecological services.

Many anthropogenic and natural drivers propagate in a river basin and may have significant effects at various scales, from small to large (see [4], this volume). Changes in ecological status may be caused locally, e.g. by effluents from an individual wastewater treatment plant which may diminish at larger scales or by upstream and diffuse sources such as large volumes of contaminated sediments that may affect sensitive downstream riverine and coastal ecosystems.

In this chapter, the focus is on how chemical, biological and physical stressors affect aquatic life (ecological status) and how key stressors may be identified, even in the presence of confounding factors (see Sect. 2.1 for definition). The WFD requires at least good ecological status of aquatic water bodies, allowing only slight changes relative to reference conditions. Thus, all stressors that lead to significant alterations in community structure are defined as negative in the terms of the Directive, even if these new community structures would be better adapted to changed conditions (e.g. following climate change). This definition is also related to the problem of “changing baselines” and, hence, changed reference conditions. The concept is currently under discussion, which may also impact on the ecological status assessment.

In general, ecological status assessment is based on intercalibrated metrics for different trophic levels (also called biological quality elements), including phytoplankton, other aquatic flora, macroinvertebrates and fish, and indicators of supporting hydromorphological, chemical and physicochemical quality elements. Three types of monitoring for surface waters are described in the WFD: surveillance, operational and investigative monitoring.

Surveillance monitoring programmes are designed to supplement and validate impact assessment, to efficiently and effectively design future monitoring programmes, to assess long-term changes in natural conditions and to assess long-term changes resulting from widespread anthropogenic activity. Operational monitoring should be undertaken to establish those bodies identified as being at risk of failing to meet their environmental objectives and to assess any changes in the status of such bodies resulting from the programme of measures. Investigative monitoring should be carried out in cases where the reason for exceedance of quality criteria is unknown and to ascertain the magnitude and impacts of accidental pollution. Thus, causal pathway assessment is typically a major element of investigative monitoring.

While monitoring is performed at specific sites and thus helps to assess the ecological status at the local scale, for river basin management, the potential of stressor propagation in the river basin is of equal importance and will be given a specific focus in the following discussions of individual stressors.

2 Stressor Impact on the Site Scale and Its Propagation at the Basin Scale

By requiring surveillance monitoring networks, the WFD aims to enable integrated, large-scale impact evaluation and management in river basins [3]. However, even though stressors may propagate in river systems and have large-scale impacts, most effects will be determined at the site scale. Hence, the assessment of river basin status is also based on the assessment of one or few sites. Although seemingly trivial, this fact may have significant implications on the interpretation of status assessments in water bodies represented by these sites and on cause identification. Even if a stressor propagates through the river systems and impacts sites far downstream, the increasing distance from the source complicates causal pathway assessment significantly. Only an extensive surveillance monitoring of all potentially relevant parameters may help to derive realistic hypotheses on predominant stressors as a basis for subsequent investigative monitoring and management.

For water managers, there are particularly two situations that may trigger investigative monitoring and impact assessment. The first one is the exceedance of environmental quality criteria (insufficient chemical status) without evidence of insufficient ecological status. In this case, higher-tier risk assessment may help to prioritise sites, risks and measures with respect to an effective and cost-efficient management. The second situation occurs if there are indications of an insufficient ecological status while the underlying pressures are unknown or at least unclear. In both situations, the starting point will be surveillance monitoring as required by the WFD, providing data on ecological and chemical status (Fig. 1, upper left box). In both cases, state-of-the-art models, indices and statistical analyses could be applied as a first step to connect status to possible underlying causes, before applying the one-out all-out principles based on biological metrics, as required by the WFD. A pressure analysis based on single biological metrics might run the risk of serious misinterpretations (e.g. no fish in a naturally fish-free lake does not necessarily indicate a poor ecological status). Advice with respect to multiple-stress analysis is given in [5], this volume. Investigative monitoring as a second step should be designed to verify the hypotheses from the previous monitoring evaluation and to identify and understand causal pathways (upper right box, Fig. 1). Based, for example, on identified chemicals, which are assumed to pose risks to ecological status, mitigation measures could be taken accordingly (lower right box, Fig. 1). The final outcome of the management measures should be subjected to operational monitoring, which might lead to a refinement of all other elements as shown in

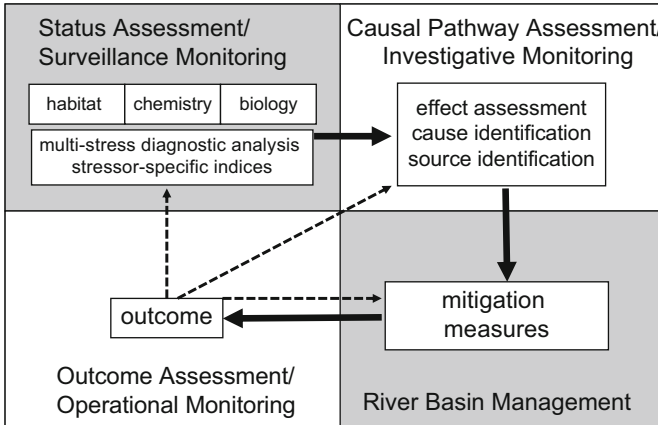


Fig. 1 Monitoring and risk assessment scheme adapting an idea by Cormier et al. [7]

Fig. 1 (lower left box). In the following sub-chapters, short overviews on approaches and tools for stressor-specific risk assessments according to this scheme are provided.

This chapter focuses on the discussion of the following stressors (i.e. risk sources; see Fig. 2 in [2], this volume): toxicity, acidification, salinisation, eutrophication, parasites and pathogens, invasive alien species, hydromorphological degradation, changing water levels and sediments and suspended matter. These stressors may be directly approached by mitigation measures and other management activities. We are aware that all processes involving stressors and receiving environments are superimposed by climate change, which often means an increase in temperature and a change in precipitation pattern. Temperature is a direct factor that restricts the distribution of fish and other aquatic organisms into specific environments. It was suggested that a mismatch between the demand for oxygen and the capacity of oxygen supply to tissues is the first mechanism to restrict whole-animal tolerance to extreme temperatures as demonstrated for eelpout in the Wadden Sea [8]. Complex changes on fish due to climate warming are also expected on the community level. These changes will include effects on fish community structure, life history traits, feeding mode, behaviour and winter survival. In an extensive review of the impacts on lake fish communities, the authors conclude that all these changes imply higher predation on zooplankton and macroinvertebrates with increasing temperature [9]. This is caused by a shift of fish communities towards small and abundant plankti-benthivorous fish, resulting in enhanced predator control of zooplankton and higher phytoplankton biomass [10]. In addition, there are pathways by which climate change enhances eutrophication, including enhanced phosphorous loading from land to streams in northern temperate zones due to higher winter rainfall and temperature-mediated higher phosphorous release from sediments [10].

Also in many other cases, temperature rise will interact with other stressors and often enhance their effect. Typical examples are invasive species and pathogens, which are favoured by enhanced temperatures. In many rivers, for example, the proliferate kidney disease (PKD) is an emerging disease and decisive stressor for salmonid fishes [11]. PKD is provoked by temperature and caused by infective spores of the myxozoan parasite *Tetracapsuloides bryosalmonae*, which develops in freshwater bryozoans. Climate change will influence the likelihood of aquatic alien species becoming established by eliminating cold temperatures or winter hypoxia that currently prevent survival [12]. Not only alien animals and plants but also invasive microorganisms supported by global warming have the potential to significantly alter community structure and ecosystem functioning [13].

Also exposure to hazardous chemicals and susceptibility of the ecosystems may be strongly affected by climate change [14]. Physical processes such as resuspension of contaminated sediments may enhance exposure of aquatic organisms, while higher temperature increases uptake rates of pollutants via changes in the ventilation rate. Models of climate change impact on bioaccumulation of persistent organic pollutants from sediments in the aquatic food web forecasted unexpected effects of climate change on contaminant cycling in the Great Lakes [15]. The actual impact of climate change on bioaccumulation potential will depend on the nature of the predator–prey relationships that emerge as well as on preferences and sensitivities of prey and predator species.

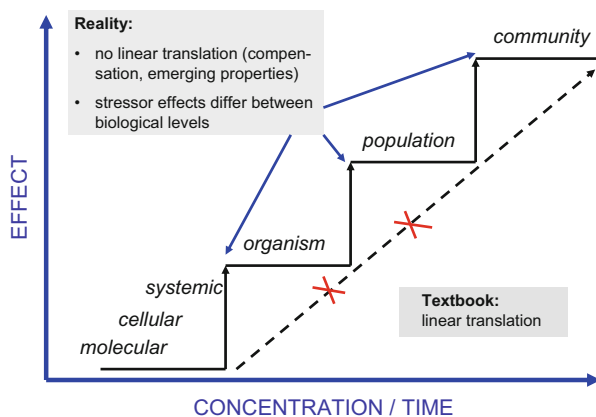
On the effect side, examples have been reported, where temperature tolerance in fish has been reduced, e.g. by pesticides [16], but also where physiological functions, such as heart beat of crabs, are more vulnerable to toxicants (e.g. copper) at higher temperatures [17]. On the other hand, climate warming may in specific cases also reduce the effects of toxicants on biota, as suggested for pesticides and amphibians [18]. The underlying mechanism is an accelerated embryonic and larval development and thus a reduced duration of contaminant exposure for taxa with aquatic early life stages.

It may be summarised that climate warming is an important factor that impacts most of the stressors discussed in this chapter and in many cases may further deteriorate the ecological status by enhancing their effects. However, since the effects of temperature rise in most cases will be detected as enhanced effects of other stressors and the mitigation measures necessarily focus on these other stressors, climate warming is not separately discussed as a stressor in this book chapter, but included in the discussion of the other stressors where appropriate.

2.1 Toxicity

There is increasing evidence that exposure of aquatic ecosystems to chemicals causing adverse effects is one of the reasons for the insufficient ecological status in many European surface waters [19–26]. These adverse effects may include acute and chronic toxicity, e.g. by the application of pesticides in agriculture, but also

Fig. 2 Textbook and realistic propagation schemes for chemical effects across levels of biological organisation after Segner et al. [34]



numerous sublethal effects, for example, on reproduction or development. The identification of such adverse effects caused by the exposure to chemicals in often multiple-stressed ecosystems is a challenging task. It is further aggravated by the fact that exposure to hazardous concentrations of specific chemicals may be not constant, but with strong fluctuations or occurring as concentration peaks, e.g. of pesticides after application or heavy rainfall and run-off events. This indicates that effects observed in the ecosystem may have been caused by an exposure that is no longer detectable. Latent effects, as well as ecological interactions causing cascades of effects [27] are particularly challenging. It has been suggested that changes in species interactions caused by chemicals are more common than direct chemical effects [28, 29]. One good example may be the increase of trematode infections in a declining amphibian species exposed to chemicals [30]. It has also been shown that strength and type of biological interaction could substantially modify concentration levels that cause biological effects. Microcosm experiments with caddisfly and mayfly larvae have shown that competitive interactions at the population level can significantly compensate and therefore ameliorate toxic effects [31, 32]. In contrast, an experiment with predation (harvesting) of brine shrimp *Artemia sp.* has shown that predation stress could exacerbate toxic effects by exhausting regulation capacity of the prey population [33]. These examples highlight that the propagation of chemical effects across levels of biological organisation is neither strictly linear nor deterministic. This complicates any causative assignment of ecological impairment to chemical contamination (see Fig. 2).

Hydromorphological alterations (see [4], this volume), such as channelisation or dams, as well as eutrophication and excess algal growth, are often visible and thus easy to diagnose. The presence of micro-pollutants that may cause adverse effects, however, is generally not detectable using the human senses and requires sophisticated analytical instruments. The detection of chemicals is further complicated by the enormous numbers of different compounds that may enter the environment. At present, a total number of about 14 million chemicals are on the market, while about

65 million chemicals are known [35]. These include industrial chemicals, solvents, flame retardants, dyes, pesticides and biocides, personal care products, pharmaceuticals, detergents, food additives, incineration products and many more, together with their respective by-products and metabolites. Chemical analysis is generally a very powerful tool to detect and quantify individual compounds in environmental matrices at very low concentrations. However, since only a limited number of chemicals can be analysed and only for a fraction of them effect data are available, classical target analysis will fail to provide a comprehensive measure of chemical contamination and adverse effects. Thus, alternative approaches based on effect detection that integrate over the whole set of known and unknown chemicals with subsequent assessment of causal pathways are increasingly important complements to individual chemical-based risk assessments. Monitoring and management focusing on a limited, preselected list of compounds, such as the EU-WFD priority pollutants, help to phase out chemicals that are agreed to be hazardous, but provide only limited information on toxic stress as such [19, 23].

Risk in general and risk of chemicals in particular are defined as a function of the probability and magnitude of adverse effects (see [36], this volume). Thus, risk assessment has to deal with the integration of effect and exposure data often represented by the ratio between predicted environmental concentrations (PEC) and predicted no-effect concentrations (PNEC) using generic data on production, fate and toxicity of the chemicals of concern. This approach may provide an appropriate tool for regulation of chemicals on the European level. However, as a basis for decisions on river reach-specific management measures, assessment of site-specific toxic impacts may be a better tool. These higher-tier impact assessment approaches [27] focus on local causes and pathways of exposure including bio-availability, transformation and transport processes and consider local receptors within the ecosystem that is actually exposed. Thus, closely linked ecological, toxicological and chemical analytical approaches are required to analyse and understand exposure–effect chains. For a given organism, these effect chains start with an external exposure concentration of a certain chemical that will be accumulated to an internal concentration, triggering an internal response that may have an individual effect on that organism. The concurrent effects on several individuals of a population may affect their demography and could finally lead to alterations in community structure. These complexities make it difficult to identify and confirm exposure to specific toxic compounds as the causes of ecological degradation in aquatic ecosystems.

2.1.1 Evaluation of Surveillance Data

The starting point for every impact assessment as discussed here is surveillance monitoring which provides data on biological quality elements (BQE), concentrations of priority pollutants and basin-specific chemicals in water, sediments and biota, as well as physicochemical and habitat parameters. For a sound-integrated evaluation of these data, it is essential that they have been

collected approximately at the same time, at the same sites, and have been measured using harmonised methods. Additional data on in vitro and in vivo effects would be helpful but are not required by the WFD and thus in most cases are not available. Unfortunately, this may also prevent the application of weight-of-evidence approaches in line with the triad concept (which requires evidence of exposure, effects and linkages [37, 38]), which can be very helpful in providing evidence of toxic stress.

In addition to monitoring data, a further line of evidence suggesting a possible toxic risk to aquatic ecosystems may be expert knowledge and available data on possible sources of toxic contamination, including industry; historical contamination sources, such as landfills or contaminated sediments and soils (e.g. industrially contaminated river banks); municipal wastewater discharge; intensive agriculture or others that may impact downstream sites and river stretches.

Abiotic Indicators

To predict local toxic impacts, concentration data for potentially toxic chemicals can be transformed to an overall impact estimate to quantify the proportion of species exposed to concentrations that have the potential to elicit an effect. Termed as multi-substance potentially affected fraction of species (msPAF), this integrates the species sensitivity distribution approach with mixture toxicity modelling and corrections for bioavailability [39].

Another approach to evaluating chemical analytical data estimates potential acute toxicity to reference species, i.e. *Daphnia magna* for invertebrates, *Selenastrum capricornutum* for algae and *Pimephales promelas* for fish, applying the model of concentration addition (see [40]). Respective toxic units (TU) are based on the ratios of measured environmental concentrations (C) of compound i and laboratory effect concentrations (EC_{50}) or lethal concentration (LC_{50}) values [19, 41]. These values (EC_{50} , LC_{50}) indicate the concentration for a given toxicant that within a given exposure duration (often, 24, 48 or 72 h) affects 50 % of the test specimen (such as *D. magna*, *S. capricornutum* or *P. promelas*). The ratios are logarithmised according to the following equation:

$$TU = \log \sum_{i=1}^n \frac{C_i}{LC_{50i}}$$

For compounds for which no toxicity data are available, quantitative structure–activity relationships (QSAR) may help to provide estimates from chemical properties. Although these values do not formally comply with the requirements for a legally binding Environmental Quality Standards (EQS, see [42]), they do allow for effect estimates in the absence of other experimental data and for a prioritisation of substances to be monitored. The required effect values can be reasonably well predicted for all three trophic levels (algae, invertebrates and fish),

using read-across methods from existing experimental values for similar compounds [43, 44]. In cases where no enhanced toxicity is expected from structural alerts [45], respective baseline QSARs could be applied that represent the minimum toxicity expected from the lipophilicity of the compounds [19, 45]. However, similarly to the msPAF approach, only compounds that have been monitored can be included in the effects assessments.

Biological Indices

A promising complement is the evaluation of community data on BQEs using stressor-specific metrics, such as the SPecies-At-Risk (SPEAR) indices [46–48], which can be used online¹ or as part of the ASTERICS software.² The main advantage of the SPEAR approach is that it is based on biological traits, species characteristics (e.g. body size, generation time) that determine a taxon's response to stressors, which are available mostly on species, genus or family level and not on higher taxonomic composition, as are many conventional bioassessment indices (e.g. the percentage of “Ephemeroptera, Plecoptera and Trichoptera” (EPT); see [49]). Therefore, it is relatively independent of confounding factors, and application of this system is not constrained by geographical and geomorphological factors and associated differences in biological communities [50]. This approach classifies species according to their physiological and ecological sensitivity to organic toxicants [51]. The SPEAR index provides a measure of the loss of species sensitive to toxicants. It has been found to be well correlated to contamination expressed as TU for *D. magna* in several river basins including the German Weser [46], the Scorff in France and the Porvoonjoki in Finland [20] as well as the Llobregat in Spain [19, 48] covering five different ecoregions (see WFD for definition), while it was not significantly correlated to other stressors such as eutrophication or oxygen depletion [48]. Furthermore, the SPEAR_{organic} index has been shown to be relatively stable over an altitudinal gradient from 90 to 2,000 m above sea level, in the uncontaminated watercourses of the Ob' River catchment, Southwestern Siberia [52]. Finally, SPEAR_{pesticides} had the closest relationship with pesticide toxicity compared to commonly used water quality metrics in a study on 24 sites in South-East Australia [53]. Moreover, the SPEAR indices showed predictive power on the family level, and therefore they can easily be included in various biomonitoring programmes employing this coarse identification level [19, 25]. Similar indices for other organism groups and other stressors are urgently required.

¹ See: www.systemecology.eu/SPEAR/Start.html

² See: www.fliessgewaesserbewertung.de

2.1.2 Confirmation of Toxicity as Causation of Degradation

If the surveillance monitoring data evaluation suggests site-specific toxic risks on BQEs, investigative monitoring could be designed in order to confirm toxic pressure and to identify causative chemicals and sources.

In Vitro and In Vivo Effect Assessment

In vitro and in vivo effects in laboratory test systems together with laboratory and in situ biomarkers are important lines of evidence in site-specific causal pathway and risk assessments. Measurable in vivo effects of environmental samples on survival, growth, morphological development and reproduction of standard test organisms including bacteria, algae, invertebrates and/or fish provide direct evidence of the impacts of contaminants as long as other factors, such as pH, oxygen content, conductivity or nutrients, are constant. In aquatic ecosystems, the water phase, suspended matter and sediments are the most relevant matrices for effect assessments. Whether the focus of biotesting is on the water phase or on sediments and/or suspended matter strongly depends on the objectives of analysis.

Sediments and suspended matter are known to be sinks for many hydrophobic organic chemicals as well as for metals. Sediment-associated contaminants may affect benthic communities and are therefore often the first links in aquatic food webs. The outcome of effect assessments for sediments is strongly affected by the experimental design of exposure conditions. Sediment contact tests are available, particularly for benthic invertebrates including insect larvae such as *Chironomus tentans*, oligochaetes such as *Lumbriculus variegatus*, crustaceans like *Hyalella azteca* [54] and nematodes such as *Caenorhabditis elegans* [55]. These tests simulate complex natural exposure conditions and bioavailability. In addition to organic toxicants and metals, other confounding factors such as grain size, organic matter content, oxygen, ammonia and hydrogen sulphide concentrations, pH and conductivity may determine the responses of the test organisms. The greater environmental realism of this approach may be balanced by its limited diagnostic power; without additional approaches such as Toxicity Identification Evaluation (TIE, [56]), it can be difficult to discriminate the effects of toxicants from confounding factors. The exposure in sediment contact tests is complex and is further complicated if dilution series are performed to determine concentration–effect relationships.

Since freely dissolved concentrations in porewater are believed to represent the major pathway of exposure for sediment-dwelling organisms, porewater isolation and testing have been applied for sediment effect assessments [57]. Unfortunately, many natural porewaters are anoxic. As for whole-sediment testing, compounds such as ammonia and hydrogen sulphide often dominate measurable effects. Since most invertebrates cannot live without oxygen, porewater testing is performed under oxic conditions. Thus, constituents such as hydrogen sulphide are oxidised,

bioavailability of metals is changed and much of the environmental realism is lost without gaining additional diagnostic power [58].

An alternative to whole-sediment and porewater testing is the use of extracts. Testing of exhaustive organic extracts using methods such as Soxhlet, pressurised liquid extraction, ultrasonic or microwave extraction with strong organic solvents avoids the impacts of confounding factors and inorganic contaminants. This approach parallels classical chemical sediment analysis and thus allows a good comparison of effects to chemical monitoring data for organic compounds, e.g. by applying TUs. On the basis of total concentrations, this approach helps to quantify how much of the measured toxicity may be attributed to measured compounds. However, with respect to risk assessment, this approach represents an unrealistic worst-case scenario, ignoring the limited bioavailability of most sediment-associated compounds. Based upon the concept that only rapidly desorbing compounds may become bioaccessible, mild extraction techniques, e.g. by using TENAX, have been suggested and applied for higher environmental realism [59]. An alternative approach to considering bioavailability in sediment extract-based testing is the application of partition-based dosing techniques instead of solvent dosing [60, 61]. In summary, there are several different approaches to effect assessment of sediments with different strengths and weaknesses; depending on the situation, different approaches or combinations thereof may be appropriate.

For many aquatic organisms, the uptake of contaminants from the water phase represents the major pathway of exposure. Thus, direct testing of the water phase, as done for whole effluent testing would be a logical approach. However, it is important to remember that pollution control measures within the last decades have fortunately resulted in a situation in most river basins where directly measurable toxic effects, e.g. of surface waters, are a rare exception. In addition, there are good arguments against an effect-oriented, rather than a risk-oriented, system of making management decisions. If actions are taken only when effects are already apparent, which is often only the case at high exposures, programmes of measures (PoM) may be too late and ecosystem recovery from such conditions may be difficult [27]. Thus, while direct testing maybe a reasonable approach for effluent control before entering surface waters, biotesting of surface waters without pre-concentration is of limited relevance. As is known from investigations using the SPEAR index, there are strong correlations between the loss of invertebrate species sensitive to organic chemicals and the contamination of chemicals already above a logTU value of -3 , referring to concentrations of 1/1,000 of the acute LC₅₀ value for *Daphnia magna* [19, 20, 46, 50]. Moreover, a number of micro- and mesocosm studies have shown that in realistic ecological contexts, lethal and sublethal effects relevant to community-level alterations (e.g. downstream drift of organisms) may appear at concentrations from 1/10 to 1/100 of the acute LC₅₀s for *Daphnia magna* or other invertebrate species [62–64]. Furthermore, laboratory experiments with metals [65] and insecticides [66] have shown that delayed lethal effects appearing after exposure may be caused by concentrations about 1/50 of those causing acute effects. Thus, a pre-concentration of dissolved contaminants by a factor of 100 or 1,000 before biotesting is recommended to see acute effects in biotests at environmentally effective concentrations [67].

While *in vivo* biotests provide valuable information on overall adverse effects, they lend little insight into the causative modes of action (MoA) of the responsible chemicals [68]. Information on MoA may be very helpful in identifying underlying causes of adverse effects and potential biological targets and thus for risk assessment. Numerous *in vitro* assays have been developed for MoA-based biotesting. Toxicological endpoints using *in vitro* assays include (1) genotoxicity (e.g. umuC test, [69, 70]), (2) mutagenicity (e.g. Ames II, [71]), (3) oestrogen receptor (ER-Calux, yeast oestrogen screen (YES, [72])), (4) androgen receptor agonism and antagonism (AR-CALUX, [73, 74]), yeast androgen screen (YAS, [75, 76]), (5) aryl hydrocarbon receptor (AhR)-mediated effects (DR-CALUX, EROD induction [77]), (6) tumour promotion by inhibition of gap-junctional intercellular communication [78, 79], (7) binding to transport protein transthyretin (TTR-binding assay, [80–82]) and (8) antibiotic activity [83]. These assays can be used as diagnostic elements in the establishment of exposure–effect chains by helping to characterise MoAs and causative pollutants. Comparison of *in vitro* effects with chemical analytical results may help to justify or revise the selected compounds for chemical monitoring by comparing observed MoAs with known or expected MoAs of the chemicals. *In vitro* assays may respond more sensitively to toxicants and thus provide a kind of early warning tool, although it should be stressed that a direct prediction of adverse effects in the environment from *in vitro* results is not possible. Proponents of the new “-omics” techniques including genomics [84, 85], proteomics [86] and metabolomics [87, 88] argue that these methods may provide powerful diagnostic tools to identify MoAs in environmental samples and to develop new biotests and biomarkers, although these have yet to be released.

Closely related tools that support the establishment of exposure–effect chains are *in vivo* biomarkers. Biomarkers are defined as a change in a biochemical response that can be related to exposure to or the toxic effects of environmental chemicals [89]. These changes can be measured in laboratory test organisms or in organisms *in situ*. As with *in vitro* assays, a number of biomarkers are MoA-based tools reflecting endpoints similar to *in vitro* assays. Frequently applied biomarkers include vitellogenin induction in male fish as a marker for exposure to estrogens [68], the induction of hepatic CYP1A1 or EROD activity [77] or the observation of neoplasia in organisms exposed to carcinogenic compounds [90]. Biomarkers provide mechanistic signposts [68], indicating whether an organism has actually been exposed to chemicals with a specific MoA. Thus, they may help to target detailed chemical and biological analysis of water, sediments and biota. They may also help to prioritise discharges of concern with regard to municipal and industrial effluents. However, as with *in vitro* assays, they are poor predictors for risks or adverse effects in aquatic ecosystems, and the lack of integrated mechanistic models, complicated time- or dose-dependent responses, the unclear link to fitness as well as confounding influences prevents their application for predictive risk assessment [91].

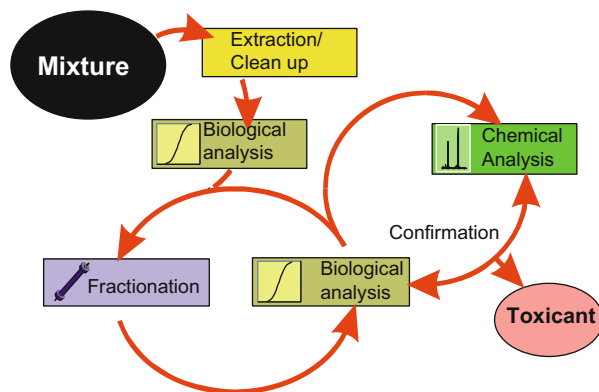
Since organisms in nature never live in isolated populations but rather in communities with ecological interactions, such as competition and predation that can modify strength of toxic effects [31–33], the assessment of toxic effects on

communities is a crucial step, particularly in site-specific effect and risk assessment. As it is rather difficult to assess effects on ecosystems as such or to discriminate toxic effects from other impacts, often micro- or mesocosms are used as model ecosystems under controlled conditions to assess community effects. One of the most promising approaches in that field may be the pollution-induced community tolerance (PICT) concept [92–94]. The concept is based on the fact that organisms survive in environments exposed to toxicants because they are sufficiently tolerant to these compounds. Since the organisms in a community differ in sensitivity, the toxicant will eliminate or inhibit sensitive species or individuals while favouring more tolerant ones. This shift in tolerance should be detectable in short-term experiments [28]. Thus far, this restricts the application of PICT to small organisms with relatively short life cycles such as bacteria, algae or biofilms. PICT is thought to be inherently sensitive as it occurs when the most sensitive components are first eliminated [92] and may provide cause–effect relationships. It may also help to confirm individual harmful toxicants in complex environmental mixtures and was thus suggested as a tool for hazard confirmation in effect-directed analysis [95]. The reason is that only those toxicants that exert selection pressure should result in community changes and increased tolerance [94]. However, these communities might also be tolerant to chemicals with similar modes of action and detoxification mechanisms [96]. From the ecosystem point of view, PICT does not only mean the disappearance of some species but may be related to reduced genetic diversity, reduced tolerance to other stressors and reduced resilience in general [28]. It has been shown, for example, that macroinvertebrate communities from metal-contaminated sites were highly tolerant to metals but significantly more sensitive to acidification [97], UV-B radiation [98] and stonefly predation [99].

Identification of Key Toxicants Causing the Effect

After confirmation of toxicity as one of the factors that may affect BQEs at a specific site and the identification of possible target organisms and MoAs, the identification of the compounds causing the effects is an essential step towards site-specific impact and risk assessment and management. In the early 1990s the US EPA developed an approach called Toxicity Identification Evaluation (TIE) to characterise and identify major toxicants in effluents as an advancement of whole effluent testing [100–102]. Simple chemical manipulations of samples such as the stripping of volatiles, solid-phase extraction of organic compounds, pH manipulations to alter ammonia toxicity and the use of EDTA complexation of metals were combined with biotesting to characterise the observed toxicity in treated effluents. A subsequent step often combined biotesting with, for example, reversed-phase HPLC fractionation and GC-MS analysis for organic compounds. In a final step, identified compounds were confirmed as the cause of the measured effect. The translation of these basic principles to sediment assessment resulted in a TIE guideline for sediments [56].

Fig. 3 Scheme of effect-directed analysis (adapted from [103])



In parallel with TIE, effect-directed analysis (EDA) has been developed by analytical chemists (Fig. 3). EDA was triggered by the motivation to focus chemical analysis of complex environmental and technical mixtures on those compounds that are toxicologically or eco-toxicologically relevant. Thus, EDA combines chemical analysis with chromatographic fractionation techniques using biological detectors to direct their analysis. EDA has been applied successfully for the identification of toxic [104], mutagenic [105], AhR-mediated [106], estrogenic [107] and androgenic [108] compounds in effluents, sediments and soils and also air particulate matter and technical mixtures [103, 109]. The complexity of a mixture that has been found to exhibit a response in a bioassay is sequentially reduced by fractionation directed by the biological response. Finally, isolated active fractions are subjected to qualitative and quantitative chemical analysis. The EDA approach has been further significantly advanced, e.g. in the European project MODELKEY [110], by developing improved clean-up and fractionation techniques [111, 112] and by advancing LC- and GC-based structure elucidation techniques including novel computer tools [113–115]. Further progress towards a greater environmental realism has also been made by integrating the concept of bioavailability into EDA of sediments [59], applying extraction methods that consider the bioaccessibility of the eluates [116, 117] and partition-based dosing [61, 118] techniques, which mimic differences in the availability in the biotests. Based upon these techniques, a shift in the priority of sediment-associated toxicants from non-polar compounds, such as PAHs, to more polar chemicals, such as triclosan, has been observed. Moreover, clear indications could be found that in complex environmental mixtures, androgen activity of certain compounds can be masked, e.g. by other compounds with antiandrogen activity, which can be unveiled by effect-directed analysis [119].

It should be stressed here that, similar to TIE, confirmation of identified toxicants is a crucial step in EDA before these chemicals should be considered for assessing risks [91]. This confirmation should include at least analytical confirmation of tentatively identified compounds and quantitative effect confirmation. A further confirmation may include *in vivo* effect assessment under realistic exposure conditions, e.g. by using sediment contact tests or by *in situ* community assessment, e.g. based on PICT.

Source Identification

An important tool for the identification of local sources of contamination is the geographical evaluation of monitoring data together with available information on possible sources, including industrial production sites, agricultural areas, landfills, harbours, municipal wastewater and others. Successful identification of the potentially causative chemicals prior to source identification is advisable, since specific chemicals are often characteristic of particular types of sources, such as certain production facilities or land uses. Source identification can be supported by analytical studies and modelling approaches including the use of tracers, pollution patterns and fingerprints. These approaches are often referred to as environmental forensics.

One example of environmental forensic comes from polycyclic aromatic hydrocarbons (PAHs), which represent a large number of individual compounds which are derived from three broad classes of sources: fossil fuels, burning of organic matter and transformation of natural precursors in the environment. These sources generate different PAH mixtures, also called profiles or fingerprints. Furthermore, diagnostic ratios of individual compounds and multivariate statistical analysis of PAH fingerprints can also provide excellent opportunities for the discrimination of, for example, different oil or soot sources [120–123]. This approach allowed for the identification of parking lot sealcoat as one previously unrecognised source of urban PAH contamination in the USA [124]. Investigating PAH-related long-term biological damage along the shoreline of Alaska due to the Exxon Valdez oil spill in 1989, a bioavailability assessment using engineered bacterial biosensors was able to discriminate effects of PAHs from the oil spill from those which were suggested to be derived from organic-rich hydrocarbon rocks [125]. While PAHs from the spill were bioavailable, no bioavailability was detected for the rock-associated PAHs.

Multivariate statistics such as cluster and principle component analysis of congeners and isomers have been shown to be excellent tools to identify specific sources of polychlorinated dibenzo-*p*-dioxins and furans (PCDD/Fs), polychlorinated biphenyls (PCBs), polychlorinated naphthalenes (PCNs), DDT derivatives and hexachlorocyclohexane isomers HCHs [95, 126, 127].

2.1.3 Considerations for Site-Specific Risk Assessments and Downstream Risks

Refinement of Exposure Assessment for Identified Key Toxicants

After the identification of potentially causative toxicants and probable sources, a refined exposure assessment for these compounds under local and regional conditions may be performed. Key elements should be the assessment of bioavailability, food chain studies and models to assess risks to higher trophic levels, as well

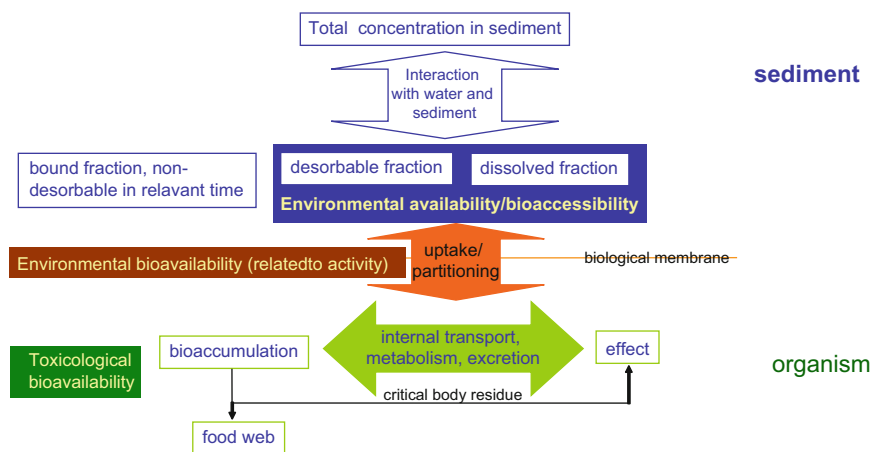


Fig. 4 Scheme of the process of bioavailability of organic pollutants (adapted from [130])

as studies and models on the fate, including persistence and partitioning between different compartments, of the respective chemicals.

Bioavailability assessment is of specific relevance for toxicants associated with sediments or suspended matter. However, the bioavailability of compounds in the water phase may also be reduced due to binding to dissolved organic matter [128]. Bioavailability has been defined as the degree to which the chemicals that are present in a sample or in the environment may be absorbed by human or ecological receptors [129]. This may be a comprehensive definition. However, it does not directly lead to a measurable value. Thus, many different concepts of bioavailability may be found in the literature that may result in different conclusions. In the following we will apply the concept shown in Fig. 4 that was adapted from Harmsen [130], separating bioavailability in sediments into three different processes: (1) environmental availability or bioaccessibility, describing the fraction of sediment-associated molecules that are already dissolved or may rapidly desorb from the particles, (2) environmental bioavailability describing the activity-driven partitioning process of the bioaccessible fraction between sediment particles, water phase and biota and (3) toxicological bioavailability including processes in the organisms including internal transport, metabolism and excretion, which finally result in a concentration at the internal receptor that may result in an effect. Hence, not all bioavailable concentrations automatically cause effects in the organism.

Which of these processes is considered in a given risk assessment may depend on the objective of that assessment. Bioaccessibility derived from mild extraction methods using TENAX [116, 131], supercritical carbon dioxide [132] or cyclodextrins [133] provides a measure of the amount of a toxicant that may desorb from the matrix in a realistic time frame and thus may pose a risk to biota. Bioaccessibility-directed extraction excludes, for example, PAH molecules that are tightly bound to soot particles and may be extracted only by exhaustive organic

extraction methods such as pressurised liquid extraction (PLE), while their desorption under environmental conditions will occur only within years to millennia [134]. Unfortunately, different bioaccessibility-directed extraction techniques may still provide different results, and these need further validation and harmonisation [59]. Rapidly desorbing concentrations as such are poor predictors of accumulation in biota and thus of risk. However, at least for non-polar compounds such as PAHs, PCBs and other halogenated aromatic hydrocarbons, rapidly desorbing concentrations may be a reasonable basis for the estimation of concentrations in water and biota using the principles of equilibrium partitioning.

Freely dissolved concentrations in porewaters and also in surface waters are used as direct measures of the chemical activity of contaminants [135] and thus may provide the most promising approach to measuring exposure and predicting risks in aquatic ecosystems. Passive sampling techniques including semipermeable membrane devices (SPMDs, [136, 137]), low-density polyethylene devices (PEDs, [138]), solid-phase extraction disks [139], silicone rubbers [140] and solid-phase microextraction (SPME) fibres [141, 142] have been shown to be powerful tools for measuring freely dissolved concentrations. Freely dissolved porewater concentrations have been shown to provide more precise bioavailability estimates than those derived from traditional equilibrium partitioning approaches using sediment extracts. However, feeding may modify bioavailability in a manner that cannot be simply predicted by equilibrium partitioning models [143].

Finally, internal exposure in organisms and thus effects are the result of equilibrium partitioning based on activity together with organism-specific toxicokinetics including internal transport, metabolism and excretion. Thus, a prediction of risks of a compound for a selected species on the basis of freely dissolved concentrations without additional information on toxicokinetics of the species remains highly uncertain. In contrast, internal concentrations causing a certain effect are more similar between different species, particularly for those chemicals acting according to the mechanism of baseline toxicity, and provide an excellent basis for site-specific risk assessment at least for persistent organic pollutants [144]. Internal concentrations also provide the opportunity to link risk assessments in different environmental compartments. Care should be taken for compounds that are readily metabolised in the organism. Despite causing effects, they will not be found in the tissues in relevant concentrations. For compounds with well-known metabolism such as for PAHs and estrogenic steroids, metabolites may be analysed, e.g. in the bile [145, 146].

Since toxicants may undergo bioaccumulation and biomagnification and may reach critical body burdens only at higher trophic levels, site-specific risk assessments should include food web studies [147]. Depending upon study objectives, food webs can be considered in bioaccumulation models based upon equilibrium partitioning as shown for aquatic food webs with sediment interaction such as those by Thoman et al. [148]. Food web models have been successfully used to derive environmental quality criteria on the basis of internal concentrations [149] and for the prediction of the interactions between eutrophication and insecticide application based on data derived from mesocosm studies [149].

Prediction of Site-Specific Toxic Risks to Populations, Communities and Ecosystems

A goal that matches with sustainable water management is to prevent risks to aquatic ecosystems and human health. Generic individual chemical-based risk assessments based on simple PEC/PNEC ratios provide only limited information on actual site-specific risks. Thus, higher-tiered risk assessments (RA) focusing on population, community or ecosystem levels under in situ exposure conditions have been developed. These RAs follow the basic idea of integrating exposure and effect assessments for individual-level risk characterisations followed by population-level risk characterisations. By applying this approach, the risks of planar polychlorinated aromatic hydrocarbons to great cormorants living around the Tokyo Bay, Japan, have been assessed [150]. Exposure assessment provided the probability distribution of 2,3,7,8-tetrachlorodibenzo-*p*-dioxin toxic equivalents in cormorant eggs on the basis of congener-specific sediment concentrations, biota-sediment accumulation factors and biomagnification factors. The procedure resulted in egg mortality risks on an individual level. On the population level, a stochastic age-structured matrix model was used to estimate the risk of population size reduction.

These models have been criticised for (1) the assessment of toxic effects of complex mixtures as a compilation of additive effects, (2) the lack of knowledge on the influence that chemicals in a mixture have on each other's toxicokinetics, (3) the influence of other habitat parameters and (4) ignoring of physiological and behavioural alterations caused by toxicant exposure [151]. An alternative model that attempts to overcome these shortcomings is the interaction assessment (INTASS) model platform, a model by which field data can be used to estimate a quantitative expression for population growth rate as a function of the various measured field variables [152].

The basic concept of a community is the interaction between different species, e.g. by predation or competition, and between species and their abiotic environment. Thus, assessment of risks to communities should consider these interactions [31–33]. Particularly, if subtle and not readily transparent ecotoxicological effects are hypothesised, the application of community-level models, which do not only characterise trophic linkages among species but also depend on knowledge of the nature of competitive and predator–prey processes as well as the intensity of those interactions together with functional food webs, is suggested [147]. One example of this kind of model is the US EPA AQUATOX model.³

Mechanistic models of sublethal effects of toxicants on the functioning and structure of simple aquatic food chains using physiological parameters, such as cost for growth and maintenance, and assimilation efficiency may also help to better understand toxic effects on nutrients–prey–predator systems [153].

³ See: <http://www.epa.gov/waterscience/models/aquatox/>

Prediction of Basin-Scale Effects

Contaminants may cause effects locally, i.e. at the source of contamination. However, these contaminants may also cause effects downstream, as water and fine particles are transporting vectors for chemicals which are dissolved and/or adsorbed to fine sediment particles (see [4], this volume). The river basin morphology, hydrology and hydraulics determine the transport direction and velocity of the water carrying the chemicals. In a downstream direction, the water discharge normally increases, which leads to increasing dilution, at least for point sources. During downstream transport, physical and biochemical processes may cause decay or removal (from the water phase) of certain chemicals, though, by definition, for the more problematic persistent chemicals, such processes are rarely quantitatively important. Particle-associated chemicals may also be deposited in slow-flowing areas of a stream or in lakes/ponds.

Hydrophobic organic chemicals and trace metals interact with suspended fine particles and may (temporarily) settle in areas with a reduced transport capacity, where they accumulate and may create sediment layers with a historical pollution burden [154]. During high-flow events, or as the result of hydromorphological change, such layers may be resuspended and the accumulated chemicals may become waterborne again, usually to settle further downstream [155]. On larger spatial scales, river basins contain a wide spectrum of areas with different return periods for such resuspension events. The net result is that not just water but also fine particles move through the basin in a downstream direction due to subsequent flood events, albeit not as fast as water, and with a substantial net retention of particles and chemicals in areas of long-term sedimentation (e.g. floodplains).

Recent research on nutrient management on a basin scale (*daNUbs* project, [156]) has revealed that there is a conflict of interest between the management of local and basin-wide effects, which is also true for toxic chemicals. Pollution sources discharging into water bodies with a high local dilution capacity usually do not lead to severe local effects, but the discharged chemicals are effectively transported downstream where they may cause problems in sensitive downstream water bodies, such as lakes, estuaries and coastal waters. On the other hand, pollution sources discharging in water bodies with a low local dilution capacity may lead to severe local effects, but the problems remain local, and, under normal conditions, only relatively small amounts of chemicals are flushed out to cause downstream impacts or downstream dilution may attenuate effects. In this context, research related to the fate of phosphorus has revealed that streams with relatively low run-off and relatively high residence time are more effective in retaining particle-bound nutrients than are streams with relatively high run-off and relatively low residence times [157, 158]. It is likely that this also holds true for other chemicals. Local effects are determined by concentrations (unit mass/L) and consequently regulated by means of Environmental Quality Standards (EQS) for water and/or sediment concentrations. Downstream effects, however, are often determined by transported loads (unit mass/s), but the ‘flushing’ of problematic compounds during flood events are poorly resolved by monitoring programmes and usually is not explicitly regulated.

The WFD calls for water management on the basin scale and therefore forces water managers to consider downstream effects of pollution sources as discussed in this section. This concerns trans-boundary effects, but also effects in downstream sub-catchments within the same country.

The assessment of downstream effects as discussed above can be supported by water quality models [159, 160]. These models can be used to estimate the downstream exposure of aquatic systems to a single pollution source or due to a cluster of sources. The requirements for such modelling can be listed as follows:

- An accurate representation of the river basin morphology, hydrology and hydraulics
- An accurate representation of the fate of individual chemicals, as the result of various degradation processes, volatilisation and sorption
- For hydrophobic organic chemicals and trace metals: an accurate representation of the fluxes and fate of fine particles
- An accurate representation of the bioavailability of the target chemicals and optionally the propagation of the target chemicals within the food chain

Starting in the late 1990s, geographically based river basin models dealing with various chemicals appeared in the literature, building on a rapidly developing variety of environmental databases. Examples of such models are Geography-referenced Regional Exposure Assessment Tool for European Rivers (GREATER, [161]), the US EPA BASINS modelling framework [162] and WATSON [163].

The EXPOBASIN model [164, 165], as developed by the EU FP6 MODELKEY project [110], has been set up with the above requirements. It is intended to be used as a rapid assessment tool, suitable for quick applications (requiring only a few man-days) based on readily available, recent databases and defaults. It requires only very limited user-provided data. The main objective of this tool is to rank different pollution sources with respect to their potential downstream impact. This tool is innovative because it provides an integrated approach, taking into account all aspects mentioned above. Another novel aspect is that it recognises the importance of the fluxes and fate of fine particles and includes this in the conceptual model. First, yet unpublished applications indicate that the influences of the fluxes and fate of fine particles on the impacts of chemicals was successfully included in the modelling concept, but validation can be difficult.

The EXPOBASIN model quantifies the downstream impact of a pollution source in the form of two types of indicators. The first indicator (I_1) is intended to express the integral of the expected impact along the river (dx) in a downstream direction. The precise definition of such an indicator depends on the situation at hand. For example:

$$I_1 = \int_{\text{impacted river stretch}} \text{TU} \, dx$$

where TU is calculated as explained at the beginning of this section (and see [19]). It allows to both rank different pollution sources with regard to their downstream impact as well as to identify the most toxic chemical within the toxic mixture.

The second indicator (I_2) is intended to express the impact to downstream estuarine or coastal waters, for which no exact volumina are available. This indicator could be expressed as the weighted sum of the mass flowing out of the river system:

$$I_2 = \sum_{i=1}^n \frac{W_i}{LC_{50i}}$$

where W is the weighted sum of the mass, i is a specific chemical i , LC_{50} is the acute lethal concentration (see full definition at the beginning of this section) of the respective chemical and n indicates the total number of chemicals considered. Please note that the resulting index has an unusual unit (L), as only the mass is considered for the impact on the estuarine or coastal waters. The index again allows to both rank different pollution sources with regard to their downstream impact as well as to identify the most toxic chemical within the toxic mixture. Obviously, when ranking different pollution sources, care should be taken that there are not too big differences in the number of chemicals considered for each source and that there are not too big differences in the volumes of the receiving waters.

The available modelling concepts used by EXPOBASIN also allow a full evaluation of the chemical status, provided that sufficient information on all relevant pollution sources is available. However, in many river basins, for most of the relevant chemicals, complete emission databases are lacking. Point source inventories are commonly available, on a European level (European Pollutant Emission Register⁴) or on a national level (e.g. Emission Register in the Netherlands⁵). Inventories of diffuse sources, however, are often missing. This implies that a complete quantitative assessment of the expected chemical status in many water bodies is hard to achieve. In exceptional cases, a full emission database is available, and in such cases, the modelling concepts can be validated [166].

A specific application of models as described here is the ranking of sites with historically polluted sediments, which is especially relevant for sediment management purposes [167]. The risk of remobilisation of such sites is a concern in different European river basins [168, 169], also because of the impact of climate change; the expected increasing frequency of floods may lead to a mobilisation of historically polluted sediments which were stable before [154].

⁴ See: <http://www.eea.europa.eu/>

⁵ See: <http://www.emissieregistratie.nl/>

2.2 Acidification

Acid rain has deposited strong acids over large parts of industrialised and adjacent countries since the onset of the industrial revolution in the 1800s, but acidification was only by the end of the 1950s linked to repeated fish mortality in Norwegian rivers [170]. Acidifying compounds, originating from the emissions of primarily SO_2 and NO_2 , dispersed by winds, were deposited on the poorly buffered, siliceous moraine soils. Despite large reductions in the emissions of acidifying compounds, acidification of inland waters is still a major environmental stressor on the Scandinavian Peninsula, parts of Central Europe and the UK. Another large-scale process that contributes to the acidification of soils and waters is the planting and harvesting of pine forests, as this results in a net loss of hydrogen ions to the soils upon which the forest grow. More intensive forestry, and whole-tree harvesting that aims at the production of biofuels, will cause a net loss of nutrients and base cations from forest soils (e.g. [171]) and may contribute to further acidification of soils and waters, unless base cations and nutrients are added. As well as the acidification due to the deposition of long-range trans-boundary air pollutants and large-scale forestry practices, point source pollution originating from, for example, mining activities can also acidify lakes and streams. Obviously, lakes and streams/ivers on poorly buffered siliceous soils experience a higher risk of acidification than those on well-buffered, calcareous soils.

The biological effects of acidification include a gradual displacement of species, thus altering freshwater communities towards a dominance of acid-tolerant taxa. This has been well established for phytoplankton (e.g. [172]), periphyton (e.g. [173, 174]), benthic invertebrates [175, 176] and fish [177, 178]. For example, the eggs and early life stages of many fish species are highly susceptible to acidification, which can lead to skewed population age structure, characterised by few individuals in young year classes. These effects are mainly due to toxicity by ionic aluminium that increases rapidly when pH drops [179]. Particularly sensitive taxa are found among the Ephemeroptera and Gastropoda [176, 180]. Instead, acidified waters typically show a strong dominance of chironomids and invertebrate predators (e.g. [181]). Among the water plants, *Sphagnum* mosses, which assimilate CO_2 rather than HCO_3^- , are abundant in acid/acidified waters. Indirect effects of acidification include the oligotrophication of affected water bodies (review by [182]) and concurrent effects on trophic relationships (e.g. [183]). Moreover, Schindler et al. [184] have described the intimate linkage between the effects of acidification and those of increased global warming and the impairing effects of increased UV-B penetration, further adding to the complexity of identifying pure effects of acidification.

Acidification is still a major pressure in parts of Europe and the assessment of its impacts on aquatic communities is part of the WFD protocol [3]. Here, we give an overview of existing tools for the assessment of the impact of acidification and their possible pitfalls.

2.2.1 Evaluation of Surveillance Monitoring Data

Natural acidity, caused by humic and fulvic acids, also contributes to the acidity of inland waters and constrains the detection of anthropogenically induced acidification. In assessment protocols, it is thus important to distinguish natural acidity from anthropogenic acidification. As no biological assessment methods based on contemporary data have yet been developed to exclusively measure effects of acidification, water chemistry variables are needed to support the biological assessment and confirm that the lake/stream indeed is affected by anthropogenic acidification. This becomes particularly important in water bodies where humic compounds contribute to the waters' acidity. Modelling concentrations of inorganic aluminium may then be an option [185].

Abiotic Indicators

Acidification is characterised by a decrease in pH, alkalinity and concentrations of base cations, and concurrent increases in concentrations of sulphate and metals. Among the metals, increasing concentrations of toxic ionic aluminium are often the primary cause of damage to aquatic organisms (e.g. [179]). In humic lakes and streams, the acid-neutralising capacity, or ANC [186], is a more appropriate measure of the loss of buffering capacity as it allows for the correction of acidity caused by humic acids. In lakes, the change in water chemistry occurs slowly, covering a period of years to decades. Hence a gradual decrease in alkalinity and calcium ions in lake water is a good indicator of ongoing lake acidification. In streams, which generally show a much higher variation in water chemistry than lakes, early signs of acidification include a decrease in alkalinity and higher seasonal fluctuations in pH.

In particular, during high water discharge events, e.g. heavy rainfall or snow-melt, stream water is diluted and shows a drop in pH and alkalinity/ANC. Such peaks in acidity are partly due to the natural dilution that occurs during high water discharge, but can be amplified by an anthropogenic component that is due to ongoing acidification [187]. These so-called acid episodes occur irregularly, on timescale of days to weeks, and may be hard to detect in monthly observations from monitoring programmes. However, although short in duration, these acid episodes may strongly affect the biota in streams, resulting in a loss of sensitive species.

Biological Indices

Several biological metrics, based upon data on macroinvertebrates (e.g. [188–190]) and epiphyton (e.g. [174, 191]), provide measures of the magnitude of acid stress on communities. These metrics apply knowledge of the sensitivity or tolerance of taxa (either family level or species level) to specific metric scores from which the

metrics are constructed. These metrics provide some integration over time and thus overcome the difficulties associated with fluctuating water chemistry. Although users of these biological metrics sometimes erroneously claim to quantify the impact of acidification, they only measure the response of communities to acidity. As these metrics have been developed in different parts of Europe, they probably perform best in the regions for which they have been developed. This is due to differences in the biogeographical distribution of species. In other words, species that occur in certain regions may not occur in other regions, thus inflating the power of the metrics. Moreover, although most metrics originally have been developed for rivers/streams, they also seem to perform well for the macroinvertebrates and epiphyton of lake littoral habitats.

Ecological quality ratios (EQR) are constructed by dividing observed metric values with those of reference values. Reference values for biological metrics can be obtained from populations of lakes/streams of similar water body types that are not affected by acidification or by any other stressors. However, legally reinforced assessment schemes should contain reference values for different water body types.

ASTERICS (including the German package PERLODES) is a European Internet-based software package that calculates a large number of macroinvertebrate-based metrics from uploaded species data files.⁶ ASTERICS therefore provides great help in assessments of inland waters. Acidity metrics based on macroinvertebrates in the ASTERICS software package include the Swedish ‘acid-index’ [188], the German ‘acid-class’ [192] and the English ‘AWIC-index’ [190]. Unfortunately, for other quality elements, no such tool is yet available at the European level.

2.2.2 Confirmation of Acidity as Causation of Degradation

As for all stressors, it should be verified that acidification indeed is a stressor that affects a specific water body before acidification-specific biological metrics and assessment schemes are used, otherwise misclassifications may occur. To avoid this, various approaches can be used to estimate if acidification is a significant stressor for specific waterbodies or populations of lakes and streams in a region.

These approaches differ for point source pollution and those that are driven by long-range trans-boundary air pollutants. The causative link between point source pollution (e.g. mining activities) that leads to acidification and subsequent changes in water chemistry and the biological communities of surface waters is usually straightforward. Similarly, it is often relatively uncomplicated to take measures against point source pollution. Conversely, diffuse pollution sources (e.g. long-range trans-boundary air pollutants) are often harder to link to ongoing acidification, although international agreements and the monitoring of emissions and deposition rates provide good insights into the pressures caused by the global dispersal of

⁶ See: www.fliessgewaesserbewertung.de

acidifying compounds. Whatever the source(s) of pollution, it should be noted that the decrease in the alkalinity (or ANC) of surface waters may start long before biological communities are affected. Few biological effects are usually noticed before critical levels of alkalinity, inorganic aluminium concentrations and pH are reached. In other words, chemical effects of acidification can be seen long before any impact on biota can be detected. Therefore, time series of water chemistry are important tools for the early detection of ongoing acidification.

One major challenge in the assessment of acidification is to distinguish between natural acidity of inland waters and anthropogenically induced acidification (see above). This is particularly important in forested areas where humic lakes and streams dominate, as humic compounds contribute to the natural acidity of lakes and streams. If contemporary monitoring data are used, only the concurrent evaluation of chemical and biological monitoring data can confirm an ongoing acidification process and its effects on biota. Biological metrics of acidification alone may indicate the loss of acid-sensitive taxa, but not automatically provide a causative relationship, as this loss may be due to an increased natural acidity (see above). Therefore, chemical indicators are needed to support the conclusion that changes in biological communities indeed are a result of ongoing acidification. Decreases in pH, alkalinity (or ANC) and concentrations of base cations, as well as a concurrent increase in concentrations of sulphate and metals, in particular inorganic/ionic aluminium, are good chemical indicators of ongoing acidification (e.g. [186]).

The critical load concept also provides a framework for quantifying whether lakes and streams in a certain area are impacted by or potentially at risk from acidification [186]. The critical load of acidity provides a quantitative estimate of the highest load that will not lead to long-term harmful effects on biological communities. The critical load concept includes a critical level, which is defined as the level above (e.g. for inorganic aluminium) or below (e.g. pH, ANC) which no negative effects are found for a given biological indicator (frequently *Salmo spp.*). Henriksen et al. [186] performed an analysis of the critical load of acidity for the Nordic countries and found that between 20 and 70 % of water bodies showed an exceedance of the critical load, indicating that they were affected by acidification. The critical load concept allows for the modelling of lake/stream populations in entire regions and thus for the identification of acidification as a potentially important stressor for these water bodies.

2.2.3 Considerations for Site-Specific Risk Assessments and Downstream Risks

It should be emphasised that most acidity metrics have been designed to quantify a negative change in biota, i.e. going from unimpacted communities to those affected by acidity/acidification. Many European waters are currently recovering from acidification (e.g. [193, 194]). Therefore, future research should study the appropriateness of biological metrics to quantify such recovery. In other words, we

cannot be sure that the metrics that are designed to quantify negative change during acidification also are suitable to quantify recovery.

For streams, dilution models can be used to estimate the contribution of natural acidity to observed episodic declines in ANC and pH [187]. These models estimate the change in pH due to dilution and due to anthropogenic acidification at peak flow and will provide information on the degree to which streams are impacted by acidification and thus help to identify acidification as a stressor.

As well as contemporary data, comparisons of present-day pH with diatom-inferred reconstructions of pre-industrial pH can also be used to assess if and to what degree lakes are subjected to acidification (e.g. [195]). Although diatom-inferred reconstructions have been done for relatively few lakes, within lake-type extrapolations of results could be made to other lakes with similar water chemistry in a catchment or region. The dynamic catchment model MAGIC that provides hind-cast estimates of pre-industrial pH can also be used to assess if water bodies are subject to acidification [196].

2.3 *Salinisation*

Salinity describes the total concentration of all inorganic-dissolved ions in water or soil. Strictly speaking, it is measured by estimating and summing the concentrations of all inorganic ions and is expressed in mg/L [197]. In inland waters, the most important ions in terms of their contributions to salinity are Na^+ , Mg^{2+} , Ca^{2+} , Cl^- , SO_4^{2-} , CO_3^{2-} and HCO_3^- ; in addition, K^+ can contribute to the ionic concentrations of some saline sources.

High salinity is not necessarily the result of anthropogenic processes and many inland saline waters support important ecosystems with many endemic species [198, 199]. Unfortunately, many naturally saline inland waters have been deliberately emptied [200] or polluted, with profound effects on their biodiversity [201].

A wide range of anthropogenic activities can, however, alter the salinity of inland waters, with significant effects on their biota. The salinity of rivers is affected by intrusions of saline aquifers, which in turn are affected by water use, by vegetation in their catchments and by water depth of the rivers. Salinity concentrations in rivers are also increased by mining (e.g. [202]) and industrial discharges (e.g. [203]) and treated sewage effluent. Proposals to recycle treated sewage using reverse osmosis technology will result in a saline effluent which could be disposed of into inland aquatic environments. Imported irrigation waters, even if fresh, carry small amounts of salt. Following evaporation and transpiration, these salts are concentrated. Thus, irrigation in hot and dry climates can lead to salinisation. By changing rainfall and evaporation, climate change also has the potential to alter salinity (see [204]).

With regard to the biological effects of salt, it is known that salt-sensitive riverine invertebrates experience acute (3 days) mortality from salinity at about 5,000 $\mu\text{S}/\text{cm}$ [205–211], while in contrast the most tolerant riverine

macroinvertebrates with recent marine ancestry are able to survive 3 days at up to 70,000 $\mu\text{S}/\text{cm}$ [206]. In general, the most salt-sensitive macroinvertebrates are Ephemeroptera, especially Baetidae, and some non-arthropods such as molluscs, oligochaete worms, flatworms and leeches (although certain members of these groups are actually salt tolerant). Some species of chironomids are also highly salt sensitive, but this large family encompasses members with a very broad range of salinity tolerances. Plecoptera and Trichoptera are in general slightly more salt tolerant than are most Ephemeroptera.

In general, freshwater fish are more tolerant to elevated salinity than invertebrates [212]. However, freshwater fish sperm, and their eggs before hardening, can be extremely salt-sensitive, even in species with salt-tolerant adults (e.g. [213–215]). The direct effects of salinity on freshwater fish populations will thus presumably depend upon whether periods of high salinity coincide with reproduction. Likewise, the salinity tolerance of invertebrate eggs is generally less than that of older life stages [209, 216, 217], though the differences in salinity tolerance between eggs and older life stages are lower than for fish.

For longer periods of salt exposure, invertebrates are more sensitive, depending upon exposure conditions [218]. Not surprisingly, sublethal concentrations of salinity have been observed to affect the development, growth and reproduction of a range of invertebrate species. Most invertebrate species examined have greater growth and/or reproduction rates at intermediate salinities (500–1,000 $\mu\text{S}/\text{cm}$) compared to low and high salinities [219]. That is, their salinity concentration–response curve follows an inverted ‘U’ rather than a threshold below which changes in salinity do have any effect [212]. This inverted ‘U’ response curve can be explained by a response to essentiality, which results from deficiencies in essential ions and/or greater expenditure of energy on osmoregulation. An inverted ‘U’ response has important implications, as increases and decreases in salinity may have effects on individual invertebrates, with the potential to propagate to their populations and communities. A number of studies have shown that macroinvertebrate community structure is highly correlated with salinity (e.g. [203, 220–224]; some families have a greater probability of occurrence at intermediate salinities [225] and there would appear to be an increase in macroinvertebrate species richness at slightly increased salinity [208, 226]). However, whether these field effects observed changes are caused by adverse effects of low salinity on freshwater invertebrates is uncertain. In contrast, the richness of Ephemeroptera, Plecoptera and Trichoptera species (which tend to be salt sensitive) decreases with increasing salinity across the entire salinity range [226]. There are few studies on the effects of salinity on functional indicators, but recently Schäfer et al. [227] found that salinity reduced the breakdown rate of allochthonous organic matter by both invertebrates and microbes, whereas it had no effect on gross primary production or net ecosystem respiration.

2.3.1 Evaluation of Surveillance Monitoring Data

High salinity concentrations are well known to cause effects on macroinvertebrate communities. However, it is not known whether the stimulation that individuals of many species of invertebrates get from slight increases in salinity has any effects at the population and community levels. Determining if there are population- and community-level effects at both ends of the salinity spectrum may have important implications.

Abiotic Indicators

Since quantification of all single ions is labour- and time intensive, salinity is often measured indirectly as electrical conductivity (EC). Salinity is present in all natural waters and the salinity of inland waters naturally varies from lowest EC recorded levels of 13 $\mu\text{S}/\text{cm}$ [228] to $>400,000 \mu\text{S}/\text{cm}$ [197], i.e. four orders of magnitude difference. Compared to the measurement of all single ions, EC has the added advantage of being able to be measured instantaneously in the field and to be logged remotely. Although EC is affected by both the identity of ions and the total concentration of ions, there is usually a strong correlation between salinity and EC [197]. Moreover, the EC of water of a given salinity is influenced by temperature and is thus usually referenced to a standard temperature of 25 °C in $\mu\text{S}/\text{cm}$, with 1 $\mu\text{S}/\text{cm}$ at 25 °C equals to about 0.6–0.8 mg/L ions, depending on ionic proportions. All values of EC mentioned here refer to 25 °C. There are several other indirect methods to determine salinity which are less common and are discussed in William and Sherwood [197].

Biological Indices

Impacts of salinity on stream macroinvertebrates (and other groups) can be defined in terms of departures of community compositions from reference conditions (e.g. [220, 221]), according to the requirements of the WFD. In order to identify these departures, the macroinvertebrate communities at a series of reference sites are compared to those at the sites with elevated salinities. The problem with such an approach is that salinisation can affect relatively large regions, and it is thus difficult to identify nearby sites that represent pristine conditions that can serve as reference sites [229]. Furthermore, salts are natural constituents of all waters and vary widely in space and time. Thus it can be difficult to determine what should be the salt level in a particular water body in the absence of human activities.

A number of indices have been suggested to identify the impacts of salinity on stream macroinvertebrate communities. Most of them are based on the relative occurrence of salt-sensitive and salt-tolerant taxa. For example, a salinity index (SI), depending on the presence/absence of macroinvertebrate families at sites of

different salinity, has been proposed. This salinity index is based upon analyses using neural networks, which assigned each family a salinity sensitivity score (SSS) where 1 refers to very tolerant, 5 to generally tolerant or 10 to sensitive families, respectively. The SI of a site is then given by the mean SSS of all families present at a site. Subsequently, Horrigan et al. [230] modified SSSs to use both the occurrence of families and 72 h LC₅₀ values [211]. More recently, the SPEAR (SPECies-At-Risk) concept, which was originally developed for the detection of pesticide stress [46, 47], has been extended to the effects of salinity [227]. Initial findings are promising as they show that SPEAR_{salinity} is relatively stressor specific in the identification of the effects of salinity. SPEAR_{salinity} is salinity specific in that it is correlated with salinity but not with other environmental variables that may potentially cause ecological status degradation [227]. This is unlike the aforementioned SI and several other biotic indices, which while correlated with salinity were also correlated with other of such environmental variables [227].

2.3.2 Confirmation of Salinisation as Causation of Degradation

If the surveillance monitoring data evaluation suggests salinity as a potential cause for degradation of the ecological status, investigative monitoring should be designed in a manner that may confirm salt pressure and identify its sources. Given salinity as EC is very inexpensive and instantaneous to measure, identification of salt sources that lead to elevated levels in a river will usually be relatively straightforward by making a series of EC measurements, which can be made in longitudinal direction along a river and from tributaries, over space and time.

To predict effects of salinity on invertebrates, a concentration–response relationship is needed. As with other toxicants, such community-level dose–response relationships can be constructed using species sensitivity distributions (SSDs) which rely on the sensitivities of numerous taxa. There is a tendency of riverine macroinvertebrate species belonging to the same family or even order to have similar 72 h LC₅₀ values, even when species are collected from different geographic regions [206–211]. However, the composition of macroinvertebrate communities in different regions differs due to differences in environmental conditions, biogeography and biotic interactions. Thus, in regions where salt-tolerant taxa dominate, communities will tend to be more tolerant to salinity changes [207, 211]. In Australia, to derive environmental quality guidelines, Kefford et al. [219] therefore suggested a modification of the SSD concept, where a local SSD is constructed based on the average salinity sensitivity of each family known to occur within the region of interest. To account for particular species richness of families, the authors [219] advocated the weighting of each family in the SSD based on the number of member species recorded from the region of interest. This results in an SSD which should reflect the macroinvertebrate community known to occur in the region. Hickey et al. [231] developed this approach further with the use of Bayesian statistical models to produce an SSD that reflects the macroinvertebrate community present in a region of interest.

2.3.3 Considerations for Site-Specific Risk Assessments and Downstream Risks

Not All Salts Are the Same: Effects of Different Ionic Compositions

As previously mentioned, salinity is the sum of the concentrations of all dissolved inorganic ions, but principally of Na^+ , Mg^{2+} , Ca^{2+} , K^+ , Cl^- , SO_4^{2-} , CO_3^{2-} and HCO_3^- . Different proportions of these ions have different toxicities even when salinity is constant (e.g. [232]), and the theoretically possible combinations of these ions are infinite. Hence, small differences in ionic proportions can potentially result in large differences in effects of saline waters on biota despite having the same (total) salinity or EC.

Zalizniak et al. [233, 234] for five invertebrates and one protozoan species investigated the relative toxicities of four test solutions with different ionic compositions (or saline water types) common in southern Australia. They found that all four had similar effects on acute (≤ 96 h) mortality but that water types with low Ca^{2+} concentrations increased chronic toxicity. Kefford et al. [216], in contrast, observed that NaCl solutions resulted in greater acute mortality in South African and Australian invertebrates than seawater of the same salinity, despite the salinity of seawater being mostly composed of NaCl. Thus, the other ions in seawater were reducing the toxicity of NaCl by unknown physiological mechanisms. Despite this difference in toxicity between NaCl and seawater, the LC_{50} values of species to both salt sources were correlated, i.e. species that are sensitive to NaCl also tend to be sensitive to seawater. If this observation holds true for salts with other ionic compositions, then, while the actual effect levels for salinity may differ, this would indicate that the relative sensitivities of taxa are similar, i.e. taxa sensitive to one salt composition are also sensitive to another composition and conversely for tolerant taxa. Hence, this would ameliorate the management of different ionic compositions by just testing a subset of reference species and use them as benchmark organisms for the setting of ecological quality standards in regulation.

Effects of Temporal Variation in Salinity

The salinities of rivers are not constant but vary depending on various water and salt balances. In the case of the disposal of saline effluents, discharges occur episodically, resulting in almost instantaneous salinity changes. During seasonal drying, salinity tends to increase over weeks to months. Although it has been poorly researched, different exposure patterns in salinity may have different effects despite similar mean or maximum salinities. A number of studies have shown that slow rises in salinity (over a period of days) are less stressful to a range of fish species than are faster rises in salinity [212, 214]. In a 5-day mesocosm experiment, Marshall and Bailey [235] compared constant salinity exposure at 1.5 g/L and four pulses of salinity at 3.4 g/L but with a similar (1.4 g/L) mean salinity over

the 5 days. They found a greater effect of the pulsed exposure on stream macroinvertebrates than for the constant exposure. Nielsen et al. [236] compared the effects of gradual rises in salinity (to 1 and 5 g/L) over 6 months with constant salinity of the same concentrations over 6 months on wetland zooplankton (microinvertebrates) and macrophytes, finding similar effects in both communities at the end of the experiment regardless of the time course of salinity increase. In conclusion, the effects of temporal salinity exposure patterns on communities are poorly understood but are potentially important, and effects may differ depending on the organisms and time scales involved.

Do Freshwater Systems Recover from the Effects of Salinisation?

Relatively little work has been done on the reversibility of salinisation. However, recovery will likely depend upon the source of the salinisation. The ions that make up salinity do not break down, and in extremely salinised landscapes, the supply of salts cannot be simply turned off. The amounts of salts that can build up in soil and groundwater are enormous and even if addition of new salts can be stopped, it can take centuries for salts to be flushed from the landscape. In contrast, where salts from industry or mining discharged into rivers from point sources are terminated, recovery should be much more rapid. This is supported by Busse et al. [237] who observed that following the ending of the disposal of saline effluents associated with potassium mining in Germany, the benthic diatom communities changed within 3 months (although not necessarily to their initial state before the commencement of disposal).

Furthermore, in extensively salinised regions, unsalinised sections of rivers with populations of salt-sensitive species are likely to be distant, so even if salts can be quickly removed, it may take some time before sensitive species recolonise desalinised reaches. Again, in cases of point source discharges of salts to rivers, there are likely to be unsalinised upstream reaches or tributaries to supply colonists.

Another factor which will influence recovery of communities is the degree to which in-stream and riparian vegetation is affected by salinisation. Riparian vegetation can obtain a significant proportion of its water directly from the adjacent water body, and adverse effects of high salinity on riparian vegetation have been documented [238]. In stream, many species of macrophytes are also salt sensitive [214]. Riparian and in-stream vegetation provides important habitat and food for macroinvertebrates and effects on the vegetation may propagate to the invertebrate community. So, the recovery of invertebrate communities following desalinisation will (in part) depend on the recovery of riparian and in-stream plants. In the case of large riparian trees being killed as a result of salinisation, it is likely to be some decades for them to be replaced following desalinisation.

Finally, Piscart et al. [222–224, 239] found that increased salinity was associated with an increased abundance of invasive amphipod species and suggested that elevated salinity might foster the introduction of invasive species (see Sect. 2.6). The recovery of macroinvertebrate communities following desalinisation may thus

depend on the degree to which salinisation allows introduced species to gain a ‘toehold’ in systems where they would otherwise not be able to establish viable populations.

What Are the Joint Effects of Salt with Other Stressors?

Laboratory toxicity experiments show that salinity tends to decrease the toxicity of most metals but increases the toxicity of organophosphorus insecticides [240]. For most other toxicants there appear to be no unambiguous effects of salinity on toxicity, as their toxicities may increase, decrease or not respond to elevated salinity. Nevertheless, it should be noted that these laboratory experiments have tended to look at the effect of toxicants within salinity ranges that do not stress the test species. There is a paucity of research examining the combined effects of salinity and toxicants when both occur at potentially stressful concentrations.

In Australia, there has been concern that elevated salinity may co-occur with low (acidic) pH (e.g. [241]). While effects of acidification are well documented (see Sect. 2.2), laboratory experiments with four freshwater invertebrates and a protozoan species showed that low pH values (but within the physiological tolerance of the species being studied) did not alter salinity toxicity [242]. However, the protozoan, *Paramecium caudatum*, was more salinity sensitive at high (10–11) pH values.

Moreover, Ca^{2+} and Mg^{2+} from saline waters can increase the settling rates of fine suspended particles [243]. Therefore, saline water inputs into a sediment-laden river may cause suspended sediments to be deposited [244] and bury organisms and habitat. Although it has not been explicitly studied, there thus exists the potential for salinity to be an important influence on the effect of suspended sediments (see also Sect. 2.9) on freshwater biodiversity.

2.4 Eutrophication and Oxygen Depletion

Eutrophication of running waters caused by high nutrient loading is one of the most widespread water quality problems worldwide [245]. Organic pollution rapidly declined during the last 20 years in Central Europe due to the establishment of wastewater treatment infrastructures. As organic pollution thus declined, anthropogenic eutrophication with increased autotrophic production became more and more a key problem in water quality management [246]. Recent inventories rank this problem as a major reason for failing the environmental objectives for the majority of running water bodies in EU member states [246, 247].

Once nutrient supplies exceed limiting values, increased primary production causes the detrimental effects of eutrophication. In general, eutrophication acts as a stressor by increasing the biomass within aquatic ecosystems with subsequent oxygen demand for increased biological turnover. This may result in complete

shifts of the autotrophic/heterotrophic balances in surface waters [248]. More specifically, plankton blooms, large streamers of filamentous algae or macrophyte standing stocks, potentially lead to low-oxygen events at night, alter the community structure, slow water flow, clog sediments and lower hyporheic exchange [249, 250]. The increasing photosynthesis of primary producers (algae and macrophytes) at daytime and the resultant respiration at night causes high diurnal changes in the physicochemical conditions of the surface waters and even water–sediment interfaces. Well-known consequences are high pH levels at daytime and low dissolved oxygen at nighttime in the vegetation period. The most important consequences comprise alterations of water quality, such as labile oxygen balances (with either excessive supersaturation or anoxia), pH fluctuations and mobilisation of harmful substances under anoxic conditions. Another well-known phenomenon is phosphorus release from sediments of eutrophic lakes with the consequence of excessive internal nutrient loading and enhanced eutrophication [251]. Excess amounts of nutrients, stratification and warm temperature in stagnant waters may stimulate adverse biological effects, e.g. algal blooms of several cyanobacteria species that are also known to produce and release hepatotoxins. Moreover, increased pH levels interfere with other water quality constituents via chemical equilibrium. Another well-known phenomenon is the pH-controlled balance between ionised and non-ionised ammonia ($\text{NH}_4^+/\text{NH}_3$). While NH_4^+ is not toxic to aquatic biota even at high concentrations, NH_3 is neurotoxic to fish at comparable low doses (acute toxicity at levels around 0.1 mg/L).

The biological consequences of eutrophication are diverse, including shifts in species compositions, biodiversity or functional patterns of communities. The excessive growth of algae and macrophytes have often been associated with eutrophication events [252], which are ubiquitous phenomena in nutrient-rich flowing waters [249]. However, the effects of eutrophication on macrophytes in flowing waters have been poorly studied, and the effects of nutrient reductions on macrophyte biomass are difficult to predict [253]. Biomass of macrophytes may either decline or increase, and the nutrient support of rooted macrophytes may be controlled by porewater biogeochemistry.

Eutrophication may cause a series of physical, chemical and biological changes. In general, biota must also cope with an increased variability of environmental conditions reaching or even exceeding their physiological tolerance limits. For example, the excessive growth of plankton algae favours active and passive filter feeders via the provision of particulate organic matter. As a consequence, the functional composition of communities at higher trophic levels shifts when compared to oligotrophic reference conditions. Some filter-feeding macroinvertebrates may exhibit mass developments which are harmful to cattle or even human populations (e.g. via mosquitoes that start their life cycle as egg > larvae in water). The excessive growth of benthic algae, such as the ubiquitous green algae *Cladophora*, may lead to comparable problems via benthic food webs.

2.4.1 Evaluation of Surveillance Monitoring Data

The trophic state of surface waters may be divided into heterotrophic (consumption of organic carbon resources as energy source) and autotrophic (production of organic carbon using sunlight) states [245]. This division allows the consideration of whole-stream metabolism as a conceptual framework for trophic assessments including the effects of external organic loads (usually from wastewater) as well as nutrients (usually nitrogen and phosphorus). The balance of heterotrophic respiration and autotrophic production has been considered as a fundamental driver of longitudinal zonation patterns of rivers and streams with characteristic shifts from upstream to downstream regions and changing relevance of external nutrient loads relative to internal bio-production [254]. Furthermore, the ratio between the width of the epilimnion (the productive zone) and hypolimnion (the respiratory zone) of stagnant waters is a fundamental factor that determines their trophic state. As a consequence, the natural background or reference trophic states of surface waters have specific characteristics which must be considered in assessment methodologies.

In eutrophication management and monitoring, there are a series of indicators and indices which indicate significant eutrophication risks within a catchment [255].

Abiotic Indicators

There is a wide range of options for eutrophication monitoring via catchment characteristics and traditional physicochemical parameters. The monitored parameters are organic load (biological oxygen demand after 5 days (BOD-5), chemical oxygen demand (COD), total organic carbon (TOC)) and total suspended solids. For discharges to sensitive areas which are subject to eutrophication, total phosphorus and total nitrogen may be considered. Operational monitoring encompasses extended sets of water quality parameters, including continuous measurements of oxygen, pH and chlorophyll. Due to the fluctuating nature of these constituents under eutrophic conditions, long-term measurements may be necessary in order to capture the full range of environmental states. These may be compared to existing reference conditions or use-specific water quality standards and can be documented in national or regional databases and reports.

Biological Indices

Based on the observation that the composition of macroinvertebrate communities changed over a gradient of different water quality classes following a distinct input of wastewater effluents, Kolkwitz and Marsson [256] introduced in 1909 one of the first biological indicator systems, the so-called saprobic system. It was designed to

indicate high organic loads, based on indicator species that are sensitive to a reduction in the availability of oxygen, associated with the input of organic loads. In the following decades, the corresponding saprobic index was further developed, which became nowadays the most important index to assess the ecological status with regard to macroinvertebrates [257].

However, under the EU WFD, four groups of biological quality elements (besides macroinvertebrates also diatoms, macrophytes and fish) have been suggested for the classification of the ecological status of a particular water body [258]. Diatoms and macrophytes are considered to be especially suitable in the context of eutrophication. Stevenson [259] reviewed the uses of diatoms for the assessment of environmental conditions in rivers and streams. They suggested the use of species compositions of algae communities to assess the amount of organic pollution. Studies that use algal assemblages as indicators of the extent of pollution rely on the concept that predictable species shifts occur with defined amounts of nutrient enrichment [260].

Finally, macroinvertebrates and fish may also be included in more specific analysis of biological effects of eutrophication [258]. For example, shifts in the functional compositions of macroinvertebrates, such as feeding types, passive or active filter feeders and the representation of hydraulic or habitat preferences, may indicate eutrophication effects on community levels. Furthermore, the age structure of fish communities and the lack of reproduction of gravel-spawning fish may also be closely linked with adverse eutrophication effects [261].

2.4.2 Confirmation of Eutrophication as Causation of Degradation

The confirmation of adverse effects caused by eutrophication in a multi-stressor analysis may consider a wide range of hydraulic, physical, chemical and biological effects. One of the problems in predicting eutrophication effects in streams is that variability caused by flooding can influence the autotrophic state considerably. At one extreme, algal biomass might not be able to accumulate, even with ample light and nutrients, if floods frequently detach periphyton biomass [262]. At the other end of the spectrum, attached algae may be able to attain high biomass in nutrient-poor waters because periphyton can use the small amounts of nutrients that continuously flow by [262, 263]. In particular, the work of Biggs [263] supports the proposition that eutrophication effects will be stronger under stable flow regimes. Therefore, in investigative monitoring, there is a need to quantify flow conditions including residence times, shear stress and the thresholds for riverbed scouring as time series in relation to autotrophic biomass development.

Excessive growth of plankton and benthic algae may decrease exchange processes between the surface water and the hyporheic zone [264]. As particulate and dissolved material is transported into streambed interstices, it may be temporarily stored or metabolised by benthic biota. Recent studies indicated that the intrusion of detached periphyton biomass contributed significantly to the sealing of the hyporheic interstitial and caused significant oxygen depletion in the hyporheic

zone while oxygen levels remained close to saturation in surface flow [250]. These patterns may be studied by specific methods in spatially distributed arrangements with reference to the local conditions (see [264] as an example).

With regard to stream metabolism, eutrophication effects may be confirmed by determination of primary production, community respiration and biomass change in the algae or macrophytes along river stretches or for stagnant water bodies. The diurnal oxygen change method for running waters, established by Odum [265], may be used as fundamental basis to estimate stream metabolism [266]. The method allows for the assessment of primary production (P), community respiration (R) and production to respiration ratio (P/R). New generations of water quality modelling [267] provide tools for a further separation of underlying biochemical processes, including food web interactions [268].

Investigative monitoring for lakes may consider the spatial heterogeneity of environmental conditions, especially in larger stagnant waters, and include stratifications. Therefore, monitoring networks must be implemented for the determination of lake-wide distributions of physicochemical conditions, phytoplankton, zooplankton and fish with complementary modelling (e.g. [269]). Moreover, stagnant waters may have a considerable inter-annual variability of environmental states that are triggered by meteorological and hydrological conditions. Therefore, appropriate time series may be needed for cause–effect specific analyses.

2.4.3 Considerations for Site-Specific Risk Assessments and Downstream Risks

A risk assessment for eutrophication impacts may be based on catchment-wide analyses of land use, including sources and pathways of carbon, nitrogen and phosphorus. This information may be used for assessments of nutrient loading at basin-wide scales [270] or eutrophication modelling of stagnant [271] and running waters [267].

Whereas wastewater treatment plants and industrial sources are directly discharged into rivers, diffuse emissions into surface waters are caused by the sum of different pathways, which are characterised by separate flow components. The separation of components of diffuse sources is necessary, because nutrient concentrations and relevant processes for the pathways are catchment specific. Consequently, different pathways must be considered including (1) atmospheric deposition, (2) erosion, (3) surface run-off, (4) groundwater, (5) tile drainage and (6) paved urban areas. Along the pathways from the sources of the emission into the river, substances are governed by manifold processes of transformation, retention and loss. Knowledge of these processes of transformation and retention are necessary to quantify and to predict nutrient discharges/losses into the rivers in relation to their sources.

Local eutrophication risks and downstream effects may be linked in coupled running stagnant water systems, especially if rivers are impounded or discharge into

lakes or reservoirs. In these cases, extended assessments including empirical, conceptual or mechanistic eutrophication models [269, 272, 273] are necessary.

The problem of eutrophication might further increase in the near future under the conditions of climate change [274]. Although there is considerable uncertainty with respect to the relationships between temperature increase, precipitation and water run-off on catchment scales, climate change scenarios indicate that the seasonal distribution, frequency and duration of rainfall patterns and droughts will change significantly [275]. Thus climate change will alter the water cycle and associated biogeochemical cycles on water body scales and therefore a series of factors, interfering with eutrophication.

2.5 Parasites and Pathogens

The ecological status of aquatic systems depends upon biological, hydrological, morphological, physical and chemical factors. Among the biological factors, parasites and pathogens are of major importance. *Pathogen* is a general term for disease-causing biological agents, whereas the term *parasite* designates eukaryotic species which utilise a host organism as habitat as well as energy and nutrient source. The relevance of pathogens and parasites in relation to ecological status assessment is based on three aspects: (1) pathogen and parasite infections may negatively affect vital rates and population growth of the host species, (2) pathogen and parasite infections may increase the sensitivity of the hosts to toxic stress and vice versa and (3) parasites may serve as bioindicators of stressed fish populations.

Parasites and pathogens can severely reduce survival and fitness of the hosts. Although the host species may have developed elaborate immune defences to combat infections, the defence response is energy costly and leads to physiological trade-offs between fitness-related traits, such as survival, growth or reproductive efforts [276–278]. Further impairment of fitness results from pathogen-induced tissue damage and associated physiological dysfunction. Consequently, parasites and diseases play important roles in population dynamics of the hosts, and in extreme cases, they can result in their extinction [279]. An example of severe adverse effects of parasites on aquatic wildlife populations is the decline of salmonid populations in Northern America caused by the whirling disease, which is caused by the myxozoan parasite *Myxobolus cerebralis* [280]. Other examples include the decline of European eel population due to infection of the swim bladder by the trematode parasite *Anguillicola crassus* [281]; the decline of brown trout populations in Swiss rivers related to a kidney disease caused by the myxozoan parasite *Tetracapsuloides bryosalmonae* [282, 283]; the collapse of black abalone populations along the Californian coastline during the late 1980s caused by a *Rickettsia*-like organism [284] and the pathogen-related mass mortalities in many European seal populations [285].

In a stable environment, co-evolution eventually leads to equilibrium in host–pathogen relationships, supporting persistence of both the host and the

pathogens. However, in the presence of other environmental stressors, this balance can be disrupted, with several possible qualitative outcomes [284]. For instance, in the case of the parasite, the additional stressors may reduce its abundance, either by directly killing the parasites or by changing the abundance of the hosts, i.e. the living habitat of the parasites. Another outcome could be that an additional stressor makes a host more susceptible to a parasite. This can result in enhanced abundance and diversity of parasites living on a host, as shown in a number of field studies which observed higher intensities of gill infection with parasitic ciliates in fishes living in polluted waters. Further, the combined action of parasites and additional stressors can lead to a disproportionate increase in mortality rates of infected hosts, as their adaptive capabilities may be surpassed [284, 286, 287].

In polluted aquatic environments, pathogen- or parasite-induced diseases may appear to be the proximate cause of adverse ecological changes. However, the ultimate cause may be a pollution-induced impairment of pathogen resistance of the host. Exposure of organisms to sublethal concentrations of pollutants increases their susceptibility to pathogens, while diseased organisms are more sensitive to the toxic action of chemicals [288–290]. An excellent example of synergistic, antagonistic and additive effects between parasites and toxicants is provided by the study of Coors and DeMeester [291] with *Daphnia magna*. A further example is the enhanced intensity and prevalence of fish gill infection with ciliates as has been reported in various field studies on pollution-exposed fish [292, 293]. Structurally diverse chemicals, such as tributyltin, polychlorinated biphenyls, benzo(a)pyrene and estrogenic compounds, are able to reduce fish resistance to bacterial and viral pathogens [294–297]. In amphibians, toxic impact has also been shown to result in enhanced susceptibility to pathogenic infections, as exemplified, for instance, by the study of Kiesecker [298] on the synergism between trematode infections and pesticide exposure in inducing amphibian limb deformities, the study of Christin et al. [299] on increased parasitic infections of pesticide-exposed *Rana pipiens*, the study of Johnson et al. [300] on the promotion of pathogenic infections of amphibians by aquatic eutrophication and the study of Rohr et al. [30] on increased trematode infections in amphibians exposed to agrochemicals.

Toxicant effects on the host resistance to pathogens can also be mediated via toxicant effects on the host immune system. A large number of environmental pollutants are able to modulate immune parameters in exposed organisms, be it at the system, cellular or molecular level [297, 301, 302]. However, in many cases, the mechanisms are not understood nor is it known whether or under what conditions the toxicant-induced immunomodulation translates into adverse effects on host fitness. Another link between toxic and parasite impacts on organism may come from physiological trade-offs. Immunological responses are energetically costly [276–278], as are responses to toxic stress [303]. Thus, the organisms may decide to suppress the immune response in favour of adaptive responses to toxic impacts. Parasites are ubiquitous components of ecosystems and appear to be sensitive to a wide range of toxic chemicals. Furthermore, parasites move through the food web and are situated at all trophic levels. Finally, changes in parasite diversity and abundance in polluted waters integrate both the direct toxicant effects on the

parasite community and the indirect toxic effects via modulation of the hosts' physiology. In fact, in a number of field monitoring studies, parasites were successfully used as indicators of chemical contamination [281, 304]. An example is provided by the study of Dusek [305], who observed that in a polluted freshwater environment, assemblages of specialist parasites exhibited significantly reduced species richness and unequal distribution of abundances, while the opposite pattern was observed for generalists. Schmidt et al. [306] showed that parasitological parameters of flounders from the North Sea differed significantly in accordance with the pollution status of the sampling sites.

2.5.1 Evaluation of Surveillance Monitoring Data

Above, we have considered pathogens and parasites as potential causes of degradation of river basin ecology, either alone or in combination with other stressors. However, parasites themselves can also be affected by stressors such as environmental pollutants. Thus changes in parasite abundance or diversity are of potential value as indicators of environmental stress [281, 292, 307]. The use of parasites in water quality and pollution assessment is comparable to the use of, e.g. macroinvertebrates, and parasite species can be incorporated into routine (surveillance and operational) monitoring systems with appropriate sampling modifications. In addition to the ecological and epidemiological importance of freshwater parasites, there are many other reasons for their use as indicators of environmental pollution [305].

Basically, the same diagnostic methods can be applied that are used to characterise infection status of hosts, but they should be extended with parameters providing information on the ecological and community status of the parasites (see [305, 306]). These include parasite life history strategies (e.g. specialists versus generalists or ecoparasites versus endoparasites), guild structure (e.g. parasite inhabiting gills, livers or intestines), as well as standard parameters for describing species richness and heterogeneity (e.g. Shannon index, Brillouin index, dominance). Effect magnitude of pollution on parasite communities can be estimated via parameters such as Cohen's d (ESd), which represents the difference in mean parasite abundance between reference and contaminated sites, divided by the pooled standard deviation or standard error [307]. From a recent in-depth literature review, Blonar et al. [307] concluded that monogenean and digenean parasites (both taxa are member of the flatworm phylum Platyhelminthes) are promising biological indicators of pollution in aquatic ecosystems. So far, however, policy frameworks such as the WFD do not yet take advantage of the indicator value of parasites for ecological status assessment [3].

2.5.2 Confirmation of Parasites and Pathogens as Causation of Degradation

If the question is whether pathogen or parasite infections cause impairment of a host population at a given site, the following methods can be employed for their detection. Parasites can be identified using routine techniques of macroscopic inspection and/or by microscopic examination of the hosts. To demonstrate infection with viral and bacterial pathogens, diagnostic methodologies include the propagation of viruses on fish cell cultures and subsequent immunochemical identification of the species, direct serological detection of viruses and bacteria as well as indirectly via host-generated antibodies against the pathogens or PCR (polymerase chain reaction) determination of specific mRNA or DNA sequences of pathogens and parasites. If the infective agent has been previously described in the literature, published information on disease signs will help to identify the pathogen or parasite and to select the appropriate diagnostic methods. Diagnosis of pathogens and parasites can be done qualitatively, i.e. 'present/not present'. It can also be done quantitatively, i.e. prevalence, intensity and, in the case of parasites, also abundance. Abundance can be defined here as number of parasites divided by total number of host individuals, thus integrating both prevalence (percent of hosts infected) and intensity (number of parasites per infected host).

In order to judge if an infection is causative to impaired population status of the host, information is needed on the virulence of the pathogen or parasite to the host species. Such information may be available from previous ecological or veterinary research. If not, experimental studies are needed to assess the virulence of the disease agent and to study the conditions of virulence (e.g. temperature dependence, impact of other stressors, etc.).

Additional complications in evaluating a causative role of pathogens and parasites in impairment of host populations at field sites come from the interaction between infectious agents and other stressors. As, for instance, immunosuppressive toxicants can enhance the susceptibility of hosts towards pathogens and parasites, the latter may appear to be the ultimate cause of population decline, although in the absence of the co-stressor, the host may well tolerate the infection. Examples of such cumulative effects between pathogenic stressors and other stressors have been discussed above. In such a multiple-stress situation, a phased approach using conceptual models, statistical models and/or weight-of-evidence approaches might be used to sort out the relative role of the various stressors ([23, 308, 309], see also [5], this volume).

A recent example of an investigative study illustrating the ambivalent role of parasites as both stressors and pressure indicators may be provided by the study of Wenger et al. [310] on the Bilina river in the Czech Republic. The Bilina river demonstrates low fish abundance and diversity. As the river is contaminated by a number of pollution sources, toxic chemicals were considered as candidate cause for this decline. Another possible cause was infection with parasites. The results of the field study showed that the low fish abundance and diversity did not show a

correlation with pollution nor with parasite pressure. Interestingly, however, parasite diversity and abundance was lowest at the river sites with highest contamination, so that the parasite status appeared to be an indicator of water pollution [310].

2.5.3 Risk Assessment of Local Impacts and Downstream Risks

In terms of risk assessments, negative impacts from parasite or pathogen infections on population growth may be estimated by using mathematical models which simulate the translation of parasite-induced alterations of vital rates in hosts to alterations in demographic values of host populations. Examples of such applications are given in the studies of Arkosh et al. [311] and Springman et al. [286]. This approach is rather straightforward as long as the parasite infection is the major driver of ecological change. The situation becomes more complex when the impacts of parasites on hosts are not direct functions of the parasites' virulence, but arise from the interactions between pathogenic and other stressors. Approaches to such multiple-stressor scenarios are further discussed in chapter by von der Ohe et al. [5], this volume.

An important factor to be considered in assessing risks arising from infective agents is the spreading and distribution of pathogens and parasites within river systems [312]. This depends on the biological traits of infective agents and hosts, but also on the hydrology and morphology of river basins. For instance, if migration barriers exist in a river system, this may restrict the spreading of pathogens between local populations of hosts. For parasites, the distributions of intermediate hosts are also key determinants for the spreading of diseases.

Anthropogenic environmental factors that are able to modulate host–parasite interactions include introduction of invasive alien species (see Sect. 2.6), habitat alteration, pollution or climate change. An example of the role of climate change is provided by the infection of brown trout populations in Swiss rivers with the myxozoan parasite *Tetracapsuloides bryosalmonae*, the cause of the proliferative kidney disease (PKD). This disease is widespread among brown trout populations in Switzerland, particularly in river stretches below 800 m altitude [266]. Brown trout infected with the parasite experience no or very low mortality at water temperatures below 15 °C, whereas mortalities increase drastically at water temperatures above 15 °C [313]. As rivers in Switzerland have experienced an elevation of water temperatures during recent decades, summer temperatures in many rivers are now frequently higher than 15 °C, particularly in the river stretches below 800 m altitude, i.e. in those stretches where PKD is most prevalent. This example may illustrate how an environmental change—in this case, the increase of water temperature—can alter a balanced parasite–host relationship, leading to increased mortality rates of the host. Hence, important additional information for risk assessment is how parasite virulence changes with environmental factors, but it may also vary with the host physiology; e.g. under conditions of starvation or reproduction, the host will be more susceptible to the parasite impact [314].

2.6 Invasive Alien Species

Invasive alien species (IAS) are species, subspecies or lower taxa that are introduced outside their natural past or present distribution. This includes any part such as gametes, seeds, eggs or propagules of such species that might survive and subsequently reproduce. ‘Introduced’ refers to the movement by human activities, indirect or direct, of an alien species into new environments [315]. IAS are those species whose establishment and/or spread threatens biological diversity and may cause economic or environmental harm. The introduction of IAS with noticeable effects on individuals, populations and communities of native species and/or resulting in adverse socioeconomic consequences can be defined as biological pollution [316, 317]. However, in most cases, impacts of alien species, specifically in aquatic ecosystems, are not easy to identify and biological contamination can be a more applicable term for the introduction of alien species ([318]; see also [317] for terminology applicable in risk analysis of IAS).

Records on IAS clearly show that their introduction into European inland waters greatly increased during the second half of the twentieth century [319]. This rise was likely fostered by the worldwide increase in mobility and trade following the Second World War. Gherardi et al. [320] showed that a wide range of human activities were responsible for alien animal species’ expansion in European inland waters. These activities, in decreasing order of importance, are extensive fish culture and sport fishing (30 %); intensive aquaculture (27 %); transportation by vessels (25 %), ornamental use, i.e. with introduction into lakes on private estates, small garden ponds and indoor aquaria (9 %); and dispersal through canals (8 %). The establishment of the European network of inland waterways greatly facilitated the dispersion of IAS into principal European river basins. Hence, during the last decade, the spread of aquatic IAS via canals and shipping were the most important pathways for their dispersion in large European river basins with 36 % (canals) and 34 % (shipping) of new introductions, respectively [317]. Ornamental trade (10 %) and fisheries and aquaculture (8 %) were to a lesser extent the pathways for new introductions.

IAS are considered a direct cause for biodiversity loss over the last century, with IAS considered to be the third most important threat to biodiversity of inland waters after hydromorphological degradation and point source and diffuse pollution [1]. Impacts on biodiversity may result from five principal interactions with native species, which frequently lead to decreased numbers and even displacement of native taxa. These are (1) resource or interference competition (e.g. *Dreissena polymorpha*, [321–323]), (2) predation (e.g. alien amphipods, [324–326]), (3) transmission of alien diseases or parasites (e.g. crayfish plague, [327]), (4) habitat modification and change of ecosystem functioning (e.g. *Chelicorophium curvispinum*, [328]; or *Procambarus clarkii*, [329]) and (5) introgressive hybridisation with closely related native species (e.g. alien fish, [330]). For other case studies of IAS impacts on biodiversity, see [331] and, specifically, Orendt et al. [332].

Biodiversity loss in aquatic ecosystems due to IAS introductions is generally irreversible [333, 334]. In addition to impacts on biodiversity, alien species may also cause damage to economies, ecosystem services or human health [331]. It is noteworthy that some species that were deliberately introduced to new areas may also be economically beneficial as trade or game organisms [335]. However, IAS generally represent a serious threat to native biota and habitats of aquatic ecosystems. This needs to be addressed, if the EU wants to achieve its target ‘to halt the loss of biodiversity by 2010 and beyond’ [331]. The related EU action plan includes specific action A5.1.4 ‘to establish early warning system ... on the emergence of IAS and cooperation on control measures across national boundaries’ [336]. This cross-cutting specific action should be considered carefully during the development and implementation of river basin management plans in line with the provisions of the WFD [337].

The WFD does not explicitly mention IAS (biological pollution) as a potential threat to ecological status, which should be mitigated to enable the achievement of good ecological status. However, Annex II lists specific pressures to which water bodies may be subjected, including ‘... other significant anthropogenic impacts on the status of surface water bodies’. Because the invasion of many IAS to European inland waters have been associated with human activities, it seems reasonable to consider those species as a particular, potential cause of ‘anthropogenic impact’ on the biological quality elements listed in Annex V of the WFD. IAS may impact native biodiversity in many ways, such as displacement of native species, negative impacts on genetic integrity of species, alterations of structure and function of aquatic ecosystems, etc. Consequently, the presence of any IAS detracts the ecological status from the reference state which underlies the WFD. There is an understanding that IAS should be considered in a holistic (cumulative) risk-informed management approach of European river basins [338]. Currently the incorporation of IAS into ecological status assessment in the context of WFD is ongoing [317, 318, 332, 339, 340].

2.6.1 Evaluation of Surveillance Monitoring Data

Generally, IAS monitoring could be incorporated into the routine (surveillance and operational) monitoring system with appropriate sampling modifications and increased sampling effort at invasion hotspots, i.e. the areas at risk. Increased attention should be paid to early detection of invasive species. When new invaders are detected or if their environmental impacts need to be evaluated, the option of investigative monitoring could be useful for (1) examining invader effects, (2) a realistic evaluation of containment of IAS, (3) identifying options for mitigation of their impacts and (4) protection of environmentally valuable sites within the river basin.

As indicated above, the presence of alien species in a system without a priori judgments about their impacts (which frequently remain unevaluated or speculative and based upon information from other studies) results in the deviation from reference conditions and can be considered as biological contamination

(biocontamination). Biocontamination can be assessed by using site-specific and integrated biocontamination indices, i.e. contamination indices based on the proportion of IAS and community disparity. Furthermore, biocontamination estimates allow for the classification of study sites or larger assessment units into five quality classes, ranging from ‘no biocontamination’ to ‘severe biocontamination’ [318]. These simple biocontamination indices can be readily calculated based upon routine monitoring data and are recommended as additional environmental indicators of the status of aquatic ecosystems [341, 342]. This would allow to assess the impact of alien species-specific deviation from naturalness, while the conventional assessment of ecological status is primarily associated with water quality and hydromorphology. Note that improvement of the quality of environment may favour IAS [342, 343].

When assessing the ecological status with respect to IAS based on routine monitoring data, the following important issues should be taken into account: (1) many introduced alien species that were recorded during monitoring may not become established in a recipient system but those that do get established often exert an adverse impact on native species and habitats; (2) owing to low abundances during early establishment (when eradication may still be feasible), alien species are usually not readily detectable by routine sampling methods and (3) IAS may severely impact the native biodiversity, which in turn may affect conventional estimates of ecological status (evaluating status with respect to physicochemical parameters and hydromorphological quality) which are primarily based on native biota. Moreover, estimates of status can be downgraded due to displacement of sensitive native taxa by IAS [342], as well as upgraded when alien species supplement a community [344].

Finally, it should be noted that ‘true’ reference conditions for many river types are actually lost because many rivers have already been biologically contaminated with alien species for a long time [318]. To date, only small tributaries and small lakes across the larger part of Europe are free from (known) IAS. Currently the discussion is ongoing on whether the presence of a single IAS with undetectable impacts is allowable for reference conditions or not.

2.6.2 Confirmation of Invasive Alien Species as Causation of Degradation

Although certain IAS differ in their impacts on native species, communities and ecosystems, it may be suggested that potential impacts from IAS on riverine native biota may generally be proportional to their abundance in an ecosystem. Consequently, biocontamination indices may serve as indicators of potential adverse impacts on biodiversity. When these indicators show high or severe biocontamination levels, it suggests that the community has actually changed as a consequence of the invasion by alien species [318]. Proved displacement or dramatic decrease of native dominant or keystone taxa, as well as alteration of ecosystem functions owing to invasions, can also serve as strong evidence for negative impacts.

To establish cause–effect relationships between certain IAS and their impacts on native communities, bioassay experiments with those IAS in mesocosms or experimental ecosystems (e.g. experimental ponds) would be of great elucidative value. However, such experiments are resource consuming and may not be granted official permission due to the risk of unintentional release of the species to the wild. Consequently, a retrospective analysis of pre- and post-invasion datasets may be the most promising manner in which to reveal causation of effects on native biota. Such analysis requires long-term datasets with representative estimates of abundances of alien and native species, monitoring of areas at risk and introduction pathways for early detection of IAS, species-specific monitoring in some cases and appropriate knowledge of the taxonomic composition of local native biota. Another possibility for the investigation of impacts of IAS may be provided by modelling. For example, RIVPACS-type predictive models [345] may also be appropriate for the evaluation of IAS impacts on native biota.

Evaluation of actual impacts on an ecosystem from IAS has been undertaken by applying the ‘biopollution level’ approach [346]. Assessment of ‘biopollution level’ includes qualitative scoring (based on expert judgment) of known IAS impacts on communities, habitats and functions of specific aquatic ecosystems to discriminate between five status classes. Such an assessment may only be feasible for well-studied systems where it can be based upon information from long-term research and other scientific knowledge. Such information is usually not gathered in routine monitoring programmes.

2.6.3 Considerations for Site-Specific Risk Assessments and Downstream Risks

Uncertainties in measuring actual impacts and assessing risks from IAS are high, especially in large aquatic system [334, 347]. On the one hand, such uncertainties imply difficulties with the practical use of decision support systems, including the application of the ‘biopollution level’ approach by Olenin et al. [346]. On the other hand, this also implies the importance of considering a ‘precautionary approach’ in line with the Convention on Biological Diversity (CBD, see [315]).

When considering the local impacts and downstream risks from IAS, the following issues should be taken into account:

- Once established, IAS may become permanent residents in a system, eliminating them (except in small, isolated water bodies) is generally impossible.
- After establishment at a place of the first introduction, IAS often actively disperse to neighbouring areas (depending on their identity and properties of dispersal pathways, dispersion is more slowly or quickly), thus posing a threat at a large scale not only in the downstream but also in the upstream direction [348].
- Local spreading of IAS (e.g. in a river basin) may proceed naturally, but it may also be greatly facilitated by human-mediated pathways.

- Effects of climate change are likely to provide conditions that encourage the spread of certain IAS beyond their present range and ease their establishment [343, 349, 350].
- Chemical pollution may enhance the invasion of riverine ecosystems by providing empty niches for IAS [343, 351], while the improvement in water quality may enhance dispersal of sensitive IAS; see [343].
- The actual IAS impact on invaded surroundings may not always be readily apparent.

Attempts to predict IAS effects at local and large scales were undertaken by Lockwood et al. [352]. However, the value of such predictions is usually limited because of the scarcity of information on performance of IAS under differing environmental conditions. At present, the invasiveness and impact of single species may be more realistically predicted using or developing empirical models based on well-studied invasion histories [353]. Currently, risk assessment tools for the screening of potential aquatic invaders are under development (e.g. in the UK). It can be argued that, in general, all IAS should be treated as ‘guilty until proven innocent’ [352].

Increasing human activities in European river basins, including further development of the European network of inland waterways may, and probably will, greatly facilitate the transfer of IAS across European inland waters and coastal ecosystems. Therefore, appropriate risk assessment-based management options are required to address risks posed by human-mediated introductions of these species [317].

Environmental Indicators of IAS Connected to the DPSIR Framework

From the risk assessment perspective, an excellent framework for addressing IAS is the Driving forces–Pressures–State–Impact–Response (*DPSIR*) framework ([317, 339] and see Fig. 1 in [2], this volume). IAS risk assessment may be connected to the *DPSIR* framework as follows:

1. Identification of the main invasion gateways, routes and corridors.
2. Selection of the assessment units within a river basin.
3. Identification and analysis of the IAS introduction pathways within these assessment units. This connects to ‘Driving forces’ in the *DPSIR* framework and ‘Low, High and Extreme Risk pathways’ can be used as environmental indicator;
4. Assessment of the inoculation rates within the assessment units. This connects to ‘Pressures’ in the *DPSIR* framework, and ‘Biological Contamination Rate’ and the ‘Pathway-specific Biological Contamination Rate’ can be used as environmental indicators.
5. Assessment of the biological contamination level of the assessment units. This connects to ‘State’ in the *DPSIR* framework, and ‘Biological Contamination Level’, the ‘Site-specific Biological Contamination index’ and the ‘Integrated Biological Contamination index’ can be used as environmental indicators.

6. Assessment of the invasiveness of IAS, established in the ecosystem (i.e. assessment of the biological pollution risk). This connects to 'Impact' in the DPSIR framework and 'Species-specific Biopollution Risk index for specific alien species', 'Grey, White and Black Lists of IAS' and the 'Integrated Biopollution Risk index' can be used as environmental indicators.
7. Development of a Risk Assessment Toolkit for a river basin with early warning services for the reporting of environmental indicators and for suggesting of recommendations for risk management options to stakeholders. This connects to 'Response' in the DPSIR framework.

This approach was recently applied for the risk assessment of IAS introductions via European inland waterways [317]. In that study, 33 assessment units within three main European invasion corridors were selected in order to consider an ecosystem-based approach to the management of IAS using river basins as the main management unit. Testing of selected environmental indicators (see above) for these assessment units resulted in the conclusion that the implementation of management measures at the DPSIR phases 'Driving forces' and 'Pressures' may include preventive actions towards 'Extreme Risk' and 'High Risk' pathways. 'Biological Contamination Rate' and the 'Pathway-specific Biological Contamination Rate' may be used as indicators for the effectiveness of these preventive measures. In contrast, the management actions implemented at the 'State' and 'Impact' phase may involve the control and eradication of established species from the Black List (according to CBD provisions). The 'Site-specific and Integrated Biological Contamination indices' (SBC and IBC) along with the 'Integrated Biopollution Risk index' (IBPR) may be used as comparatively simple indicators of the effectiveness of these measures and can be recommended as cost-effective quality elements (QEs) according to the Common Implementation Strategy of the WFD for the assessment of ecological status of aquatic ecosystems [317].

Further developments of assessment indicators and methods are urgently needed. There is a particular need for methods for the assessment of the physico-chemical water quality and of the river hydromorphological quality with respect to the establishment of alien species. Furthermore, holistic estimates of ecological status should be developed that incorporate biocontamination. The scoring of IAS with respect to their impacts on native biota (for effects evaluation) and their sensitivity to environmental conditions (for physicochemical status evaluation) in analogy with Biological Monitoring Working Party (BMWP) metric appears to be promising.

Development of Early Warning Systems

The opportunities for eradicating IAS are best at an early stage before they become established. Hence, an effective early warning system identifying potential IAS, including newly established ones and/or species expected to spread, is urgently needed and should be closely linked with an IAS-focused monitoring [319].

The Regional Euro-Asian Biological Invasions Centre (REABIC⁷) is an independent regional data centre for IAS serving as an international repository of geo-referenced record data on IAS. The centre is currently focusing on elaboration of effective mechanisms of online open access to the datasets of geo-referenced IAS monitoring data. Specifically, REABIC provides services for data holders, respecting (protecting) their author rights on IAS-related information. This is done via publication of their papers in the international open access thematic journals ‘Aquatic Invasions’, ‘BioInvasions Records’ and ‘Management of Biological Invasions’ that have been established by REABIC.⁸ These thematic journals include a peer-reviewing system as the mechanism of quality control of IAS data, available after their publication in the online information system of REABIC. These scientific journals, as part of the information system of REABIC, serve to provide a unique opportunity to develop early warning systems, based on the most recent geo-referenced records of IAS and species-specific risk assessments.

The development of REABIC-based journals as early warning instruments was supported in 2006–2009 by the EC FP6 Project ALARM⁹ and in 2009–2013 by the EC FP7 Project EnviroGRIDS.¹⁰

2.7 Hydromorphological Degradation

In aquatic environments, the term ‘habitat’ comprises characteristics of the water medium (e.g. the temperature or the current) as well as characteristics of bottom substrates (e.g. grain sizes, organic content) and plant coverage (e.g. filamentous algae). Habitats can be defined individually for each species or group of species. In case of semiaquatic species, terrestrial riparian structures must be considered as well. This section does not consider any habitat parameter related to water quality, which is addressed in Sect. 2.1 but focuses on hydrological characteristics and on morphological features, which can be summarised by the term ‘hydromorphology’.

Hydromorphological degradation, in contrast to organic pollution or acidification, describes an assortment of different impact types, e.g. the removal of riparian vegetation, straightening of rivers, bank or bed fixation, flow regulation, water abstraction, dams and the separation of river and their floodplains. Such changes and their impacts act at various spatial scales: While the impact of small dams or local bed fixations is restricted to a few metres’ stream length, large dams may prevent the supply of bedload for entire downstream catchments and may obstruct fish migration to the upstream catchment. As a consequence, most river sites are not only affected by local hydromorphological conditions but also by large-scale

⁷ See: <http://www.reabic.net/>

⁸ See: <http://www.reabic.net/journals/>

⁹ See: <http://www.alarmproject.net>

¹⁰ See: <http://www.envirogrids.net>

conditions and their impacts in the upstream and downstream catchment. Hydromorphological degradation, caused by straightening, dam construction and disconnection of rivers from their floodplains, represents one of the biggest stressors on river systems in Europe and elsewhere [354]. Almost 80 % of total water discharges of the main European rivers are affected by flow regulation measures [355], while 90 % of original floodplain areas have been ‘claimed’ by mankind [356, 357]. Although this is a removal of land from its original ‘use’, it has been proposed that the correct term should be ‘land claim’, not ‘reclaim’.

In Germany, a nationwide study to assess stream hydromorphology based on a method described by Länder-Arbeitsgemeinschaft Wasser [358] revealed that 3 % rated ‘unimpaired’, 19 % ‘slightly’ or ‘moderately changed’, while 78 % were classified in the four poorer classes ‘distinctly/obviously/strongly/completely’ changed [359]. There are, however, distinct differences between ecoregions: In mountainous regions of the Federal State Hessen, 19.7 % of the streams classified in the best three (out of seven) morphological quality categories, while 64.1 % of the streams belonged to the lowest three classes [360]. In lowland catchments of the Federal State North Rhine-Westphalia, only 2 % of the stream sections were reported to be in a good hydromorphological condition (highest two out of seven classes) compared to 54 % classified as having a poor hydromorphological condition (lowest two out of seven classes, [361]). In the Netherlands, only 4 % of the streams are considered to have a near-natural morphology and hydrology [362], whereas in Denmark only 2 % are considered as natural [363]. Hence, based on these hydromorphological surveys, it is obvious that the improvement of river hydromorphology is among the most urgent challenges to achieve good ecological status, particularly in lowland regions.

2.7.1 Evaluation of Surveillance Monitoring Data

Abiotic Indicators

In monitoring programmes under the WFD, the Annex V hydromorphological parameters may include hydromorphological variables supporting the biological elements. The next parameters may be measured in particular: quantity and dynamics of water flow, connection to groundwater bodies, river continuity, river depth and width variation, structure and substrate of the river bed and structure of the riparian zone. In practice, many European countries had already initiated hydromorphological surveys before the WFD entered into force. The data of these surveys are now a valuable source of information for planning of the programme of measures.

In Germany, habitat quality is assessed by hydromorphological surveys (e.g. Strukturgütekartierung) for 100 m stretches, both as an overall indicator and as three separate indicators for river bed, riparian vegetation and urban surroundings [358]. For Europe, the ‘Fluvial Habitat Index’ has been suggested that combines river bed with hydrological characteristics and shading [364]. Recently, the

‘Habitat Degradation Index’ has been tested for Swedish conditions, considering in-stream microhabitat characteristics [365].

Biological Indices

Biological Quality Elements (BQE) monitored for WFD purposes should relate to hydromorphological quality, in particular the fish fauna, benthic invertebrates and to a lesser degree macrophytes, while phytoplankton and benthic diatoms are less responsive to hydromorphological stress [366].

Habitat-related variables play an important role in most fish-based assessment systems. A selection of fish-based metrics clearly related to habitat degradation include sentinel metrics, reproduction metrics, feeding types, habitat preferences, tolerance metrics, migration metrics and longevity metrics (e.g. [367]). There is a long history of building ‘Indices of Biotic Integrity’ (IBIs) based on fish fauna, mainly in the North American literature [368, 369], which include metrics that are indicators for different types of stress, including various forms of hydromorphological degradation. Similar approaches have been developed for Europe [367, 370] and are frequently applied.

A number of assessment systems are designed to use river macroinvertebrates to indicate the effects of hydromorphological degradation on the catchment, reach and habitat scales [365, 371–375]. As an example, the German invertebrate-based assessment system PERLODES comprises a module called ‘general degradation’, which includes a number of metrics responsive to hydromorphological features and to intense catchment land use. Metrics included in this assessment system with a well-supported link to hydromorphological degradation are, among others, the following: (1) the Fauna Index [361], a metric mainly composed of specialists for high-quality or degraded habitats; (2) the percentage of Ephemeroptera, Plecoptera and Trichoptera (EPT taxa), responsive to various types of flow modification and habitat degradation; and (3) the percentage of mud-dwelling (Pelal) taxa, indicating a strong input of fine sediment or stagnation.¹¹ Recently, the SPEAR[%]_{habitat} metric has been introduced which clearly indicated in-stream microhabitat characteristics, while it did not correlate to other physicochemical or toxic variables [365].

2.7.2 Confirmation of Habitat Degradation as Causation of Degradation

The impact of hydromorphological degradation on river biota is often difficult to detect, because (1) hydromorphological degradation is composed of several different individual stressor types affecting riparian and in-stream habitats, substrates and

¹¹ See: www.fliessgewaesserbewertung.de

current; (2) hydromorphological degradation acts at different spatial scales (catchment, reach, site) and (3) hydromorphological degradation is often correlated to various other stressors as most hydromorphologically altered rivers sections are located in intensively used areas also exposed to pollution sources. The detection of hydromorphological impacts on biota therefore requires either experimental approaches or multivariate analyses considering a wide range of environmental pressures.

In a multivariate study of 183 stream sites in nine European countries, Hering et al. [258] compared the responses of diatoms, macrophytes, invertebrates and fish to different stress gradients. While all correlated strongly to eutrophication and organic pollution, correlations to other gradients were less strong. All organism groups correlated to varying degrees to land-use changes, hydromorphological degradation on the microhabitat scale and general degradation gradients. However, the correlation to hydromorphological gradients on the reach scale were mainly limited to benthic macroinvertebrates (50 % and 44 % of the metrics tested correlated significantly in mountain and lowland streams, respectively) and fish (29 % and 47 %).

Hydrological pressures (such as residual flow or stagnation) and morphological pressures, (such as dams, obstructing fish migration) are among the parameters which best correlate to the composition of fish fauna in European rivers (e.g. [367]).

Effects of hydromorphological degradation on riverine benthic macroinvertebrates were shown or hypothesised for various river types in Europe [361, 372, 376], in the USA [371, 377, 378] and in New Zealand [379]. Macroinvertebrates demonstrate good correlation with the following stressors: stagnation, the removal of bed structures such as wood debris, the removal of riparian vegetation (an important habitat for the terrestrial stages of adult insects) and residual flow, as well as to other stressors.

Macrophytes, mainly correlating to eutrophication, also show some correlation to the hydromorphological state [380, 381]. For example, liverworts, mosses and lichens decrease in abundance with hydromorphological degradation in mountain streams, while species such as *Elodea canadensis*, *Sparganium emersum* and *Potamogeton crispus* were indicative of degraded lowland sites [381].

However, the linkages between the hydromorphology and biota, as presumed by correlation coefficients between hydromorphological parameters and biological response variables, are usually low. This can at least partly be explained by the complex interactions between both groups of parameters. An example of respective cause–effect chains is given in Fig. 5. Most biological variables, e.g. assessment metrics, are simultaneously affected by a variety of hydromorphological variables.

2.7.3 Risk Assessment of Local Impacts and Downstream Risks

The WFD states that hydromorphological assessments of streams and rivers should form part of the operational monitoring programmes of EU member states (to be updated at 6-year intervals; see Fig. 3 in [2], this volume) and that

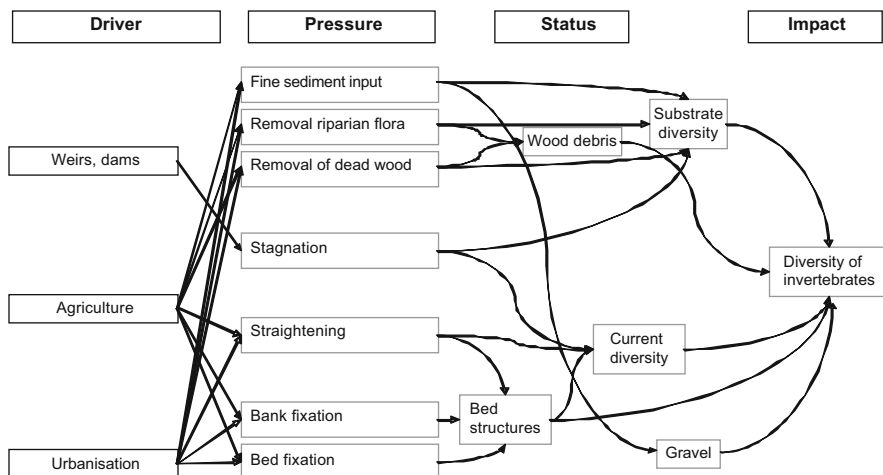


Fig. 5 Example of a Driver–Pressure–State–Impact chain involving hydromorphological variables and their impact on invertebrate communities (i.e., an invertebrate-based assessment metric)

hydromorphological assessment methodologies should adhere to the relevant CEN/ISO standards or equivalent national or international protocols. Therefore, hydromorphological assessments form an integral part of the WFD monitoring and quality assessment process [382]. There are a variety of well-established methods for hydromorphological surveys, e.g. the ‘River Habitat Survey’ [354] or the related method in Germany [358]. Because hydromorphological surveys are not mandatory under the WFD, more frequent use is made of the indicative potential of fish and macroinvertebrates to indirectly assess hydromorphological degradation.

The main purpose of monitoring under the WFD is to detect environmental change for better or worse, including the effects of restoration measures. Among measures presently implemented in Central Europe projects, those aimed to improve river hydromorphology prevail. The restoration of hydromorphological features, however, does not necessarily lead to an improvement of the ecological state. Even well-designed and well-implemented restoration measures often do not lead to the establishment of fastidious benthic invertebrate species, e.g. species requiring rare habitats (see review in [383]). The EC FP7 project REFORM¹² aims to provide a framework for improving the success of hydromorphological restoration measures to reach, in a cost-effective manner, the target ecological status or potential of rivers.

Most authors hold the lack of source populations of sensitive species and the negative impacts of the upstream catchment responsible for the slow recovery. Other possible reasons are time lags and hysteresis effects in the recovery processes and the fact that several sensitive species are extinct in entire catchments.

¹² See: <http://www.reformrivers.eu/>

Conversely, hydromorphologically degraded sections in near-natural catchments may be characterised by near-natural invertebrate communities, as parameters acting on the catchment and reach scale strongly influence the local community structure [384–387]. In general, fish and macrophytes respond more quickly to hydromorphological restoration measures and may thus be used as early indicators of recovery, while the recovery of invertebrate fauna may require more time [388].

2.8 Changing Water Levels: Floods, Droughts and Water Level Regulations

The term flood implies that water covers areas which are usually not covered by a river or other water body. Floods are therefore generally characterised by horizontal dimensions, although they are usually measured by vertical means, such as HW (highest measured level of water) or MHW (average of highest measured annual water levels). Some dictionaries state that flood is simply a fast rise of water level in a surface water body followed by a slower decrease [389]. The general public views floods negatively, and there is a desire, especially in industrialised Europe, to eliminate them due to their impacts on agriculture, waterborne traffic and housing. Therefore, many European rivers are regulated to avoid large fluctuations of the water level. On the other hand, civilisation is rooted in cultures which relied on the floods of large rivers in arid climates and the resulting transformation of fertile sediments into agricultural soils to support their farming. Despite the fact that a large proportion of the world population still relies on agriculture which depends upon floods, there is a large trend towards avoiding floods and further narrowing areas where water level fluctuation is allowed to take place.

Droughts represent another extreme event occurring rarely to regularly. Droughts can be described as deficiencies of water in general, low flow in rivers or decreasing water levels, but from the hydrological point of view, this can be simplified to ‘drought is a sustained and regional extensive occurrence of below-average natural water availability’ [390]. Despite their regularity of occurrence, they also have a negative connotation and are among the major reasons for large-scale modifications of water courses, especially in the southern part of Europe.

Besides droughts and floods, normal fluctuations between the low and the high flow of waters are typical properties of water bodies, depending on the area, lake percentage and precipitation of drainage basin [391]. Dynamics of water level fluctuation and further differences between hydrologically dry and wet years are among the major stressors affecting biota of lakes and rivers [392, 393]. The water level fluctuation zone is a part of the littoral zone of lakes and rivers with prevailing aquatic vegetation.

Floods and droughts in streams and rivers may be viewed as disturbances in which river flow and water availability rise or fall to extreme values for a short or extended period of time. Knowledge of the ecological implications of floods and

droughts on stream-dwelling organisms is scattered and rather fragmentary (e.g. [394, 395]). While very important for river management [396], much of the available information is obtained by chance, due to the unpredictable nature of these events. As a result, before–after investigations are usually lacking and only a few studies exist where this investigational approach has been used (e.g. [397]).

Flood disturbances can be best described as ‘pulse disturbances’ [398]; during floods, invertebrates are affected by water velocity (e.g. [399, 400]) and bed movement [401, 402]. General responses of invertebrate communities to floods include a reduction in density and taxonomic richness [403] and a shift in community composition to one dominated by opportunistic taxa tolerant of unstable conditions [404, 405]. Despite the large effect that floods can have on the invertebrate communities, their effects are only temporary [404, 406–408].

Drought disturbance differs from flood disturbance in such that the disturbance effect on stream communities increases slowly over time; they are ‘ramp disturbances’ (in line with [398]). Both flood and drought disturbance events are usually large-scale events and can be either periodic, seasonal or infrequent, but usually with high magnitude [394].

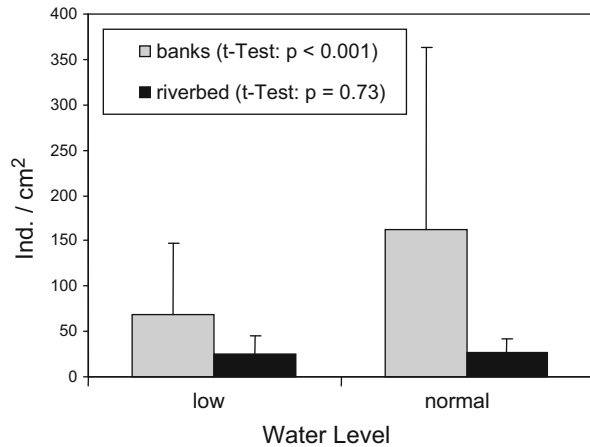
The vertical depth gradient (depth factor) constitutes the basis for the zonation of aquatic macrophytes in littoral areas [409]. Direct effects include factors such as water level fluctuation, effects of ice, penetration of light and bottom substrate type. Temperature and hydrostatic pressure changes as a function of depth also may influence the distribution of macrophytes [409]. Indirect effects include changes in seed germination dispersal processes, which are the result of the shifted zonation of macrophytes [410]. Aquatic macrophytes avoid areas where they are subject to the risk of drying, as their life forms are morphologically adapted to survive in the environment of the littoral zone.

Helophytes occupy the uppermost part of the littoral, whereas areas of permanent submersion are inhabited by submerged macrophytes and other aquatic macrophytes with floating leaves (i.e. isoetids, nymphaeids, elodeids, ceratophyllids, charids, bryids), which can be also called hydrophytes. Helophytes are rooted plants that are continuously in contact with the open air. The growth areas of the *Carex* species (sedges) are directly related to water level fluctuations. A high spring flood followed by diminishing water levels towards summer affects proliferation favourably. A high flood prevents competition by shrubs, grasses and mosses, because sedges can resist relatively long periods of submersion in a state of dormancy. Hydrophytes are much more susceptible to the risk of drying and therefore they cannot resist long drought periods.

Benthic invertebrates in the littoral zone are also susceptible to water level fluctuations in the littoral zone. Depending on their life strategy, they may be affected directly by desiccation or indirectly by decreases in habitat availability and food resources. In northern regulated lakes, littoral communities have changed due to increased freezing of sediment during winter [411].

Depending on the species, benthic organisms can resist the effects of droughts by using refugial areas or develop resting or dormant stages (e.g. winter eggs or cysts) which strongly influence the capacity of the species to recover. Therefore, different

Fig. 6 Comparison of macrofauna abundance during extended periods (at least 30 days) of low and normal water level at the littoral banks and in the river bed of the Rhein River



results have been found concerning the abundance of benthic invertebrates during droughts: both decreasing [394, 412–414] and increasing invertebrate abundances [415] have been reported while species composition often remains unchanged. Generally, pollution-tolerant taxa replace the more sensitive and flow-requiring taxa during droughts [397]. On the community and ecosystem level, drought events affect ecosystem functioning (e.g. litter breakdown) more than the structure of the invertebrate communities [416]. In the River Elbe, extended periods of drought usually result in reduced overall abundance of macrofauna in the littoral part of the banks, while the river bed itself remains mainly unaffected (see Fig. 6).

Recovery of biota varies between seasonal and infrequent drought events [394]. Faunal recovery from seasonal drought seems to often be relatively rapid, with a distinct and predictable sequence of biota (e.g. [417]), while the recovery process from a supra-seasonal drought can vary considerably, making the process less predictable. Recovery rates ranging from month (e.g. [418, 419]) to a whole year have been reported (e.g. [420]). These inconsistent responses to drought events are frequently attributed to the withdrawn amount of water [395]. Two main processes are most influential on the recovery process: species succession and the temporary loss of species [394].

The effects of droughts on the available volume of water in rivers can range from a small reduction of water and a respective exposure of banks to the complete loss of surface water and connectivity. There are both direct and indirect effects of drought on stream ecosystems. Direct effects include the loss of water and the loss of habitable space [421, 422] for aquatic organisms, while indirect effects include the impairment of the water quality, the alteration of food resources and changes in the strength and structure of interspecific interactions. Usually, droughts have pronounced effects on the abundance of species, the size or age structure of populations, the community composition and diversity and on ecosystem processes [418, 419].

Flood events again can also induce disturbance on benthic communities. Usually shear velocity and substrate mobility increased as a result of flooding resulting in decreasing food availability for herbivore species [423], causing changes in the high-level characteristics of ecosystem structure (e.g. periphyton versus macrophyte dominance) and function (e.g. balance between auto- and heterotrophy) [396]. Benthic macroinvertebrates are observed to migrate from deep water to shallow water [424], resulting in increased densities in shore areas. However, when flow speed rises above the level at which small cobbles are moved (approximately 30 m/s), abundance of benthic species usually drops [406] due to the refuge-seeking behaviour of many species.

Most of the rivers in Central Europe are fed by snowmelt waters from mountainous areas, and therefore, highest flows are during the summer time [425]. Lake water levels react more slowly to increased flow and therefore flooding phenomena do not create similar zonations to those seen in the northern part of Europe [426]. Reed-dominated wetlands are typical of shores of large Central European lakes, such as Lake Balaton, Lake Constance and Neusiedler See. Southern European wetlands and littoral zones are more often created as the consequences of floods originating in the rainy winter periods. Most of the natural European wetlands and flooding zones disappeared due to human activities centuries ago. Most floodplains have been transformed into farmlands throughout Europe, and delta areas of many large rivers are now more often industrial zones than natural wetlands. Only a few representative wetlands, such as the Danube Delta, are still in a relatively pristine state. Moreover, there are still some representative extensive wetlands remaining on the shores of some Balkan lakes such as Lake Shkodra and Lake Prespa.

Water level regulation is related to the human need to control the water levels of lakes and the flows of rivers in such a way that it is beneficial for various users of the watercourses [355]. The typical situation in regulated water bodies in the northern hemisphere is that water levels during summer period are normally high or rising, while during the winter period, when the need for electricity is normally highest, the water level is strongly lowered [427]. Flood prevention regulation follows a similar pattern during wintertime, 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 kept even more stable than natural ones. If the water level is regulated for water supply use, the water level fluctuation is more irregular and depends upon the specific use of raw water. Typical water level fluctuation curves in regulated and non-regulated Finnish lakes are described in Fig. 7.

2.8.1 Evaluation of Surveillance Monitoring Data

Abiotic Indicators

In order to quantify the effects of floods in lakes, their height and timing are the most important indicators describing their biological significance. Height of a flood can be expressed as the vertical difference between water levels in summer time and

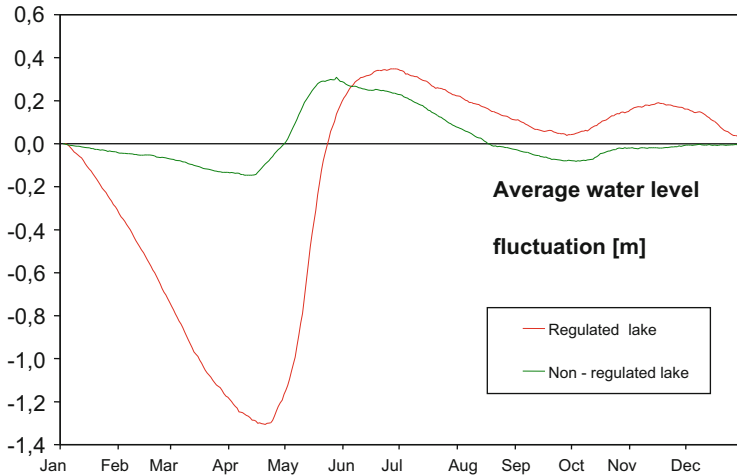


Fig. 7 Average water level fluctuations in 18 regulated (*red line*) and 18 non-regulated large lakes (*green line*) of Finland (1980–1999)

during flood peaks [391]. Higher floods provide larger habitats for adapted biota, although the socioeconomic effects on humans are usually perceived as negative.

Additionally, the timing of floods is an important indicator, especially in the northern hemisphere where flood peaks are related to availability of spawning habitats of spring-spawning fishes such as northern pike (*Esox lucius*). Delayed spring flood decreases the extent of sufficiently shallow water habitats, which are also important for fries of autumn-spawning fishes, such as whitefish (*Coregonus sp.*).

A third important indicator is the water level fluctuation range during open water periods or, more exactly, during growth periods. Large variations result in extensive littoral habitats, providing higher productivity and, hence, a greater biodiversity value.

A fourth important indicator, especially for small benthic invertebrates, is the speed of the water level change: During floods, benthic macroinvertebrates shift from deep water to more shallow waters [424], while drought induces the reverse behaviour. Benthic invertebrates can follow slow changes in water level more easily than fast ones.

In ice-covered-regulated lakes and reservoirs, the winter drawdown, which can be expressed as the difference between highest water level during autumn and lowest water level during spring, is the most significant stressor for biota [428]. Down-dwelling ice causes the freezing of sediments and also exerts mechanical pressure.

In river ecosystems, floods are more often described as mean flood discharge (MHQ) with indicator values similar to MHW in lakes [392]. Deviation of the average value will cause also changes in the vertical and horizontal extent of flood-related ecosystems. Timings of high and low flow in rivers are also important factors, causing pulsing phenomena essential for ecosystem health and services [429].

In heavily modified rivers used for hydropower production, short-term, usually daily, dynamics also play important roles as local stressors [355]. Daily differences between highest and lowest flow are the best indicators, although some more

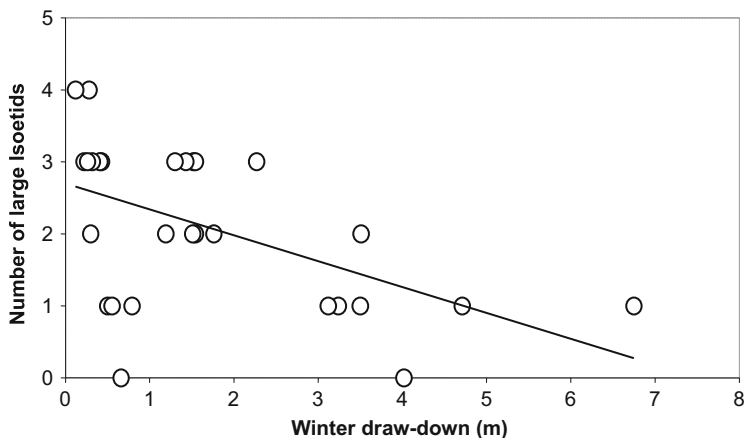


Fig. 8 Relationship between total number of large isoetids and winter drawdown of lake water level in some selected Finnish lakes

sophisticated indicators, such as the number of high-flow peaks, are also proposed [392].

The effects of droughts are best quantified in water bodies by lowest water levels or by lowest flows, although the durations of such phenomena also play an important role. Hydrological duration curves of submersion ranging from 100 to 0 % therefore provide more valuable information for water managers than does one simple number. From the ecological point of view, the exposed area is essential; lakes or rivers with mildly sloping shores are more vulnerable than are similar water bodies with steep littoral zones.

In rivers, droughts often result in the reduction of dissolved oxygen [430–432], decreasing turbidity [397] and increasing nutrient loading [394].

Biological Indices

With regard to floods, the vertical extent of the helophyte zone is clearly connected to annual water level fluctuations and provides one of the best biological indices [433]. Regular monitoring data rarely provide information on helophytes, which are often not included in the list of analysed macrophyte species. In the northern hemisphere, the presence of flood-tolerant sedges (*Carex* spp.) provides a positive indication of the status of lakes or rivers.

Aquatic macrophytes are among the best indicators of hydromorphological changes (see Sect. 2.7) in lakes related to water fluctuations [433, 434]. Large-sized isoetids (*Isoetes lacustris*, *I. echinospora*, *Lobelia dortmanna*, *Littorella uniflora*) are key species in the littoral zone of soft water lakes and are sensitive to the effects of descending ice. Winter drawdown of regulated lake results in pronounced degradation of large isoetids (Fig. 8).

In addition to aquatic macrophytes, benthic invertebrates also react to fluctuating water levels. Depending upon the availability of habitats, large-sized taxa such as Ephemeroptera, Trichoptera, Coleoptera and Megaloptera are usually replaced by smaller species [411]. Furthermore, eggs of autumn-spawning fishes are subject to the effects of ice, particularly in regulated lakes with descending ice [435]. Collapsed stocks of whitefish are typical indicators of increased water level fluctuations.

The effects of floods and droughts on benthic invertebrates and the successive recovery process can be measured with a number of different variables, such as species richness, population densities, species composition and species traits [395], whereby the last seems to be most sensitive. For example, Lake [394] found marked effects on the densities and size or age structure of populations, on community composition and diversity and on ecosystem processes. For droughts, taxa sensitive to organic pollution are replaced by more tolerant species (species which are tolerant to low levels of dissolved oxygen [397, 415]). Moreover, changes in the riverine macroinvertebrate community associated with low flow included a reduction in species richness and the number of individuals per sample, particularly aquatic insects [413].

Effects of drought can be seen as a disappearance of desiccation-sensitive hydrophytes and the dominance of rapidly spreading, flexible, small-sized plants [433]. However, they are not very good indicators of droughts due to fact that they also grow under normal conditions in the lake littoral zone. Similarly, macroinvertebrate community structures in Lake Constance in Central Europe do not change much during drought; temporary drying of the littoral zone affects only abundance of typical species without changes in species composition [436].

2.8.2 Confirmation of Changing Water Levels as Causation for Degradation

In the northern hemisphere, the timing of spring flood plays an important role in the survival of biota [433, 437]. Lakes and rivers are covered by ice for several months, which also means that littoral vegetation, benthic invertebrates and eggs of autumn-spawning fishes are in a resting state. Ice break-up usually results in increasing amounts of melt waters and the start of the spring flood. Hence, spring floods have a great influence on the structure of the upper littoral zones, usually consisting of sedges (*Carex spp.*). The sedge-dominated zone is also called eulittoral, which forms ideal surroundings for young fish fry and spawning grounds for spring-spawning fishes, such as northern pike. Full development of the littoral vegetation, including different helophytes (*Phragmites*, *Scirpus*), nymphaeids (*Nymphaea*, *Nuphar*) and elodeids, requires a decreasing trend of water levels during the summer month. Changes in the extent or timing of floods thus result in changes in this vulnerable zone. A change in the eulittoral zone could therefore be used as indicator for the effects of changing water levels.

In the southern part of Europe, droughts play a more important role as ecological factors, and their effects can be seen in wider spatial scales. However, such phenomena can also be observed in the northern hemisphere. For example, Lake

Neusiedler See is extremely shallow, with an average depth of 1.1 m and water levels are mainly supported by groundwater and precipitation [437]. Seasonal water level fluctuations are quite regular with annual amplitudes of about 0.4 m, but due to high evaporation from the lake surface and extensive reed beds, there are variations between different years. Annually, evaporation is low during cold periods and increases in the summer. Hence, water levels decrease during summer and autumn and reach their minima in late autumn (October/November). Changes in the water level are therefore also visible in ecological condition of the lake [437]. When the water levels have a normal height, the reed stands and the open water areas are connected. However, if the water levels are too low, the water surfaces are discontinuous, with some water areas becoming disconnected. This can also have a great influence on the chemical composition of the lake water. Conductivities and ion concentrations increase in low-water years [438], which is a good indication for the effects of droughts.

2.8.3 Considerations for Site-Specific Risk Assessments and Downstream Risks

Effect of European Legislation: Heavily Modified Water Bodies

Hydropower, flood protection and waterborne traffic greatly affect the ecological status of water courses. According to the WFD [3], these hydrologically and morphologically altered water bodies can be designated as heavily modified, meaning they may have lower environmental objectives assigned and designated good ecological potential (GEP, see [439]). The assessment of whether the water body meets GEP is a multiphase and complex process. It requires that hydromorphological mitigation measures to improve ecological status are identified, and their impacts on uses are estimated. In the process, those measures which cause significant harm to the uses of the water body are excluded. The WFD requires that the opportunities to improve the ecological potential in regulated watercourses are assessed and, where needed, implemented. This may cause losses to hydropower production and other human-related interests such as flood protection. Black et al. [440] described how heavily modified rivers can be designated using the flow regime estimation tool 'Dhram' in Scottish hydropower rivers. For lakes, a similar tool called 'Regcel' is applied in case studies dealing with the designation procedure of Finnish heavily modified lakes [391].¹³

¹³ Detailed instructions on how to use these tools are downloadable at <http://toolbox.watersketch.net/>

Mitigation of Regulated Lakes by Ecological Regulation Practice (ERP) in Finland

Effects of water level regulation can be diminished by changing the critical water level and using ecological regulation practices introduced by Hellsten et al. [441]. These methodologies have been applied in many regulated lakes of Finland since the beginning of the 1990s. In principle, a slight increase in water levels in February can improve the ecological condition of frost-sensitive large isoetids and benthic invertebrates by lowering the zone where ice penetrates sediments and freezes its surface. Similar effects can be obtained by limiting water level drawdown during late spring. This particularly decreases the area of down-dwelling ice with positive effects on autumn-spawning fishes (fish egg survival of whitefish), aquatic macrophytes (ice-sensitive species such as *Isoetes lacustris*) and some benthic invertebrates (e.g. *Ephemeroptera* species). The third main aim of ERP is to restore water level fluctuation dynamics and range during summer to enhance zonation of aquatic macrophytes and to create a more natural water level fluctuation zone. Finland has applied ERP in most of its regulated lakes, usually related to the re-licensing process or on a voluntary basis. In most cases, changes in water level fluctuations are limited to some centimetres, but positive ecological signs are already visible.

Assessment of Downstream Risk: The Environmental Flow Approach

Downstream risk assessment related to drought events can be realised by using simple hydrological parameters [392], although an ecological impact assessment depends very much on the area of the floodplain or riparian zone. However, especially in hydrologically altered rivers, definition of environmental flow as described by King et al. [442] is more important from the point of view of the biota. Various definitions of environmental flow can be found in literature. Meijer [443], for instance, defined environmental flow as the proportion of the natural flow regime that is maintained in a river, wetland or coastal zone to sustain ecosystems and the benefits they provide for people (i.e. ecosystem services; see [444], this volume). Various methods have been developed to quantify the environmental flow requirement. Most of these environmental flow assessment methods focus on the relationship between river flows and ecosystem condition, but do not explicitly quantify the importance of flows for the well-being of various groups of people [443]. Determination of suitable and sustainable environmental flow needs detailed information on ecological preferences of biota and availability of suitable habitats.

2.9 *Sediment and Suspended Matter*

Soil can be defined as ‘. . .the top layer of the earth’s crust. It is formed by mineral particles, organic matter, water, air and living organisms. Soil is an extremely

complex, variable and living medium'.¹⁴ Sediment has been defined by the European Sediment Network (SedNet¹⁵) as 'Suspended or deposited solids, acting as a main component of a matrix which has been or is susceptible to being transported by water' [445]. The first thing one notices when comparing these two is that the above definition of soil does not specifically exclude sediments. A further examination of the SedNet sediment definition reveals that many soils, during weathering and transport processes, have been sediments and also suggests that they will be sediments again during their lifetimes. Clearly, both soils and sediments often have a common origin—the weathering products of rocks and organic material. Generally, if they are deposited (or formed) in a terrestrial setting, they are considered soils, while if deposited in an aquatic setting, they are considered sediments. Because waters are continuously dynamic, their associated particles are moving as well, resulting in a cycle of erosion, suspension and deposition throughout the natural (and anthropogenically altered) environment.

Both soils and sediments are the product of past geogenic and cultural activities (see [4], this volume), forming an essential part of the landscape. Both are accumulators (and long-term donors) of persistent pollutants such as heavy metals and persistent lipophilic organic compounds, as well as excess nutrients and organic matter. The physical, chemical, mineral and nutrient compositions of both soils and sediments, whether natural or anthropogenically altered, affect the composition of plant and animal assemblages associated with them and can thus impact the use of land and dredged sediments and the quality of crops. Shifts in grain size distribution, excess or lack of nutrients and/or organic matter and the build-up of contaminants can all affect substrate quality and ecosystem services. The deposition of uncontaminated sediments over contaminated deposits can isolate contaminants and reduce risks to ecological endpoints, while newly erosive conditions may expose buried contaminants (see also Sect. 2.1).

Unlike some of the other stressors considered in this chapter, uncontaminated soils and sediments are natural and essential parts of aquatic ecosystems, providing essential ecosystem functions and services, habitat and substrate for a variety of organisms, as well as playing a vital role in the hydrological cycle. Changes in hydrodynamics can affect the stability of soils and sediments, deplete soils and alter erosional and depositional patterns for sediments, thereby changing habitats and reducing habitat diversity. In many cases, streams or channels are in a state of 'dynamic equilibrium'; over time, the amount of sediment carried is approximately in balance with the transport capacity of the stream, and sediment erosional and depositional patterns as well as the resultant substrate and habitat types are somewhat stable. It is important to note, however, that even under 'stable' conditions, all these parameters vary on a range of spatial and temporal scales. Normal flows help avoid channel bed aggradation or degradation, higher seasonal flows help support habitat complexity and even higher occasional flows cause more extensive changes

¹⁴ See: <http://ec.europa.eu/environment/soil>

¹⁵ See: www.sednet.org

or even reposition or rebuild habitats or change channels. This habitat complexity is essential to healthy ecosystems, as heterogeneity is a fundamental and necessary component of ecosystem functioning [446].

However, when conditions change, due to natural or anthropogenic factors, alteration to the channel morphology, flow and sedimentation patterns occur, and these can result in both positive and negative impacts to various habitats, functions or ecosystem services in a river basin. For instance, an increase in sediment supply may result in deposition in spawning gravels, navigational channels, dams and reservoirs (reducing water storage capacity) but, on the other hand, may also enhance wetlands, mudflats, deltas and coastal defences. Reduction in sediment inputs, to protect some endpoints (e.g. water storage capacity), may put downstream endpoints at risk. For example, decreased sediment run-off as the result of reservoirs has resulted in wave-induced erosion of coastlines [447], and the lack of sediment below a dam or barrier can result in 'sediment-hungry water' and downstream erosion. While channels are dredged to support navigation, this changes channel morphology, possibly resulting in changes in erosion or deposition within, upstream or downstream of the dredged channel. While sediments naturally move in rivers, excess sediment suspended in the water column (turbidity) can block light, inhibit primary productivity, reduce oxygen in water, clog gills, interfere with feeding, reduce water quality and inhibit water abstraction.

As a natural and essential component of ecosystems, uncontaminated sediment is not per se a stressor on the ecosystem. However, the rate and magnitude of changes in sediment inputs, hydromorphological changes, discharge rates and habitat changes caused by anthropogenic change may, in many cases, overwhelm the resilience of a number of ecological endpoints on a reasonable timescale. Furthermore, the extensive use of rivers and floodplains for a vast range of human uses means that even natural sediment dynamics may come into conflict with many human uses of river basins such as navigation, flood control and abstraction. Sectoral management in support of one river basin objective, such as fishing [448], dredging and forestry [449], can negatively impact other services, either in that system or in other connected systems. Thus, besides the management of sediment to maintain good ecological status in river basins, there is often a need to also protect river basin socioeconomic objectives.

2.9.1 Evaluation of Surveillance Monitoring Data

As described above, sediment excess or lack can pose risks to ecological and socioeconomic endpoints. The magnitude of release of sediment-associated stressors and the likelihood of impacts on various biotic and abiotic endpoints are complex functions of site-specific and regional intrinsic factors (including soil type, topography, climate and habitat distribution) and management factors (including land use, farming practices and river management). Some experimental results using invertebrates from lowland [450] and upland streams [451] show that uncontaminated suspended sediments have limited direct effects, but other studies have

suggested significant impacts [452]. Deposited sediments can have large direct effect on stream invertebrates, particularly if their grain size or other characteristics do not match those required as habitat for specific endpoints [453]. This suggests the importance of monitoring the in-stream deposition of sediment and respective physicochemical properties of deposited sediments, rather than measuring only suspended sediments in the water column. This section addresses the assessment of sediment as a stressor, although this issue is more complex than for other stressors. Risks are diverse and site specific, and the same sediment that may pose a risk to one endpoint in one location may provide benefits to endpoints in other locations of the same river or stream.

Abiotic Indicators

Sediments dynamics in river basins are the result of interactions between a number of variables including amount of flowing water (the ‘discharge’), sediment supply and grain size distribution, as well as channel characteristics including width, depth, velocity, slope and roughness.

Grain size is an especially important factor in determining sediment fate, behaviour and impact. Although rivers, at different levels of flow, have the ability to move a vast range of sediment grain sizes, from silt to boulders, grain size will affect transport and settling velocities, as well as habitat type and quality upon settling. Fine-grained sediments (also called silt, mud, fines or wash load) are the most likely to remain in suspension at all but the lowest flows and thus provide the greatest risk of turbidity impacts, and if siltation is a concern, siltation impacts. Coarse-grained sediments (also called gravels or bedload) settle quickly and are less erosive, while medium-grained sediments (also called sand or suspended bedload) possess characteristics between the two.

Sediment composition can also affect modes of impact—sediment-associated chemical contaminants can cause impacts that are addressed in Sect. 2.1, while organic-/nutrient-rich (Sect. 2.2) or pathogen-bearing particles (this section) coming from urban, sewage, industrial or agricultural sources (sometimes called suspended solids, SS; or particulate organic matter, POM; or suspended particulate matter, SPM) may cause different impacts, although they may behave in many other ways like mineral-rich natural sediment particles.

Depending upon how risks are assessed, ‘sediment-hungry water’ (less sediments in water than should be in equilibrium with present water energy and sediment grain size distribution) can be seen either as a sediment-related stressor or an impact of hydromorphological change (see Sect. 2.7). As the potential risks from sediments are complex, it is important to consider, at a site-specific and regional scale, the types of sediment-related stressors that are being released and transported to endpoints of concern.

Streambed-sediment indicators are applied to address potential impacts in which excessive or insufficient levels of fine sediments or unnatural substrate size composition directly and indirectly impair aquatic habitat in many ways and during

many key life stages [454]. There are a variety of channel condition indicators which may indicate effects on aquatic communities, including pool/riffle ratios, cross sections, width/depth ratios, sinuosity, gradient, scour, bank stability and others.

Biological Indicators

A complete biological assessment of sediment impacts in rivers requires an integrated assessment, comparing habitat (e.g. physical structure, flow regime), water quality and biological measures with empirically defined reference conditions (via actual reference sites, historical data and/or modelling or extrapolation) [371]. Indicators of impact are developed based upon changes in biological communities or differences from expected conditions. Biological indicators can be used to infer impacts of changes in key habitat characteristics (e.g. aggradation, degradation, suspended sediment, changes in channel diversity) on aquatic species. They are useful when aquatic habitats are key concerns, and enough is known about endpoint life histories and use of habitat. However, they may require extensive expertise, and it is important to bear in mind that cause–effect relationships between sediment sources, hydromorphological change and in-stream habitat impacts are poorly understood [454].

Barbour et al. [371] provided a detailed review of bioassessment tools and frameworks used in the USA for this purpose. The EEA [455] lists changing fish stocks and changes in aquatic plant communities, in macroinvertebrate communities and in diatom communities. Extence et al. [456] have developed a silt-sensitive invertebrate metric (PSI – proportion of silt-intolerant invertebrates) which can act as a proxy to describe temporal and spatial siltation impacts. When PSI is combined with flow and the velocity-sensitive metric LIFE (Lotic-invertebrate Index for Flow Evaluation), spatial differences in sediment–flow interdependencies may provide information on catchment sources of fine sediment to rivers. This may help to separate issues of sediment source and sediment impact as a result of hydromorphological change. The predictive programme RIVPACS [457] models the unstressed invertebrate community expected at a site from that site’s physical and chemical characteristics, which can be integrated with PSI for greater standardisation. De Jonge and De Jong [452] described changes to primary producers and ultimately to many ecosystem components as a result of increased turbidity from dredging. MAG-ACQUE-Thetis [458] observed that, although contaminated sediment is an issue in the Venice Lagoon, a much greater driver of ecological change is the loss of habitat diversity due to historical loss of sediment inputs into the lagoon. It is important to remember that biological endpoints show the combined effects of all stressors to which they are exposed and that separating the impacts of sediment as a source from the impacts of flow and hydromorphological change and other issues is not a trivial issue.

Reiser et al., among others [459–461], reviewed the impacts of turbidity, suspended solids and sedimentation on river ecological health and addressed the

ecological impacts of habitat loss due to sediment deficits [462]. Moreover, Reiser et al. [459] examined the effects of different types of sediment influxes to river gravel beds on invertebrate and fish communities and how this affected ecological recovery sequences. They found that both the duration and volume of sediment inputs were important factors in evaluating ecosystem response. They categorised sediment inputs ranging from short duration/low volume (SDLV) to long duration/high volume (LDHV), the characteristics of which affected the intensity, type and duration of impact and recovery.

2.9.2 Confirmation of Sediments and Suspended Matter as Causation of Degradation

As discussed throughout, sediment imbalance can be seen as a stressor by itself or an indicator of exposure or impacts from other stressors (flow and hydromorphology, invasive alien species, eutrophication, contamination). These processes are intimately linked, and it is not possible to address sediments alone as causation. Management of sediment requires an understanding of the complex interactions between these issues at the site specific and basin scale, and sediment only becomes a problem when these other issues are not managed.

Exposure Assessment

As a mobile connecting medium between various parts of the ecosystem via the hydrological cycle (see Fig. 1 in [4], this volume), sediments both contaminated and uncontaminated play both positive and negative roles in the viability and sustainability of social, economic and ecological objectives. How these roles are interpreted depend upon whether sediment status (defined in terms of sediment quality, quantity, location and transport) is appropriate to the needs of a given endpoint [453].

As most (non-contaminated) sediment impacts are a result of sediment imbalance rather than just the (necessary) presence of sediments in the system, indicators of exposure can often be seen as indicators of imbalance. The focus of this section will be on the various issues that must be considered to address exposure for basin-scale sediment assessment.

It is worth noting that sediments are generated and released throughout the basin (see [4], this volume). Thus, impacts that are the result of excessive or inappropriate deposition (e.g. the fouling of gravel beds) are affected by both intrinsic (e.g. slope, flashiness, channel characteristics) and management factors (e.g. channel modification, dams and weirs) that affect flow. Impacts that are a result of suspended sediment are further affected by natural processes (native animal activity, wind) and disturbances (invasive species, poaching, navigation, dredging) [463]. These issues are affected by processes (e.g. reduced flow (Sect. 2.8), hydromorphological

change (Sect. 2.7) or invasive alien species (Sect. 2.6) that are addressed in other sections of this chapter.

Direct indicators of these imbalances can be observational (e.g. visual reports of water colour, turbidity, or sedimentation), they can be more quantitative (sediment collection or trapping to determine sediment quantity and transport samples collected can also be analysed to determine their physicochemical characteristics and contaminant content and other measures of sedimentation or erosion rates), or they can be modelled. These issues are briefly addressed in the source indicators section below and are reviewed in [454, 461] and references therein.

Water column sediment indicators include measures of suspended sediment (the fraction of sediment load suspended in the water column), bedload sediment (the portion of sediment load transported downstream by sliding, rolling, or bounding along the channel bottom) and turbidity (a measure of the amount of light that is scattered or absorbed by a fluid, used as a measure of cloudiness in water, which can be affected by suspended sediment and the presence of organic matter).

The European Commission's Freshwater Fish Directive [464] seeks to protect those freshwater bodies identified by member states as waters suitable for sustaining fish populations. For those waters it sets physical and chemical water quality objectives for salmonid and cyprinid waters. The Directive presents a guideline standard of 25 mg/L for suspended solids based on prevention of chronic damage to fish gills. However, many argue that a specific standard for suspended sediment does not take into account many factors that may increase or attenuate endpoint vulnerability, and much work is still being carried out to address these issues (e.g. [460, 465, 466]). An important feature of the assessment of sediments at the river basin scale is that 'sediment sources', over time, are not only able to impact endpoints nearby but also, as they are moved, can impact (positively or negatively) endpoints downstream. Thus, although sediment sources may need to be managed at the local scale, impacts must be considered at the basin scale, and downstream endpoints are at risk from the cumulative releases upstream [167, 338, 453, 463, 467–478]. Thus, Apitz et al. [474] proposed an iterative, multi-scale approach to the management of sediments that considers basin-scale- and site-specific risks and options and their interactions (Fig. 9).

As sediments move downstream from any sediment source, exposure of endpoints of concern is a function of the relative locations of sources and endpoints. If a critical habitat of concern is upstream or in a different tributary than the source of concern, it will not be impacted by pollution. Furthermore, a river reach with rare or extensive endpoints will be of greater concern than one where one ubiquitous endpoint is exposed in only a small area. Indicators of exposure of this sort involve an understanding of the locations of potential impact on endpoints of concern relative to sediment sources and hydrological flow. Spawning gravel upstream of a sediment source is not at risk, while one downstream may be. Sediment management at the river basin scale requires a mapping of the locations at which objectives and their indicators (biotic and abiotic endpoints) can be impacted relative to sediment sources and transport. Such endpoint locations depend upon the endpoint and the mode of impact—endpoint locations can be the water column in a given

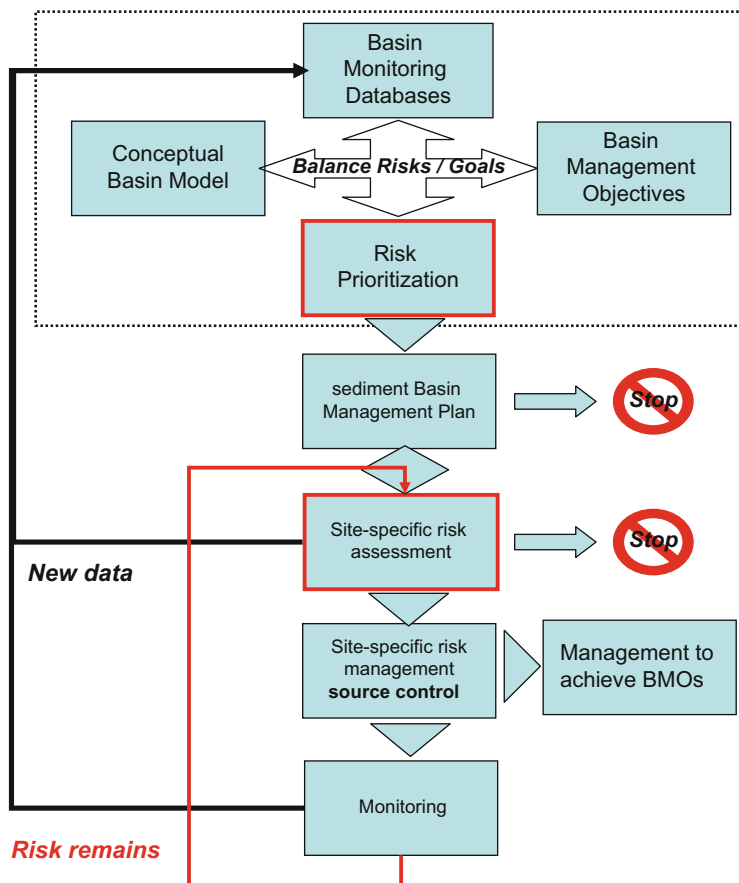


Fig. 9 Process diagram for basin-scale- and site-specific sediment risk assessment and management. Sediment-ecosystem regional assessment considers the role of sediment in sustaining basin objectives and ecosystem service sustainability at a basin scale (ref below). A manager may “enter” the process at the basin scale, or at the site-specific scale. Note: while the focus of this figure is on sediment risk management, the same concepts could be applied to other media. Adapted from [453]

location, water abstraction points, bottom habitats, floodplains, wetlands, etc., depending upon the sediment impact of concern [453].

Source Identification

One attribute of sediment status is sediment quality, which can be defined as the composition of a given unit of sediment in terms of grain size, organic content, nutrients, contaminants and pathogens [453]. While there is often a significant focus on the role contaminants play in sediment quality, other quality aspects, such as grain size, organic content, nutrient levels and the presence or absence of pathogens, are as

important in the role sediment plays for some endpoints, and thus the characteristics of sediment sources are quite important as well. Different sources will release sediment-related stressors with different characteristics, and thus the characteristics of sediment released are as important as the volumes when risks to endpoints are being evaluated. This section addresses indicators of sediment inputs into river basins.

Sediments have a broad range of sources (see [4], this volume) that are a function of land-use type and basin characteristics. They only cause quantity-related impacts to river basin objectives and endpoints if there is an ‘imbalance’ between sediment inputs and transport. Sediments move, over time, from sources to streams and rivers, between locations in river basins and eventually to coasts and the sea [454].

Land-use practices have significant effects on rates of sedimentation and on watershed hydrology, but landscape characteristics alone (whether altered or not) also help define the sediment yields of a given parcel of land and its vulnerability to changes. Indicators of sediment sources or source risk can be relatively simple, based upon the presence or absence of certain land uses or characteristics, or they can address the interactions between these processes. The complexity of sediment processes results in a range of assessment tools and indicators of sediment source.

Walling et al. [465] reviewed a number of methods for assessing sediment yields, the details of which are outside the scope of this chapter. There are a few categories of source assessment methods for estimating soil erosion, storage and delivery. These are reviewed in some detail in [454]; only some of the advantages and disadvantages will be mentioned here:

- *Indices including estimates of erosion vulnerability or future erosion:* These methods do not provide quantitative sediment loads, but they can be used to compare the relative vulnerabilities of different areas based upon specific land-use practices and river basin characteristics.
- *Erosion models, which can evaluate source loading alone or also the delivery process:* Simple models aggregate large areas based on empirical relationships and expert judgment, and are generally good for comparisons but not for quantitative load estimates. Mid-range models compromise between empirical and mechanistic approaches, providing order of magnitude estimates, but require more data and calibration. Detailed, generally mechanistic models can delineate sources at fine scale to greater accuracy and address transport and management issues, but are much more demanding of data and validation.
- *Direct estimations based on sediment budgets:* Direct measurements can provide direct evidence of sedimentation from specific sources and can, over time, be used to develop estimates of long-term erosion or to calibrate and validate models. However, they can be time and resource intensive, and given the spatiotemporal variability of sediment input and transport processes, results may not be representative, nor may they be a good basis for predicting how sedimentation may respond to future conditions.

Apitz et al. [479] developed a conceptual model in which various types of sediment sources (within rural landscapes, urban landscapes, non-urban roads, point sources and river banks) can be ranked at a field scale and then aggregated to a risk region scale to inform river basin sediment management (see Fig. 10).

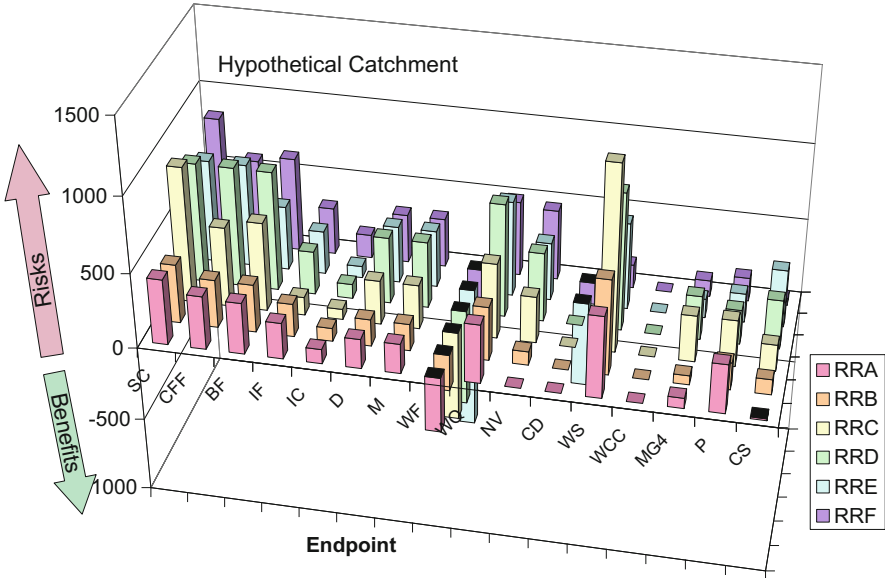


Fig. 10 Example output from the Sediment Regional Risk Model, showing the cumulative risks and benefits of sediment to a range of biotic and abiotic endpoints in six risk regions (RRA-RRC) of a hypothetical catchment. Endpoints: *SC* salmonids/cyprinids, *CFF* column feeding fish, *BF* Bottom feeding fish, *IF* Invertebrates/fine, *IC* invertebrates/coarse, *D* benthic diatoms, *M* macrophytes, *WF* waterfowl, *WQ* water quality, *NV* navigation, *CD* Coastal defence, *WS* Water storage capacity, *WCC* Water conveyance capacity, *MG4* MG4 plants, *P* Property, *CS* compliant sediments. From [480]

Rather than just *extent* of sources in a risk region, source strength was ranked as a function of intrinsic risk parameters that affect how vulnerable soils are, management practices that increase or decrease those risks and connectivity issues that affect whether sediments, when released, can reach river basins. They then developed numerical approaches to developing and aggregating these ranks based upon site-specific information. This approach ranks the risk of a parcel of land as a source of sediment based upon its combined likelihood to be subject to compaction, run-off, poaching, water and wind erosion and transport to the river. The model focuses on these specific processes rather than on specific land-use classes. Thus, risky practices on a vulnerable landscape are a greater source of sediments (as other studies have shown), and different land uses can be evaluated in a consistent manner. A similar approach is applied to the other source classes, with class-specific modifications. The purpose of this approach is not to provide comparable quantitative estimates of sediment loads but to provide a consistent, transparent approach to ranking the relative risk of sediment release into river basins in a spatially explicit manner that is sensitive to changes in management practices that may affect source strength or connectivity. These results then feed into the sediment risk ranking model to rank the risks of these releases to various endpoints and objectives at a river basin scale [479]. Using this approach, sediment source risk

and impacts can be evaluated as a function of source category, sub-basin, land parcel and specific management practice, depending upon how the data are aggregated and what question is being asked (Fig. 10).

2.9.3 Considerations for Site-Specific Risk Assessments and Downstream Risks

Due to the need to both dredge and dispose of sediments to maintain waterways and to manage the impacts of extensive human activities, sediment management is an important issue. Until recently, most regional sediment risk prioritisations and site-specific sediment Ecological Risk Assessments (ERAs) have focused primarily on the risk of contaminants in sediments on associated organisms. Occasionally, some food chain effects are explicitly or implicitly addressed, but rarely are contaminated sediments assessed within an overall ecosystem model. Sediments figures extensively in the Marine Millennium Ecosystem Assessment [481], but contaminated sediments are not the dominant issue. Rather, habitat change and loss, due to changes in sediment inputs, were major drivers, with significant consequences for biodiversity and ecosystem function.

The European Sediment Network SedNet has extensively discussed the issues of managing sediment at the basin scale; many of the outputs have addressed the management of both sediment quality and quantity at various scales, and they present frameworks, case studies and guidance on various aspects of these issues (e.g. [445, 467, 468, 477, 482–485]). While there are a wide range of tools to address various aspects of these problems in great detail, many of these approaches, if done correctly, are expensive and time consuming. Thus, regulatory and monitoring programmes that must address a wide variety of complex, interacting uncertain and spatiotemporally variable processes at many scales face severe resource limitations.

The WFD requires management at the river basin scale to achieve good ecological and chemical status of all surface waters and good chemical status of groundwater [3]. Management of sediment, as a pollutant in its own right, as a vector for other stressors and as an integral part of healthy habitats is not directly required in the EU WFD. However, there is widespread recognition that if sediment-related issues are not addressed, good status may be unachievable. Therefore, several draft RBMPs talk of producing a sediment management plan for their river basin. To achieve this, there is a need to develop approaches for dealing with sediment as a stressor, a vector of contaminants and as a supportive habitat in an integrated way that will enable appraisal of targeted mitigation options (by sector/source) to determine the best combinations for implementation for specific pressure/issue management or overall environmental improvement. Understanding and managing the dynamic interactions of sediment status on a diverse range of endpoints and services at the landscape or watershed scale should be the focus of sediment management [453]. Apitz [453] reviewed these issues and sought to provide a language and conceptual framework upon which sediment-ecosystem regional assessments (SECoRAs) can be developed in support of that goal.

3 Conclusions and Recommendations

Different physical, chemical and biological factors have been identified that are able to impact river ecosystems. As these stress factors often occur in conjunction, individual contributions to the deviation from the reference status have to be assessed to identify predominant stressors. Surveillance monitoring as required by the WFD may provide a valuable database for impact assessments and their, if desired, subsequent management. For this it is highly recommended to measure relevant chemical, biological and habitat parameters in an integrated approach, i.e. at the same sites and at approximately the same time. Gradient analysis of stressor-specific abiotic and biotic indicators may help to identify stressors likely responsible for the deterioration of single water bodies in the absence of other indications. However, despite a long tradition of using biological metrics in bioassessment (e.g. saprobic system), these methods are not yet available for all kind of stressors or for all trophic levels concerned [48]. Besides that, an array of statistical models and approaches are available to draw maximum benefit out of the monitoring data (see [5], this volume). Integrated monitoring and modelling approaches may thereby significantly enhance cost-efficiency of monitoring and provide good hypotheses on potential stressors impacting the ecological status in a certain water body. To verify these hypotheses and thus to optimally focus mitigation measures, this chapter provided tools and approaches for causal pathway assessment to identify and possibly confirm potential stressors in the frame of investigative monitoring programmes, as suggested by WFD. Scientists and water managers involved in monitoring and assessment of European river basins are encouraged to apply and further advance these tools. Finally, before managing single sites individually, local status assessments should be put into the perspective of downstream risks to identify basin-wide stressors and to select appropriate local measures.

However, in a multiply stressed ecosystem, the assessment of individual stressors does not necessarily provide sufficient understanding of the complex interactions between different environmental factors and stressors on the one hand, and communities with complex interactions, such as predation and competition, on the other hand. Global change will impact basically all processes and interactions between abiotic factors and biota as well as the structure and interactions within the communities. Promising novel approaches have been developed, which may help to unravel adverse effects of complex mixtures of contaminants together with a multitude of environmental stressors to aquatic communities. One example is the ‘adverse outcome pathways’ (AOPs) approach that describes the linkage between a direct molecular initiating event (e.g. a chemical that interacts with a biological target site), leading to a cascade of higher order effects at the organism level that eventually produces an adverse outcome at the population level (e.g. impaired reproduction), which is relevant to risk assessment [486, 487]. Using impaired vitellogenesis by oestrogen antagonists, inhibitors of aromatase activity and androgen receptor agonists as an example, Ankley et al.

demonstrated how AOPs may explain the interaction of various chemicals acting on different pathways that converge at intermediate steps into a similar effect [486]. It may be hypothesised that the AOP approach will also be promising to understand and predict the joint effect of endocrine disruptors and adverse habitat properties for spawning on the reduced reproduction of certain fish species. Hence, AOPs may also help to develop a better understanding of stressor interactions as addressed in chapter by von der Ohe, et al. [5], this volume.

Taking up the AOP concept system biology approaches unravelling and modelling the integrated, interacting networks of genes, proteins and biochemical reactions, or even higher order biological units that give rise to function and ultimately support life [486] will open new opportunities to assess multiple stresses. Williams et al. addressed the challenge of identifying molecular mechanisms of compensatory and toxic responses from the transcriptome, the metabolome, conventional biomarkers and data on contamination with a novel network inference strategy [488]. The authors found that the overall molecular state of the livers of flounders sampled from marine environments were representative of the chemical contaminant burden of the corresponding sediments and that chemical exposure could be partly predicted from the molecular state of the flounder livers. Although promising, systems biology-based approaches are still clearly beyond of what can be applied in regular monitoring and assessments in multiply stressed European river basins. However, first successful studies demonstrate the potential of these tools to unravel causal pathways and to develop novel biomarkers for monitoring which can be translated into endpoints meaningful to ecological risks.

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Monitoring Programmes, Multiple Stress Analysis and Decision Support for River Basin Management

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Abstract The identification of plausible causes for water body status deterioration will be much easier if it can build on available, reliable, extensive and comprehensive biogeochemical monitoring data (preferably aggregated in a database). A plausible identification of such causes is a prerequisite for well-informed decisions on which mitigation or remediation measures to take. In this chapter, first a rationale for an

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extended monitoring programme is provided; it is then compared to the one required by the Water Framework Directive (WFD). This proposal includes a list of relevant parameters that are needed for an integrated, *a priori* status assessment. Secondly, a few sophisticated statistical tools are described that subsequently allow for the estimation of the magnitude of impairment as well as the likely relative importance of different stressors in a multiple stressed environment. The advantages and restrictions of these rather complicated analytical methods are discussed. Finally, the use of Decision Support Systems (DSS) is advocated with regard to the specific WFD implementation requirements.

Keywords Decision support systems • Integrated assessment • Investigative monitoring • Multiple stress • Weight-of-evidence

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1 Monitoring of Relevant Stressors

While natural hazards, such as heavy floodings of human settlements, may have been recorded for several centuries in some places, regular and predefined monitoring programmes of the indirect effects of man-made changes to the environment have only been carried out since the beginning of the last century. At that time, high organic loads stemming from untreated wastewaters led to local oxygen depletions and consequential fish kills. Based on the observation that macro-invertebrate communities also changed along water quality gradients following distinct organic inputs, Kolkwitz and Marsson [1] introduced in 1909 one of the first biological indicator systems, the ‘saprobic system’ (see also Sect. 2.4 in [2], this volume). In the early 1970s, maps based on that saprobic index revealed the still ongoing

deterioration of water quality of many streams in Germany and Europe due to high loads of organic pollution from untreated wastewaters [3]. This led to a European-wide introduction of sewage treatment plants, reducing organic inputs and thus improving water quality (see [4, 5]). However, biodegradable organic load and the subsequent depletion of oxygen are not the only pressures that occur in many European river basins. Following the introduction of the Water Framework Directive (WFD, see [6], this volume), all European Union Member States were required to submit Article 5 status reports (i.e. Pressure and Impact Analysis) for their water bodies to identify major causes of deterioration. An overall analysis of these reports by the European Commission (EC) revealed that degradation was most frequently attributed to hydromorphological changes and to eutrophication due to high nutrient inputs from agricultural practices ([7]; see also [2], this volume). Furthermore, this analysis also suggested that the relative importance of diffuse and point source pollution were inadequately described [7].

However, these general conclusions on hydromorphological changes and eutrophication on the European scale are not sufficient to manage the deterioration within a specific water body. The improvement of a water body status requires a site-specific identification and management of the relevant stressor sources throughout the water body and the whole river basin [8–13]. This calls for an integrated water body status assessment, considering all potential stressors simultaneously, to identify the likely causes of deterioration. Suggestions for potential biological metrics and physico-chemical indicators, which can be used for the identification and assessment of specific stressors, were discussed in von der Ohe et al. [2], this volume. However, since stressors seldom occur in isolation, less pronounced effects of additional stressors or stressor interactions might be overlooked [14, 15]. A better understanding of the ‘risk sources–pathway–risk receptor’ interactions (see Fig. 2 in [6], this volume) in river ecosystems, and a more systematic approach to their assessment, is needed to be able to identify the driving factors that affect the ecological status and thus the functioning of aquatic ecosystems. This understanding is a prerequisite for well-informed evaluation and selection of measures for the restoration of impacted ecosystems or for the enhancement of the natural potential for recovery of heavily modified water bodies.

Hence, the aims of the present chapter are the following: (1) to discuss the advantages of historical monitoring programmes in comparison to current WFD-compatible monitoring programmes with regard to a sound multiple stress analysis; (2) to suggest modifications to current monitoring programmes to more effectively record all key parameters needed for these analyses; (3) to present novel, state-of-the-art, multivariate statistical approaches that enable the unraveling of potential causes of deterioration of ecological status and (4) to discuss the advantages of Decision Support Systems to support informed decisions by providing necessary computer tools with a user-friendly interface.

If the suggested modifications are applied, then a retrospective analysis of such surveillance or operational monitoring data could be used to iteratively identify the most degraded water bodies and the likely causes. Specific investigative monitoring programmes, as described in von der Ohe et al. [2], this volume, will then confirm

whether the stressors of concern are the likely causative agents for the observed degradation. See also Fig. 1 in von der Ohe et al. [2], this volume, for an overview of an overall monitoring and risk assessment approach. However, it should be noted that these novel approaches can only be applied successfully, if reliable, extensive and comprehensive biogeochemical monitoring data, preferably aggregated in a database, are available.

1.1 Historical Monitoring Programmes

Water quality monitoring programmes have a long history in many European countries. These time series allow for the analysis of long-term trends, if monitoring programme designs have not changed too much over time. However, these programmes were often based on local expert knowledge or specific local concerns and did not follow European or nationally harmonised protocols. Hence, comparisons between regions or countries can be difficult, if not impossible. Moreover, monitoring programmes were and still are often not designed with causative analysis of multiple stressors in mind and thus may not be useful for this purpose. Many historical monitoring programmes focus on stream-dwelling invertebrate communities, whereas other groups of organisms, such as fish or algae, were sampled less often [16, 17]. In addition, biological and chemical monitoring programmes were often not aligned, as different governmental institutions were responsible for them. There are exceptions; for example, in Catalonia [18], the monitoring responsibility has been delegated to a regional level. Moreover, data on physico-chemical parameters—such as temperature, electrical conductivity or pH—are most often available at similar quality in river basins across Europe. However, the availability and comparability of data on specific chemical analytes (e.g. certain pesticides or PAHs) often depends on specific investigations (e.g. agricultural versus industrial effluents) and differs even from site to site. In a recent study on the effects of chemicals on benthic invertebrates in the Scheldt basin, the number of toxicants analysed in the historical monitoring data ranged from 1 to 285 [19].

Since changes in biotic communities cannot easily be linked to specific chemicals, the Triad approach was introduced (see [2], this volume, and see [20]). This approach involves three simultaneously explored and complementary lines of evidence (LoEs): (1) analytical chemistry to identify and to quantify individual pollutants, (2) *in vivo* and *in vitro* bioassays to identify and to quantify toxic effects and (3) *in situ* biological assessment (community surveys) to identify and to quantify community alterations (although this third LoE is often replaced by a range of other assessments [21]). However, to date, routine applications of the Triad approach are rather rare in Europe. One example is the Flemish Environmental Agency (VMM) that routinely applies this approach [22], including chemical analysis of organic and metal compounds, the assessment of the benthic invertebrate community and the application of bioassays with *Hyalella azteca*. The focus on the benthic invertebrate community can be explained by the fact that the VMM approach focuses on the assessment of potentially polluted sediments. In most European countries, however,

environmental quality monitoring is limited to water samples, as required by the WFD [23]. Other matrices, such as sediment, suspended matter or biotic tissue have often been neglected, although they may contain much higher concentrations of metals and a significant fractions of the lipophilic organic toxicants in aquatic environments.

With regard to the selection of monitoring sites, they are often designated in support of specific monitoring programmes, e.g. focussing on contaminated hot spots or dredged material disposal sites, and thus insufficiently represent reference conditions. Hence, a potential approach to obtaining data on reference conditions is to use data from less impacted regions (see also [2], this volume). As a result, data on pristine or reference conditions are rather scarce, as was confirmed by a recent study on reference conditions based on historical monitoring data, where less than ten reference sites were included in each of three river basins, compared to several hundred potentially impacted sites [24]. However, even when reference or background data are available, it remains unclear how geographical variability may hamper direct between-region comparisons [25]. Moreover, data on habitat quality are often not available in a format or at a scale that is consistent with those of biological and chemical data. For instance, habitat quality parameters are usually recorded for 100 m stretches. Thus the challenge is to match data for these parameters to the Triad data that are usually obtained on point samples. For all of these reasons, a multi-stressor identification based on historical data can be difficult. In most cases the available data do not comply with all the requirements for an integrated assessment of all stressors [18, 19]. These requirements are further described in Sect. 1.3.

1.2 WFD Monitoring

The WFD calls for a different approach to monitoring than that which has been carried out in most historical monitoring programmes. European Union Member States are required to establish surveillance and operational monitoring programmes of the status of their water bodies but may also need to establish programmes of investigative monitoring [26].

Surveillance monitoring:

- Aims to monitor the actual status as well as to assess long-term changes resulting from widespread anthropogenic activity or natural processes
- Must be carried out at a sufficient number of sites to be able to provide an assessment of the surface water status of each water body within the river basin

Operational monitoring:

- Should be undertaken to establish the status of those water bodies that are at risk of failing to meet the environmental objectives of the WFD
- Is carried out to assess any changes in the status of such water bodies that result from the programmes of measures

Investigative monitoring:

- Aims to ascertain the causes of a water body failing to achieve the environmental objectives, where the reason for any exceedance of environmental quality standards (EQS) or environmental quality ratios (EQR) is unknown or where surveillance monitoring indicates that the objectives are not likely to be achieved and operational monitoring has not already been established
- Aims to determine the magnitude and impacts of accidental pollution to inform the establishment of a programme of measures for the achievement of the environmental objectives or to validate specific measures necessary to remedy the effects of accidental pollution [26]

The establishment of these programmes bears the risk that existing biological monitoring programmes may be phased out or be reduced in compliance with the new WFD monitoring concept. Moreover, the directive requires the concurrent analysis of at least three biological quality elements (BQE), namely, invertebrates, fish and algae, using standardised assessment methods. While the WFD allows Member States to maintain national assessment methods, it requires an inter-calibration of the methods' outputs (i.e. biotic metrics), namely, a similar classification of high and good ecological status [27]. However, for many biological quality elements (BQE, these play an important role in defining the ecological status), no metrics are available to address all relevant stressors [18]. As with historical monitoring, some WFD chemical and biological monitoring programmes are still carried out at different times and frequencies and with differing sampling locations and densities, making weight-of-evidence evaluations challenging. For example, the WFD requirements for the assessment of chemical status are restricted to a short list of currently 48 priority substances (PS) (see [28], and amending Directive 2008/105/EC [29]). The ecological status assessment may require the consideration of a few more compounds, based upon the list of river basin specific substances [26]. Hence, there is a risk that existing local and more comprehensive chemical monitoring programmes could be reduced in scope in compliance with the minimum requirements of the WFD.

As stated above, many lipophilic compounds are found at low levels in the water column and, hence, may be difficult to detect. To address this problem, the chemical status assessment is required to consider whole water extracts, although usually only a fraction of the total concentration is bioavailable to organisms. This may overestimate risks, as bioavailability of chemicals is usually lower in waters with high levels of dissolved organic matter (DOM) than in waters with low DOM. However, it is a conservative approach as bioavailability varies with water conditions (see also [2], this volume). More relevant matrices for the detection and risk assessment of such constituents may be sediment and tissue, but the monitoring of these compartments has often been neglected in WFD monitoring programmes thus far. This may change with the introduction of the new Technical Guidance Document (TGD) for the derivation of EQS. The TGD includes guidance on how to derive standards for sediment and biota [30].

It is important to note, however, that the WFD is a 'one-out, all-out' framework. This implies that if a water body fails due to any of the status metrics (e.g. chemical,

biological, hydrological), it is considered to have failed. The framework is not a weight-of-evidence (WoE) or risk-informed framework and does not consider the interactions between lines of evidence (LoEs). Such information can, however, be essential in the identification of relevant stressors and, thus, the design of programmes of measures to improve ecological status [31].

1.3 Monitoring Requirements for an Integrated Status Assessment

Many of the WFD surveillance monitoring networks have been designed to monitor the effects of habitat degradation or nutrient input with low sampling density at single points representing large areas. This may be adequate for these kind of stressors, as they tend to cause rather large scale impacts. However, these low sample densities may hamper the identification of other stressors that have rather local inputs (e.g. point sources of pollution), but whose impact may still propagate through aquatic systems, and hence might be overlooked or misallocated. At the same time, the installation of additional operational or investigative monitoring programmes will only be triggered, if a deterioration of the BQE at the surveillance monitoring sites is correctly identified.

Clearly, sampling design is a critical issue if overall status assessment is to be successful. Sampling should cover a variety of parameters to address all stressors of potential concern, and, if possible, their interactions and combined effects, and avoid neglecting the less obvious ones, such as toxicants or pathogens [32]. Moreover, it is important to describe geographical aspects of biological and chemical sampling sites with as much detail as possible. Assessment of habitat characteristics should quantify the level of habitat degradation at the microhabitat scale [33, 66]. This section outlines the sampling requirements for an integrated river basin assessment based on a survey of all 'local' stressors and thus for a sound system understanding, applying a tiered approach. Only if certain trigger values are exceeded (e.g. EQS or EQR), may this lead to investigative monitoring programmes to evaluate cause-effect relationships (see also [2], this volume).

However, true reference sites can be difficult, if not impossible to find (even remote sites may suffer anthropogenic impact, e.g. from airborne emissions), and may be subject to the effects of climate change [32]. Hence, it is suggested that assessments are based on gradient analysis approaches [34, 35]. Such approaches use a set of sites of different status classes to extrapolate from poor or moderate status to good or high status via regression analysis. Ideally, all surveillance monitoring sites should be sampled in the same way, so that they can be included in such gradient analyses. This would allow the creation of a database that is sufficient for data-intensive analyses, such as evaluations using the effect and probable cause (EPC) diagnostic model [36] or related ones, as described in Sect. 2. All relevant parameters that should be sampled in surveillance monitoring programmes designed to evaluate the integrated impacts of a disparate range of stressors are listed in Table 1.

Table 1 Overview of parameters to be sampled in routine monitoring programmes to allow for an integrated risk assessment of all stressors

Stressor	Parameter	Remark
Toxicity	<ul style="list-style-type: none"> • Chemical concentrations in water and suspended matter • Total organic carbon content (TOC) in water • Water hardness • Bioassays for all BQE, including in vitro assays and for endocrine effects 	<ul style="list-style-type: none"> • To assess the toxicity of river basin specific organics and metals • To estimate bioavailability of organics • To estimate bioavailability of metals • To identify hot spots
Acidification	<ul style="list-style-type: none"> • pH • Alkalinity and concentrations of base cations • Concentrations of sulphate and metals • Acid-neutralising capacity 	<ul style="list-style-type: none"> • To assess temporal variability in pH • To assess temporal variability in alkalinity and base cations • To assess the toxicity of dissolved sulphate and ionic aluminium • To quantify loss of buffering capacity
Salinisation	<ul style="list-style-type: none"> • Electrical conductivity 	<ul style="list-style-type: none"> • Indirect measure of salinity
Eutrophication/ oxygen depletion	<ul style="list-style-type: none"> • Dissolved oxygen (DO) • Nutrients • Biological oxygen demand occurring over a 5-day period (BOD₅) • Chlorophyll A 	<ul style="list-style-type: none"> • To quantify anoxia • To quantify eutrophication potential • To quantify organic load • To assess the eutrophication status
Parasites and pathogens	<ul style="list-style-type: none"> • Parasites on fish • Biomarkers in fish 	<ul style="list-style-type: none"> • Number and kind of parasites on fish • Infections and health status
Invasive alien species (IAS)	<ul style="list-style-type: none"> • Number of IAS 	<ul style="list-style-type: none"> • Specific sampling procedures to identify IAS in low abundance
Habitat degradation	<ul style="list-style-type: none"> • Coordinates • Elevation and slope • Stream order and drainage area • Alterations in channelisation • Structure and substrate of the river bed • Dynamics of flow conditions and turbidity • Structure of the river banks and riparian zone • Sedimentation and grain size • Colour and temperature • Connection to groundwater bodies • River depth and width variation 	<ul style="list-style-type: none"> • Many of these parameters are already covered by stream habitat surveys but should be carried out at the specific biological sampling site that is being assessed
Changing water levels	<ul style="list-style-type: none"> • Highest flow or discharge • Timing of high and low flows • Lowest water levels or the smallest flow • Hydrological duration curves of submersion 	<ul style="list-style-type: none"> • To assess the effect floods and their seasonality • To assess the effect droughts • To assess the impact of water level fluctuations
Sediment and suspended matter	<ul style="list-style-type: none"> • Assess excessive or insufficient levels of fine sediments • Pool/riffle ratios • Cross sections 	<ul style="list-style-type: none"> • To identify sediment-hungry water • To indicate sediment transport and balance

(continued)

Table 1 (continued)

Stressor	Parameter	Remark
	<ul style="list-style-type: none"> • Width/depth ratios • Sinuosity and scour • Bank stability 	<ul style="list-style-type: none"> • To indicate naturalness • To quantify erosion potential

The application of similar sampling procedures for all trophic levels of aquatic biota is another mandatory part of basin-wide integrated assessments. Species census data should be gathered according to a standardised quantitative sampling protocol that ensures that equal effort is put into sampling events at different sites. The taxonomic identification should preferably be at the level of individual species, or at least be consistent between samples. In order to assess temporal variability, each site should be sampled multiple times, and issues such as seasonality, migration, diurnal and tidal cycles and other temporally relevant natural parameters should be borne in mind in sampling design and data interpretation. If possible, additional sites should be identified that are minimally disturbed and demonstrate a natural and balanced species composition and biodiversity. Here, the sampling of known invasive alien species should also be a requirement, as their detection at low abundances is crucial for the evaluation along gradients and for the implementation of successful mitigation measures (see Sect. 2.6 in [2], this volume).

For an integrated assessment, a spatially and temporally integrated physico-chemical and biological monitoring programme is essential if the links between causes and observed effects are to be established. The existing infrastructure for chemical analysis should enable a rapid inclusion of novel compounds where needed. However, the trade-off between the number of chemicals analysed for and the cost of their analysis must be considered. Bioassays may help to identify hotspots of pollution. Such a pre-screening with bioassays may also identify ‘good’ status sites that do not require chemical analyses. Similarly, the modelling of transport processes for nutrients (see Sect. 2.4 in [2], this volume) and salts (Sect. 2.3 in [2]) should be linked to the monitoring of respective stressor parameters in order to increase system understanding (see next section).

2 How Does One Assess Multiple Stressors?

Ecosystem integrity is generally indicated by the occurrence and abundance of characteristic species, appropriate biodiversity and proper functioning in terms of nutrient cycling and energy flux. A rather simplistic but widely applied and justifiable paradigm in ecological risk assessment is that ‘an ecosystem is protected when all species belonging to that system are able to survive and reproduce’ [37, 38]. Ecological risk can thus be defined as the proportion of species liable to be affected in their well-being.

Ecosystems are subject to a range of anthropogenic stress factors such as habitat degradation, toxicants, eutrophication and oxygen depletion, dredging, shipping,

salinisation as well as competition or introduction of invasive alien species, etc. (see chapters by Négrel et al. [40] and by von der Ohe et al. [2], this volume). These different types of stressors may affect species composition (community structure and abundance of single species) of exposed ecosystems and the rate of ecosystem processes (function) [41]. The presence of one specific stress factor may also change the sensitivity of the ecosystem to other stressors. For instance, increasing water temperatures arising from climate change may influence the environmental distribution and metabolism of chemicals, which in turn would alter the exposure of organisms to these chemicals (see [41]). At the same time, increasing water temperature influences bioaccumulation of chemicals, since the elevated temperature accelerates ventilation of the organisms, increasing the uptake of these chemicals. On the other hand, elevated temperature may also increase the rate of biotransformation, thereby reducing body burdens of the same chemicals. These non-linear and indirect interactions are factors, amongst others, that make the assessment of combined impacts of multiple stressors difficult [42].

A stressor identification process as shown in Fig. 1 provides a formal and rigorous tool that enables the identification of stressors causing biological impairments in aquatic ecosystems [43]. It also provides a structure for organising the scientific evidence supporting the conclusions on the likely causes. A critical initial step in the evaluation of the roles of different stressors in ecological impairment of a river site or basin is to establish a conceptual model that leads to a list of candidate causes (see Fig. 1).

A conceptual model can be a visual representation of the stressors along with their (known or assumed) direct and indirect effects [44]. Once such a list of candidate stressors is established, a weight-of-evidence (WoE) approach offers a structured, transparent post hoc methodology to identify the most likely causative factor(s) [45, 46]. WoE diagnostic procedures have been designed by several other authors (e.g. Sediment Quality Triad [47] and in situ community experiments [48, 49]). These methods were introduced to assess the effects of chemical pollution and to integrate chemical monitoring data with single or multiple species bioassay experiments and ecological field studies. In this way, the most likely causes of biological ecosystem impairment can be identified. These developments have greatly enhanced our ability to attribute the underlying causes to the observed impacts in a qualitative sense. However, the approaches do not allow for a quantitative evaluation of the stressor impact. Continuous biological monitoring often indicates a shift in species composition, whose underlying causes cannot be identified easily, as species composition represents a response that integrates the impacts of a variety of factors and stressors. A first requirement for the identification of likely factors is the availability of information on reference conditions for the species, the community and the functioning and health of the particular type of ecosystem under consideration [50–55]. Assessments of aquatic ecosystem health or status often focus on ecological functioning. As well as their contribution to the maintenance of biodiversity, near-natural, unpolluted streams have various functions of direct importance to society, e.g. supply of purified water, recreational value (enjoyment of nature) and retention of water and sediment, etc. These are often termed ecosystem services (see further at [57], this volume).

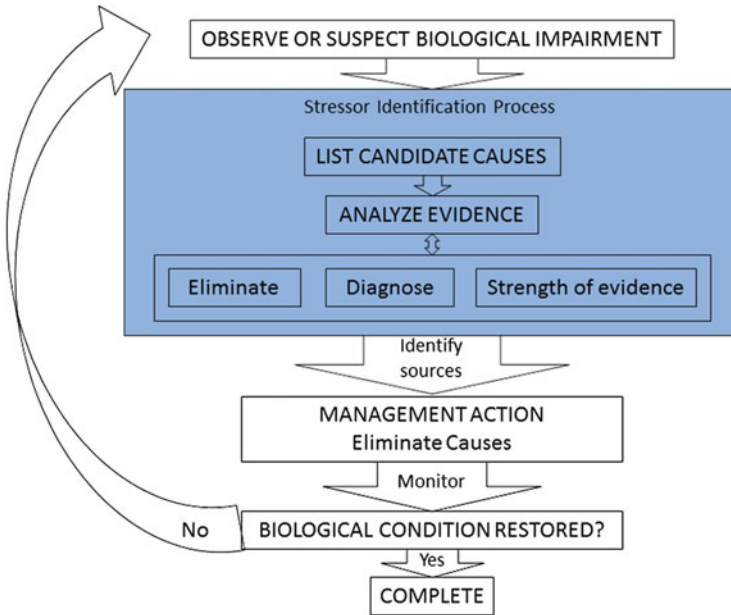


Fig. 1 Suggested stressor identification (SI) process in river basin ecosystems. The SI process is shown in the *blue-shaded*, centre box. SI is initiated with an observed or actually detected biological impairment. Decision-maker and stakeholder involvement is particularly important in defining the scope of the investigation and in listing candidate causes. Data can be acquired at any time during the process. The accurate characterisation of the probable cause allows managers to identify appropriate actions to restore or protect ecological status. However, as river basin ecosystems are complex, uncertainty about causal links and the effectiveness of management actions always remain. Thus, adaptive management practices, including monitoring, and, if necessary, iterative processes are essential for ecosystem management (adapted from [43])

High ecological quality is associated with high ecosystem functionality. The long-term, sustainable use of stream ecosystems urges the definition of quality or reference targets. When a deviation from a target has been observed, stressor identification may then proceed via stressor elimination and diagnostic evaluation, followed by an analysis of the WoE for each of the remaining candidate causes. After stressor identification, ‘eco-epidemiological gradient analysis’ may quantitatively reveal the contribution of the different types of stress to the observed, overall ecological impact [42]. Eco-epidemiological gradient analysis, which relates a quantified measure of toxicity to observed ecological effects in the field, has not yet been widely applied. This is mainly due to a lack of the high-quality and high-density datasets needed to perform this type of study [42].

2.1 *Selection of Reference Conditions and Sites*

In ecology, the concept of reference condition is increasingly used to describe the standard against which impacted sites are compared in order to be able to separate significant, anthropogenic impacts from natural variation [57]. Especially in the WFD, the European Union has adopted the concept of ‘reference conditions’ in a legal context with the aim to protect and improve ecological status [58]. However, the phrase ‘reference condition’ has many meanings in a variety of contexts. In our context, ‘reference condition’ should be used solely for referring to biotic integrity, defined in terms of ‘naturalness’ of the biota (structure, composition, abundance single species, function, diversity of species), where naturalness implies the absence of significant anthropogenic disturbance or alteration. Thus, for this section, the term ‘reference condition’ refers to a natural condition of biodiversity as it would be with the respective physico-chemical boundary conditions, in the absence of anthropogenic disturbance. As such, reference conditions have to be formulated for different types of water bodies (e.g. lakes, small streams, rivers, etc.). If adequate numbers of representative reference sites are not available in a region, or for a particular type of water body, data from similar regions or types of water may be used to simulate reference conditions. While this approach defines ‘reference’ in a spatial context, reference conditions may also be formulated in a temporal context, i.e. comparing recent data to historical data [25]. This method is commonly used in areas where anthropogenic stress is widespread and unperturbed references are few or lacking entirely. From an eco-engineering point of view, any regional ecological community of a desired typology can be used to define reference conditions, provided that auto-ecological information¹ is available. Reference conditions can also be established using expert judgment in areas where reference sites are lacking or scarce, but caution should be exercised to avoid subjectivity and bias in case of a lack of quantitative data for validation. One of the major problems in selecting reference sites used to derive reference conditions is the fact that the absence of impact from stress factors, such as toxicity, cannot be evaluated from superficial visual examination alone. This is the case as toxic and other effects can have occurred at other times or locations, and thus the origin of toxic circumstances may be located far away or in the past. Therefore, a better approach is the derivation of reference conditions from a thorough data base encompassing data for both biota and a wide variety of habitat and environmental parameters (Sect. 1.3, this volume).

Most commonly, reference conditions are defined by a range of index values. This implies that reference conditions are described as a distribution rather than a single absolute value. The range of values results from natural variability, both in time and in space. At any point in time, a set of sites, all in undisturbed condition, will exhibit a range of biological attributes. Sites with intact biotic integrity will display variation over time, due to natural disturbance. Once this distribution is

¹That is, information describing the physico-chemical requirements that species present have regarding their habitat

described, parameters in that distribution can be selected as thresholds for classifying the condition of individual sites and ultimately for assessing the sites of interest. The selection of these thresholds can be as much a policy decision as a scientific one. In the following paragraphs, statistical approaches are described that allow for the discrimination of anthropogenic impacted from reference sites.

2.2 *Quantifying Biological Conditions Using RIVPACS Modelling*

The comparison of observed fauna with that expected to occur in the absence of anthropogenic stress provides a basis for the quantification of the biological condition of potentially stressed ecosystems. RIVPACS²-type models [51, 53] estimate the probability of capturing a species from the regional species pool at a specific location. As such, RIVPACS-type modelling allows for the generation of an estimate of a local biological reference state. The local probability of capturing a particular species is used to estimate the 'observed outcome/expected outcome (O/E) ratio', where 'E' is the number of species expected at a site and 'O' is the observed number of the predicted species present. The O/E ratio is an intuitive and easily interpretable measure of biological condition, i.e. the proportion of species present that were expected. The RIVPACS system, originally designed for the evaluation of the river invertebrate community structure, can easily be adapted to cope with other groups of fauna as well as with flora. RIVPACS models are built from data collected from a network of non- or minimally stressed reference sites that span the range of naturally occurring conditions in the region of interest. Given a standard sampling effort, RIVPACS-type predictive models assume that the probability of capturing (P_c) is a function of the preference and tolerance of different species to naturally occurring environmental conditions. Reference sites are classified into biologically similar groups so that any variation in species occurrence between sites can be associated with these groups. The biologically identified groups of reference sites are then explained by predictor variables that are not easily modified by human activities (e.g. longitude, latitude). Based on the model output for the reference sites, the ideal species composition for any other site can then be predicted. Comparison of the local reference with the locally observed species pool will identify the species that are presumably lost from the ecosystem.

2.3 *Multimetric Analysis*

In the United States of America, the protection of biological integrity is the main objective of the 1972 Clean Water Act [59]. Originally, it was assumed that biological integrity could best be evaluated by measuring physical and chemical variables.

²RIVPACS = River Invertebrate Prediction and Classification System

However, measuring these variables can be impractical and/or expensive, and related biological effects can depend on the interactions between the variables and how abiotic variables vary in time and space. More recently, the direct measurement of biological variables, expressed as biological indicators, plays an important role in environmental management. Comparing the results of these indicators between sites, or with respect to standards based on reference (i.e. relatively undisturbed) sites, may help guide assessment and restoration efforts. To facilitate such comparisons, individual indicators were combined into overall multimetric indices to summarise biological condition and identify degraded sites. The best-known multimetric indices are the 'Index of Biotic Integrity' (IBI) for fish [60] and the 'Invertebrate Community Index' (ICI) [61, 62]. Both indices are calculated by selecting a set of biological indicator variables, such as numbers of species in various classes [like indigenous species, insectivores, 'intolerant to degradation' or the percentage of Ephemeroptera, Plecoptera and Trichoptera (EPT)]. The metric values are assigned numerical scores, based on a comparison with the values at undisturbed sites. Values at minimally disturbed sites are assigned a score of 5; those that deviate to some extent from such sites receive a score of 3; and those that deviate strongly are assigned a score of 1. It is claimed that the combination of different biological measures into a single metric is more precise than single indicators and more sensitive to many types of degradation. These multimetric biological indicators may be very useful in detecting deviations from biological reference conditions and may be used for the identification of reference sites. However, their differing responses to different stressor types may hamper the identification of the types of stressors which are causing the deviations from the biological reference conditions. Hence, what is needed in ecosystem diagnosis are biological indicators that are sensitive to particular types of stressors [24]. These will be helpful in distinguishing effects of particular stressors from confounding factors (see [2], this volume, for definition). Several of these types of metrics are available. The 'Biological Monitoring Working Party' (BMWP) and 'Average Score Per Taxon' (ASPT) scores are used to detect general degradation [63]. The 'Nutrient Biotic Index for Nitrate' (NBI-N) and for Phosphorus (NBI-P) indicate effects of nitrogen and phosphorus eutrophication [64]. The 'Species at Risk' (SPEAR) index (see also [2], this volume) indicates toxic effects [65] or habitat degradation [66]. The specificity of these indices relies on the classification of taxa or their traits based upon their sensitivity to specific stressors. As well as stressor-specific indices, analytical methods such as multivariate statistics or artificial neural networks are required to indicate effects of specific stressors [18, 67].

2.4 *Quantitative Diagnostic Modelling*

Assessing the degree of alteration and assigning causality often requires a wide array of tools. Methods for measuring the magnitude of biotic degradation in aquatic communities are well developed (e.g. [51, 68]), but WoE diagnoses of probable causes rely on various combinations of expert judgment and the application of multivariate statistics. Unfortunately, these methods often require a great deal of

expertise to use and interpret, and their results are often difficult to communicate. Furthermore, mixtures of potentially toxic compounds are usually not taken into account in such assessments. Although identification of the causes of biological impairment may always require application of sophisticated tools, there may be more elegant and effective ways of presenting complicated information. A method would be welcomed that communicates both the magnitude of impairment and the likely relative importance (i.e. related to causing the impairment) of different stressors.

All currently existing methods for diagnostic evaluations of biological water quality require extensive quantitative and qualitative information in order to be able to establish the degree of biological deterioration. Importantly, the actual biological status must be described and compared to the status that would occur under pristine conditions. However, due to the enormous variability in biological communities and the complexity of biological, physical and chemical interactions, it is virtually impossible to identify a reference condition that fully reflects completely natural conditions. Here, a gradient approach (see Sect. 1.3) or the use of controls instead of reference sites may help. While a reference site is a location that is chosen to be as close to the state of an environment undisturbed by human activity, a control site is chosen to be as similar as possible in all respect to the impacted location [58]. Moreover, ecological diagnosis requires quantitative information on a variety of environmental factors that may cause biological deterioration. Selection of stressors to be included in the analysis requires a thorough knowledge of the normal operating ranges (i.e. range of observed parameter values, see Sect. 1.3) of the potential stress factors.

Once the biological effects as well as the potential stressors have been quantified, the process of diagnosis continues with an analytical step to relate deviations in species composition in comparison to a reference state. In a qualitative sense, this may be accomplished by building evidence for causality in a careful iterative process of reasoning called 'Stressor Identification', as presented in Fig. 1 [43]. A more quantitative approach may involve the use of statistical techniques to suggest associations between environmental factors and observed effects. Statistical diagnosis always relies on gradient analysis where diagnostic evidence is inferred from the strength of association, as presented in Table 2. In order to retain analytical power in a statistical analysis with many stress factors involved, it may be worthwhile to aggregate similar stress factors to one measure of overall stress. This can for instance be done by replacing the concentrations of individual toxicants with an aggregated measure of overall toxic risk [18, 69], for instance using the concentration addition and response addition models for deriving Toxic Units (TU), as explained in Sect. 2.1 in [2], this volume.

A major complicating factor in inferring causality between observed ecological impairment and candidate stressors is the fact that propagation of effects along the biological hierarchy is often not linear (see Fig. 2 in [2], this volume). For instance, a toxic cellular effect does not necessarily lead to a toxic response at the organism level due to the existence of compensatory mechanisms at the supra-cellular level [70]. For this reason, the use of biomarkers to predict ecological effects can be

Table 2 Types of associations between measurements of causes and effects among site data and the evidence that may be derived from each

Type association	Example evidence
Spatial co-location	<ul style="list-style-type: none"> • Effects are occurring at the same place as exposure • Effects do not occur where there is no exposure • Effects occur only downstream of a discrete source • Effects occur where there is exposure from dispersed sources, but not at carefully matched reference sites
Spatial gradient	<ul style="list-style-type: none"> • Effects decline as exposure declines over space
Temporal relationship	<ul style="list-style-type: none"> • Exposure precedes effects in time • Effects are occurring simultaneously with exposure (allowing for response and recovery rates) • Intermittent sources are associated with intermittent exposure and effects
Temporal gradient	<ul style="list-style-type: none"> • Effects increase or decline as exposure increases or declines over time

difficult (see above). Similarly, ecotoxicological test methodologies put emphasis on metrics such as survival, growth and reproduction since these changes in individual fitness are considered to be ‘ecologically relevant’ and to influence directly the status of the population. However, translation of phenotypic variations in life history traits of individuals into demographic changes in population depends greatly on the life history strategy of a particular species [71, 72]. A 50 % loss of fertility may have completely different demographic consequences for two species with contrasting life histories, e.g. an opportunistic and a periodic species [73]. Such non-linear effect propagations from the initial adverse effect of a stressor to the ecological status complicate cause–effect relationship inferences, in particular as our understanding of how processes and mechanisms at different biological levels relate to each other is not well developed [74].

Besides direct and indirect gradient analysis and multivariate ordination techniques, which have been summarised by Ter Braak [75] and which have been discussed above, three additional quantitative methods are available for the diagnosis of ecosystem effects: (1) the application of predictive and of general linear models, (2) the application of a GIS-based WoE approach and (3) the application of self-organising maps and of artificial neural network approaches. These methods will be further discussed in the following sections.

2.4.1 Application of Predictive Models and General Linear Models

The complex statistical method presented by De Zwart et al. [42] describes a general eco-epidemiological diagnostic framework for linking measures of ecological impairment with likely causes [76]. The approach uses predictive models (species sensitivity distributions (SSD) and RIVPACS) and ‘general linear models’ (GLM) in statistical regressions to quantify and attribute potential effects of mixture toxicity and habitat alteration. The end point (risk receptor) is the local species composition. The specific method combines ecological, ecotoxicological and exposure modelling to provide statistical estimates of the effects of different natural and anthropogenic

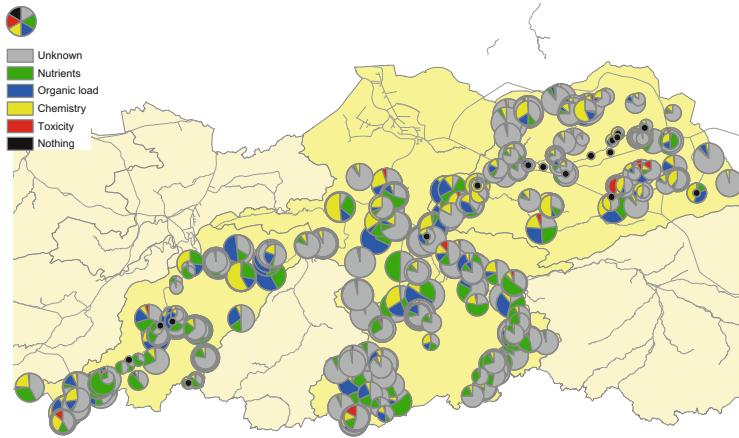


Fig. 2 Example output of the ‘effect and probable cause’ (EPC) pie charts method, showing the map of the Belgian part of the Scheldt river basin. The figure is only used to give an idea about how the EPC output is graphically presented. The output is not further discussed here

stressors on stream fish assemblages. However, the method proposed can also be applied to other taxonomic groups (e.g. macro-invertebrates, algae, macrophytes, etc.). The outcome of the analysis—of which an example for macro-invertebrates is given in Fig. 2—can be simplified using ‘effect and probable cause’ pie charts (EPC). These can subsequently be GIS-mapped to aid in communication. The size of the EPC-pie is proportional to the magnitude of the effect. The size of each pie slice is proportional to the relative importance of each probable cause. This ‘pies and slices’ method requires excellent quality data for a range of parameters discussed above. In summary, the data have ideally been collected in concert (same site, same location, same time) and for a large variety of sites, with independent gradients in values for all parameters that are examined (e.g. clean to polluted, pristine habitat to degraded habitat, different stream orders, etc.). The EPC method also requires the inclusion of reference sites that are minimally disturbed by any of the potential stress factors.

2.4.2 Application of a GIS-Based Weight-of-Evidence Approach

The ‘Geographic Information System’ (GIS)-based weight-of-evidence (WoE) and ‘Weighted Logistic Regression’ (WLR) techniques presented in Kapo and Burton [77] are multidimensional applications of an epidemiologically based approach that defines spatial relationships between measurement end points (risk receptors) and multiple environmental stressors. The quantitative WoE method is based on Bayes’ rule and was first conceptualised in the context of epidemiological research.

Thereafter it has been employed in biological assessment research using various design strategies [78]. The GIS-based WoE/WLR method used within the MODELKEY project was developed for and is currently used in mineral exploration [79]. In river basin assessment applications, WoE/WLR analyses are performed to establish spatial associations between multiple predictor variables and the discrete biological conditions ('impaired' and 'non-impaired') to which these variables potentially relate. It is important to note that the identification of a spatial association between a specific stressor and a biological response does not directly imply causality. Instead it may be considered a 'suspect' relationship to be further explored and refined by a variety of procedures in the later steps of a biological assessment (see [2], this volume for information on investigative monitoring programmes). Those steps also include the determining of a relative ranking of both potential stressors and affected locations within a study area [80, 81]. The use of WoE in combination with WLR allows identification and inclusion of all the predictor variables that show significant spatial association with the biological condition (i.e. risk receptor) of interest. This is especially important when the role of stressor mixtures in the environment should be assessed, as the unique conditions produced and used in WLR enable the identification of specific mixtures of stressors and provide evidence on potential interactions among those stressors. The output of the WoE/WLR analysis can be graphically visualised as colour-coded maps that include the list of potential stressors with positive association to impairment and their relative ranking. With relative ranking it is meant that the WLR coefficients of the significant dominant stressors allow for a ranking of stressors based on their importance as probable cause for biological impairment, thus delineating the most influential stressors within each model study area (see Fig. 3).

2.4.3 Application of Self-Organising Maps and Artificial Neural Network Approaches

Aquatic communities are rapidly disturbed by modifications in the physical or chemical environment ([83], and see [2], this volume). However, a major difficulty is the detection of the response of these communities to specific environmental gradients [84]. One of the main barriers to detecting the effects of specific stressors on aquatic communities is the difficulty of sampling of ecologically comparable habitats, both exposed and not exposed to anthropogenic stressors [85], as has been described above. The use of advanced statistical methods in conjunction with large databases can circumvent such problems, as an extensive range of values are available for each variable. In addition, the relationships between communities and their environments are often characterised by complex, non-linear and difficult to model responses. 'Artificial Neural Networks' (ANNs) have been widely applied for the interpretation of complex and non-linear phenomena in computing and electronic engineering [86–89]. In ecology-related fields, ANNs have been implemented in data organisation and classification of groups [90–93], patterning complex relationships between variables [94] and predicting population and

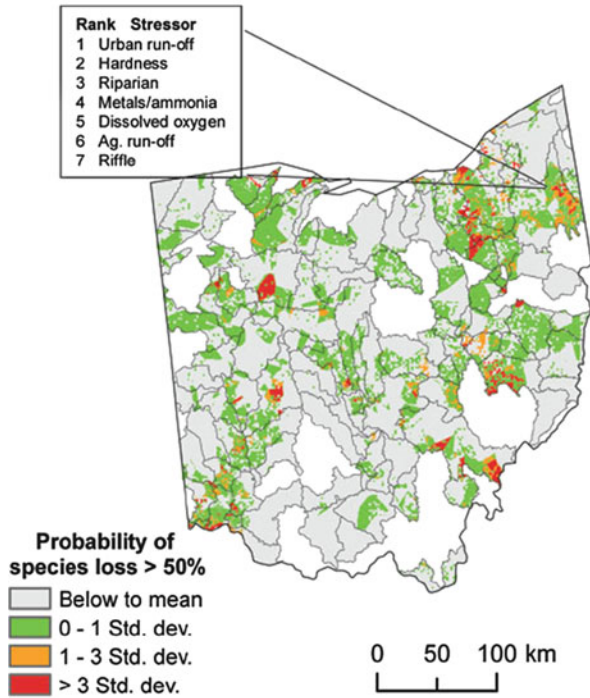


Fig. 3 Example output of the GIS-based WoE/WLR model, showing a site on the Mahoning River in Ohio, USA [82]. The output is not further discussed here

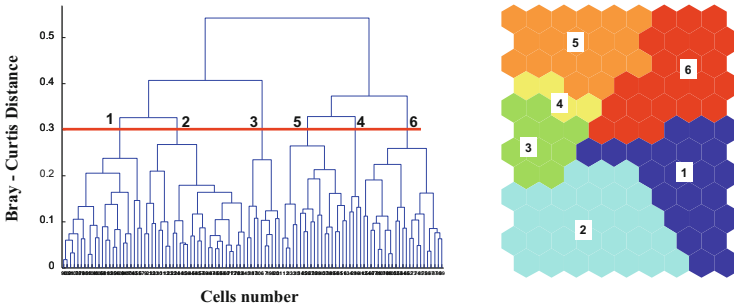


Fig. 4 Example output of a ‘hierarchical clustering analysis’ of macro-invertebrates in the Scheldt river basin showing a dendrogram (left) and six clusters on a ‘self-organising map’ of 108 cells (right) (after [98]). The output is not further discussed here

community development and structure [95–97]. However, except for the studies conducted within the MODELKEY project [98], little research has been conducted on the implementation of ANNs and ‘self-organising maps’ (SOM) in ecotoxicological studies. Fig. 4 shows a dendrogram resulting from hierarchical clustering analysis carried out on the virtual vectors of each cell in the SOM map and the

representation of the six clusters on the SOM map with 108 cells for the macro-invertebrate data in the Belgian Scheldt River. The approach allows the classification of sites within clusters characterised by multiple stressors and, hence, enables the identification of likely causes of impairment.

3 Decision Support for Risk Assessment and Management

3.1 Decision Support Systems

Many problems of water quality and quantity management currently affect fresh waters and coastal waters in Europe [82], thus posing different challenges to decision- and policymakers ([83], and see chapters by Brils et al. [6], Négrel et al. [40] and von der Ohe et al. [2], this volume). Decision support system (DSS) may be of great help to them. In general, a DSS can be defined as a computer-based tool used to support complex decision-making and problem solving [99]. The range of “systems” that might be called DSS is wide. The least complex ones are information retrieval systems that provide access to information, while the most complex ones are frameworks that allow any decision to be modelled, using a formal decision analysis approach. A DSS can be designed as follows: (1) very specific to a particular decision or component of a particular decision (e.g. a nutrient loading model built for a specific river basin), (2) a framework that allows a particular type of application to be modelled (e.g. river basin management, sustainable land reuse) or (3) a generic framework for modelling any type of decision (e.g. Analytica,³ GoldSim⁴).

DSSs can help users to understand their problems better, to address issues that they previously ignored and to provide a robust and repeatable structure for the decision-making process [100]. They also allow the integration of different types of information. They can include integrative methodologies, such as cost-benefit analyses, that evaluate site management alternatives. DSSs can also provide powerful functionalities for analysis, visualisation, simulation and information storage that are essential to complex decision processes. Information and different simulated scenarios can be presented in an ordered structure that can be more easily discussed among the involved parties. Moreover, they can facilitate one of the most important aspects of environmental management, which is communication (see also book Section C). The use of a DSS allows parties to openly discuss potential decisions and their implications under different scenarios and assumptions of risks, benefits and costs. This fosters a dialogue and consensus-building processes among decision-makers, experts and stakeholders. It may provide greater transparency in the decision-making process for communication with the wider public and authorities.

³ See: <http://www.lumina.com>

⁴ See: <http://www.goldsim.com>

Different DSSs for the assessment and management of fresh waters and coastal waters are currently available or under development. Agostini et al. [101] listed some examples of existing DSS related to river basin management and WFD implementation:

- The 'Elbe River DSS',⁵ developed within the study 'Towards a generic tool for river basin management' supported by the German Bundesanstalt für Gewässerkunde (BfG), addresses different river problems, such as the improvement of socio-economic use of the river basin and the definition of sustainable level of flood protection.
- The MODELKEY DSS,⁶ developed under the European Commission-funded MODELKEY project (2005–2010), interlinks and integrates different analytical tools and exposure/effect models in order to evaluate risks posed by pollution to aquatic ecosystems at river basin scale and to identify areas (hot spots) in need of management.
- The MULINO DSS⁷ (MULTi-sectoral, Integrated and Operational DSS for sustainable use of water resources at the catchment scale) developed within the European Commission-funded project with the same name, became an operational tool which meets the needs of European water management authorities and which facilitates WFD implementation.
- The RiverLife DSS,⁸ developed within a project of Finnish institutes, is an interactive computer-based DSS, used via the Internet, which helps to integrate environmental considerations into land use planning and management practices in river basins.
- The TRANSCAT DSS, developed within the European Commission-funded project with the same name, is used for integrated water management of trans-boundary catchments.

The available DSSs range from simple to very complex software systems and from generic systems to systems targeted to a specific problem or location. They often include various tools, such as geographic information systems (GIS) and multi-criteria decision analysis (MCDA) [100]. Water managers and decision-makers are therefore confronted with a wide choice of adoptable systems and should apply some criteria to select the most suitable one for their specific environmental problem. For example, they have to decide how simple the system should be, if the implemented tools should be technically elementary and straightforward, if the included models—although accomplishing their analytical tasks—should be equally understandable and not perceived as black boxes and if their use should be

⁵ See: <http://www.riks.nl/projects/Elbe-DSS>

⁶ See: <http://www.modelkey.ufz.de/index.php?en=3150#board>

⁷ See: <http://siti.feem.it/mulino/>

⁸ See: <http://www.environment.fi/default.asp?contentid=141642&lan=EN>

time and cost effective [83]. In the case of contaminated waters, further development and improvements are needed to bridge the gap between DSS developers and end users, in order to make the contaminated water DSSs routinely used by decision-makers [100].

3.2 End User Involvement in DSS Development

Collaboration with end users in designing a DSS is extremely important in order to effectively fulfil the needs and expectations of decision-makers and to ensure DSS application and updating over time. Decision-makers are usually reluctant to use DSSs for several reasons:

1. They perceive DSSs to be black boxes, so they do not trust them and prefer to make decisions without the use of DSSs.
2. They do not want to spend time and effort on a tool that they perceive to make their problem more complex.
3. DSSs are often developed without the close involvement of the end user community and so may not be ‘fit for purpose’.

These reasons might explain why a lot of tools have been developed but are rarely applied at a European level, and it confirms the need for the closer involvement of end users in DSSs development.

Within the MODELKEY project, the involvement of end users in the design, testing and application phases of the DSS development was secured from the beginning. Meetings were organised to discuss the conceptual framework of the MODELKEY DSS and to capture (through questionnaires) end users perspectives and needs related to the WFD implementation. Moreover, end users from the Elbe, Scheldt and Llobregat river basin, i.e. the three case studies in MODELKEY, were invited to test on-line the preliminary and the final prototype version of the DSS and to provide the developers with suggestions for improving both methodological and technical features. Based on these suggestions, the MODELKEY DSS was made ready to support water managers involved in the 1st and/or 2nd update of the WFD River Basin Management Plans (see Fig. 3 in [6], this volume).

3.3 The MODELKEY DSS: Methodological Approach and Technical Solutions

This paragraph focuses on the MODELKEY DSS [102] as an example of how different data, models and tools have been integrated into a conceptual framework for river basin assessment and management.

One of the most critical issues regarding the implementation of the WFD is the ecological status (ES) classification. A list of parameters (or metrics), indicative of each biological, physico-chemical, chemical and hydromorphological quality

elements (QE), is recommended for monitoring, and a general framework for classification of ES is provided ([103] and see [2], this volume). To this end, a variety of environmental quality indicators and metrics have been developed and applied in national monitoring programmes in order to be in compliance with WFD requirements [104, 105]. As far as the water body status classification is concerned, the 'one-out, all-out' principle is applied at QE level, so that the ES is initially determined by the lowest class corresponding to the most impaired biological community. If monitoring results for physico-chemical, chemical or hydromorphological QE highlight a worse situation than for biological QE, the ES will be downgraded to the lower class. However, this 'one-out, all-out' principle is now regarded to be 'too conservative', especially for prioritisation purposes [106]. In order to achieve a more comprehensive and realistic ES classification, the importance of integrating available heterogeneous information by means of a reliable and flexible method has been recognised by the MODELKEY DSS developers. The solution adopted in MODELKEY relies on the weight-of-evidence (WoE) approach [107] referring to the site-specific Ecological Risk Assessment [38, 108]. As used in other environmental studies, WoE can be defined here as the determination of possible environmental impacts based on multiple lines of evidence (LoEs).

The basic principle of WoE approaches, when conducting an ecological risk assessment, is that if a majority of the assessment results suggest impairment, there is a greater likelihood that the ecosystem is actually impaired. Correspondingly, if most of the assessment results suggest no impairment, there is most likely none [107]. This means that the analysis of a single LoE is in most cases insufficient to achieve credible evaluation of ecosystem impairments and several lines of evidence might be needed to adequately assess the exposure to and the effects of a stressor.

In the MODELKEY DSS, a WoE approach implementing a dedicated 'Fuzzy Inference System' [109] has been developed to aggregate different types of environmental indicators for classification purposes. The so-called integrated risk assessment methodology uses algorithms combining two operators, i.e. the minimum (i.e. 'one-out, all-out') and the arithmetic average, and applies it to indicators grouped into five LoEs, namely, biology, physico-chemistry, chemistry, ecotoxicology and hydromorphology. All available biological indicators are hierarchically aggregated through a set of expert rules and checked with responses provided by other supportive QE, resulting in a modified version of the classification framework [110], as provided by the European Commission [103]. The proposed WoE methodology allows for the identification of the most likely relevant causes of impairment (key stressors and key toxicants) and of the most impaired biological communities (key ecological end points). This outcome enables to suggest additional (i.e. investigative) monitoring programmes (see [2], this volume) or to suggest the implementation of appropriate measures.

Another critical issue in the WFD implementation context is the integration of environmental and socio-economic information on river basins for management purposes. The directive requires a consideration not only of pressures, impacts and status of waters but also of water uses and the social and economic conditions of

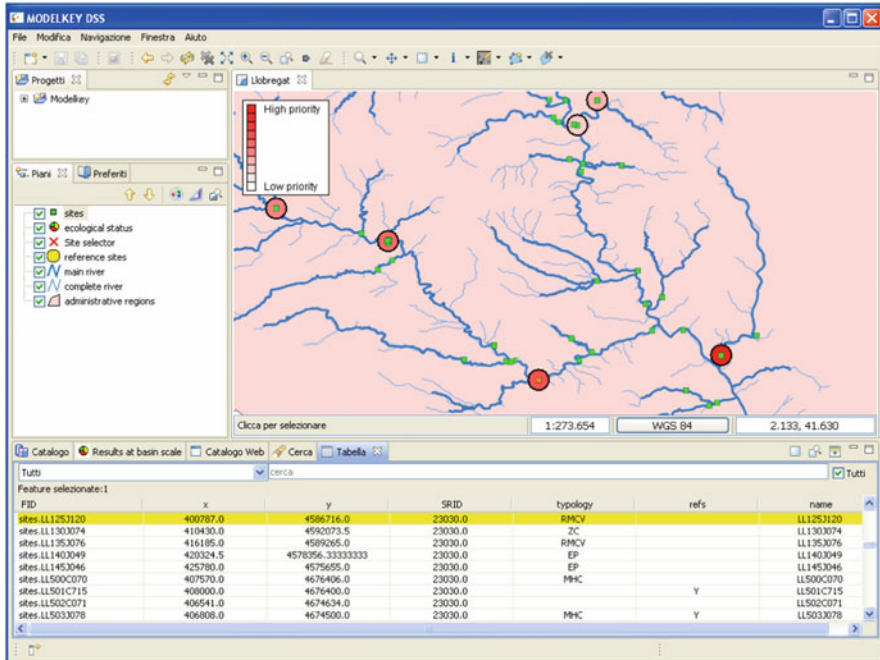


Fig. 5 Example output of the MODELKEY DSS interface, showing the priority of sites for remediation

regions crossed by the river. To this end, a ‘Multi-Criteria Decision Analysis’ (MCDA) procedure [111] has been developed to prioritise hot spots, where management decisions should take into account both environmental and socio-economic perspectives. The procedure is based on three main criteria. The first criterion considers water quality scores in relation to the physico-chemical pressures on the river basin (i.e. organic pollution, eutrophication, acidification, toxic pressure and hydromorphological pressure). The second criterion evaluates the vulnerability of the considered water uses (i.e. industrial, agricultural, energy production, residential, recreational, etc.) to the different pressures. The third criterion is represented by the socio-economic importance of the water uses in the region of concern. The MCDA procedure allows the decision-makers and stakeholders to evaluate different preferences in the prioritisation algorithms, thus facilitating communication in the decision-making process.

In order to extend its applicability, the DSS software system is developed on the user-friendly and open source Desktop Internet GIS (uDig) platform.⁹ It is free of charge, easy to use and offers multi-language facilities, thus allowing the translation of the DSS interfaces into Member States national languages. Moreover, it

⁹ See: <http://udig.refrains.net/>

offers coherence with the language and the reporting requirements of the WFD. Another important feature of the DSS is its 'open configuration', enabling the integration of different assessment tools into a single structure (i.e. indicators, models and databases). The DSS not only currently embeds tools developed by project partners, such as databases collecting substance properties and data from river basin monitoring programmes and queries for the calculation of indicators (e.g. SPEAR index), as well as exposure and effect models (e.g. species sensitivity distributions), but also allows to include external models or assessment results into the decision process. The embedded tools can be applied separately for specific purposes, or chains of tools can be created and used to define and evaluate different assessment scenarios.

By implementing these risk-informed methodologies into a software system (see Fig. 5), the MODELKEY DSS guides water managers in the identification of areas of major concern, taking decisions on intervention alternatives and assuring decision-makers' and stakeholders' involvement and participation, as well as in communicating results in a transparent and simple way.

4 Conclusions and Recommendations

In spite of the fact that European rivers, lakes and estuaries have become less polluted since the 1980s, many water bodies still fail to meet the WFD goal of achieving a good ecological and chemical status. While biological assessments are important tools for determining ecological deterioration, models or statistical evaluations are needed that link observed biological effects to likely causative factors. Since many different types or classes of stressors may cause the same type of ecological responses, stressor identification should be based on as many potentially disturbing factors as can be quantified. Moreover, to allow for comparable results, monitoring protocols should be harmonised across Europe, so that the same assessment tools can be applied.

The identification of the likely causes for ecological deterioration will be much easier if it can build on available, reliable, extensive and comprehensive biogeochemical monitoring data (preferably aggregated in a database). This chapter presented a few quantitative diagnostic approaches for such identification, i.e. approaches for estimating the magnitude of impairment as well as the likely relative importance of different stressors in a multiple stressed environment. The presented approaches differ considerably in their type of analysis, applied statistics and outputs. However, applied on the same datasets, they may indicate similar stress factors as the perceived major causes for the observed ecological impacts [82, 112]. This will increase the confidence in the results and thus further underpins the implementation of respective management measures.

Multiple stressor assessment may revert to toxic effect models that have been established for chemical mixtures (see Sect. 2.1 in [2], this volume), such as the model of response addition that demonstrated in a recent study the joint effects of predation threat, parasitism and pesticide exposure on *Daphnia magna* [113]. However, so far

there is no rigorously tested and widely accepted model for the interaction of chemical and non-chemical stressors, which may occur at the same time or sequentially, which may affect different life stages or cause indirect effects on competitors, predators or parasites. Thus, to provide a better mechanistic understanding of multiple stressors, and to integrate this understanding in applicable tools for monitoring and assessment, is one of the great challenges for environmental science today.

Only through better diagnosis it will be possible to identify the causes of river ecosystem impairment. A plausible identification of such causes is a prerequisite for well-informed decisions on which mitigation or remediation measures to take. DSSs are promising tools in these efforts as they support integrative analysis, can help to visualise the results in the context of a river basin and can test alternatives in the context of socio-economic uses. Socio-economics are often disregarded in scientific models but need to be considered by water managers, who must balance a range of issues to make sustainable decisions.

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Section B
Anticipating Change

Downscaling Scenarios as an Exploratory Tool for River Basin Management: An Introduction

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Abstract Global change in river basins can be anticipated by using downscaled scenarios. This approach is introduced in this chapter, and it is argued that scenarios are a well-developed and suitable tool to explore the uncertain bandwidth of the future states of river basins, their rivers and occupant societies. Quantitative and qualitative scenario exercises are introduced and a worked-out example is presented for the Scheldt river basin.

Keywords Land use change • Narratives • River basin management • Scenario downscaling • Water resources management

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1 Introduction

Changes in climate, land use and the future political and economical configuration of society will affect risk source-pathway-receptor linkages (see Fig. 2 in Brils et al. [1], this volume, and see Section A in this book). This may have profound consequences for both the rivers that are to be managed and the institutions that carry out this management. At the same time, we are uncertain of the direction and magnitude of these changes. This inherent uncertainty is a critical element that has to be addressed in any planning and management approach targeting well ahead of the coming years. Compounded in this uncertainty are known and unknown risks, such as the possibility of highly infrequent but disastrous floods or droughts (see [2], this volume), the volatility of future markets but also the current costs and future benefits that societies are willing to bear and expect to earn as a consequence of currently taken decisions. Planners have used blueprints of future society [3, 4] and accepted no-regret options and flooding thresholds. However, all of these projections, predictions or anticipations generally foresaw a single and often inarticulate trajectory for societal change that can be grasped as ‘more of the same’ or ‘business as usual’ [3, 4]. At the same time, they do not consider the possibility of major deviations in moral choices, rearrangement of institutional constellations in society at large or a role for major technological breakthroughs. In short, projections are generally linear and augmentative and do not foresee societal restructuring and trend reorientation. Scenarios, in contrast, have been designed to meet this inflexibility and cover some of the divergence linked to drastic change. This introduction argues that the application of scenarios, whether quantitative or qualitative, is useful as a means to get a grip on some of the inherent uncertainty tied to future outlooks.

The European Environment Agency (EEA) made a catalogue [5] of all sorts of forward-looking indicators and placed scenarios somewhere between facts, predictions and projections, on the one hand, and explorations and speculations, on the other: all along a gradient of increasing complexity and uncertainty. Since their early use in public policy debates and private business in the 1960s and 1970s [3, 4], scenarios have become a well-established tool to explore how the world would look somewhere in the distant or near future. The EEA [5] suggests that environmental policy has become increasingly complicated since the 1970s with an increased notion of uncertainty and of possibly highly dynamic change over unknown spatial scales. This has triggered an increasing demand for scenario-based assessments. A scenario can be seen as a coherent, internally consistent and plausible description of a possible future state of the world [4–8]. Scenario descriptions are often qualitative and broad-brush ‘narratives’, contrasting but broad, over-our-head trajectories of world development and should be distinguished from smaller-scale management options that are within grasp of the manager or policy maker and can be implemented comparatively easily.

Scenarios can be used as input for numerical models (see Section A of this book), but also for rational, deductive thought exercises without interference of

formal models (e.g. [8]). Scenarios are used to tentatively explore into an uncertain future, as exemplified by national economic planning exercises (e.g. for the Netherlands: [9–12] or the Foresight exercise in the UK: [13]; the ‘Intergovernmental Panel on Climate Change’ (IPCC) assessment of the possible consequent avenues of the interactive effects of climate and world economy change [14–17] and the ‘Millennium Ecosystem Assessment’ [18]).

From a modeller’s perspective, scenarios just form a sensible array of input conditions that make the model produce its output. The formulation of this sensible array is dictated by the demands and needs of the modeller’s client, the study objectives and extent, the time horizon and the broader institutional and disciplinary setting of the work. Together, models and scenarios allow answers to ‘what-if’ questions within a broad but plausible bandwidth by spanning the width of all possible outcomes. Greeuw et al. [19] offer a useful review and analysis of different sets of scenarios that have been developed since their early applications.

The application of scenarios has taken flight around the turn of the millennium, mainly because the IPPC [16, 20] and the Millennium Ecosystem Assessment [18] have made extensive use of a set of four rather similar scenarios. Particularly the four IPCC Special Report on Emission Scenarios (SRES-scenarios) have been a successful, well-cited ([20, 21]) attempt to describe strongly contrasting potential directions of world development and have attracted some convergence among scenario users. These SRES-scenarios depict possible future trajectories as spanned by two dimensions of global societal change, the first contrasting globalisation versus regional differentiation and the second contrasting a focus on economic growth and expansion versus one of sustainable resource use (Table 1).

Although the SRES-scenarios working group decided to use the neutral labels A1, A2, B1 and B2 [21], quite notably, different users have felt the need to attach qualifying labels to similar sets of four scenarios, ranging from charismatic animal names (Sea Eagle, Beaver, Dolphin, Lynx; [10]) to imaginative sentences (‘pull up the drawbridge!’ for a scenario similar to A2 and ‘we got the whole world in our hands’ for B1; see [22]). Busch [23] compared a range of similar orthogonal or two-way scenario articulations for Europe (among others [13, 15, 24]) and concluded that similarity in the articulation of the narrative was often striking. At the same time, large differences existed among scenario studies in future land use depending on the assumptions about global trade, increase in agricultural productivity and biofuel production. Langmead et al. [22] argue that the two-way scenario work-out would be enriched by the inclusion of a fifth ‘business-as-usual’ extrapolation, which would then be seen as a trajectory anywhere between the other four (A1, A2, B1, B2).

The European Commission (EC) funded project PSI-connect¹ came to some hands-on guidance on how scenarios can be developed as well as explored by water managers in a participatory approach. Furthermore, new scenario-type future visioning has also been developed in the EC project RESPONSES.²

¹ See <http://www.psiconnect.eu>

² See <http://www.responsesproject.eu>

Table 1 A verbal characterisation of the four socio-economic scenarios of the Special Report on Emission Scenarios (A1, A2, B1, B2) adopted from Lorenzoni et al. [15]. The labels applied by the EURURALIS project^a [22] have been added for comparative reasons

Scenario—label	Narrative
A1—World Markets; GE—Global Economy	The A1 storyline and scenario family describes a future world of very rapid economic growth, global population that peaks in mid-century and declines thereafter and the rapid introduction of new and more efficient technologies. Major underlying themes are convergence among regions, capacity building and increased cultural and social interactions, with a substantial reduction in regional differences in per capita income. The A1 scenario family develops into three groups that describe alternative directions of technological change in the energy system
A2—Provincial Enterprise; CM—Continental Markets	The A2 storyline and scenario family describes a very heterogeneous world. The underlying theme is self-reliance and preservation of local identities. Fertility patterns across regions converge very slowly, which results in continuously increasing population. Economic development is primarily regionally oriented and per capita economic growth and technological change more fragmented and slower than other storylines
B1—Global Sustainability; GC—Global Cooperation	The B1 storyline and scenario family describes a convergent world with the same global population that peaks in mid-century and declines thereafter, as in the A1 storyline, but with rapid change in economic structures towards a service and information economy, with reductions in material intensity and the introduction of clean and resource-efficient technologies. The emphasis is on global solutions to economic, social and environmental sustainability, including improved equity, but without additional climate initiatives
B2—Local Stewardship; RC—Regional Communities	The B2 storyline and scenario family describes a world in which the emphasis is on local solutions to economic, social and environmental sustainability. It is a world with continuously increasing global population, at a rate lower than A2, intermediate levels of economic development and less rapid and more diverse technological change than in the B1 and A1 storylines. While the scenario is also oriented towards environmental protection and social equity, it focuses on local and regional levels

^aSee <http://www.eururalis.eu/>

This introduction to ‘Section B—Anticipating Change’ provides arguments for the use of the SRES-scenarios. It is argued that this set of common, well-developed and frequently used scenarios have proven their use in water resources management and socio-economic modelling and thus can serve as a benchmark. Section B then continues with two quantitative and a qualitative scenario analyses. Te Linde et al. ([25], this volume) show how the socio-economical and hydrological components of future flood risk interact in river basins and address the approaches towards

pinpointing the uncertainty in both. Hellmann and De Moel ([26], this volume) survey foreseen land use change as a function of SRES-scenarios in a range of European river basins. Land use change, both in the upper and lower stretches of a basin, can be a critical driver of change in both elements of the risk equation, probability and effect. The modelling effort by Hellmann and De Moel suggests an increased urbanisation across scenarios, but otherwise limited changes in land use towards 2030. However, still substantial impacts on sediment delivery to rivers are anticipated. The chapter by Vermaat, et al. [27], this volume, reports on the outcome of a qualitative Dahlem-type expert workshop on scenario articulation for European river basins. Together, these findings would assist future river management and planning in allowing a broad-brush reflection on the consequences of unexpected, but possibly major societal changes.

2 The SRES-Scenarios Have Become Popular

SRES-scenarios can be encountered in a growing range of applications, both as highly specific inputs for models of variable complexity and typology as well as broad-brush narrative starting points for qualitative sketches. An example of the latter is given by [8] and [24]. Examples of the former can be found in [23, 24, 28–34]. All the latter involve some sort of region-specific and issue-specific downscaling of the broader SRES-scenarios to provide tailored and articulated bandwidths of model inputs. These regionalised efforts also provide much of the necessary refinements suggested by Arnell et al. [17].

Notably elaborate modelling tools that incorporate the SRES-scenarios are DIVA (Dynamic Interactive Vulnerability Assessment: a global coastal database and dynamic model for climate change vulnerability assessment; see, e.g. [33]³); the ‘Land Use Scanner’ (a dynamic GIS combining land use pricing and an economic equilibrium model for the Netherlands; see [28, 34]⁴) and ‘EURURALIS’ (logistic land use modelling for Europe coupled with an economic equilibrium model; see [31]). EURURALIS has been applied by Hellmann and De Moel ([26], this volume) to quantitatively estimate land use change across a wide range of European river basins. The four SRES-scenarios do not necessarily lead to four divergent outcomes. The overall range is often spanned by two scenarios, although not always by the same two, depending on the issue of interest (e.g. [30, 34, 35]; Fig. 1).

Regionalised climate change projections have been equated, with some caution, to the SRES-scenarios (e.g. for the Netherlands by Van den Hurk et al. [36]; Table 2). Thus it is possible to associate regionalised climate change patterns of temperature and precipitation with socio-economic scenario trajectories.

³ See also <http://ebmtoolsdatabase.org/tool/diva-dynamic-interactive-vulnerability-assessment>

⁴ See also <http://www.lumos.info/landusescanner.php>

Fig. 1 Changes in land use in Europe over the period 1990–2030 estimated with EURURALIS for the four SRES-scenarios (Table 1): increases in new nature versus areas subject to expanding urbanisation (data from [30])

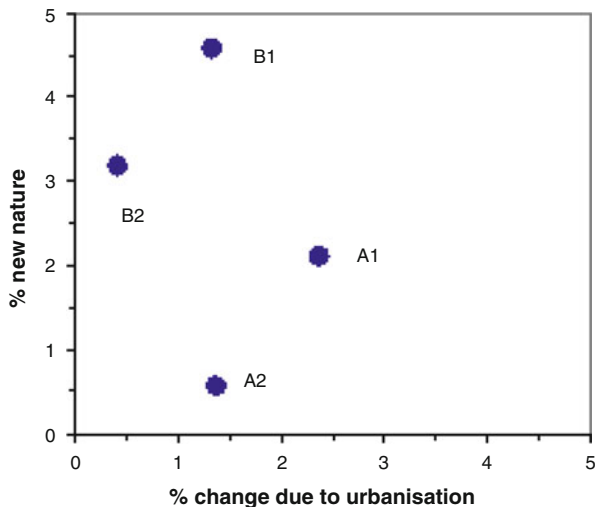


Table 2 Correspondence of SRES-scenarios (Table 1) and KNMI meteorological climate scenarios for the Netherlands (Adopted from [36])

SRES-scenario	Projected temperature rise in 2050 compared to 1990 (°C)	Corresponding KNMI climate scenario ^a
A1	1.1–1.8	W, W+
A2	1.2–2.0	W, W+
B1	0.8–1.4	G, G+
B2	1.0–1.8	G, G+

^aKNMI scenarios: W = warmer, that is a stronger increase in temperature by 2050 (+2 °C); G = moderately increased temperature (+1 °C); the affix ‘+’ suggests a much stronger air circulation involving warmer, wetter winters and warmer, dryer summers

Van den Hurk et al. [37] highlight that they have not derived their climate change scenarios from those of IPCC SRES. Hence, the difference in circulation strength that discriminates two of these four KNMI (Royal Dutch Meteorological Institute) climate scenarios cannot be matched to an SRES-scenarios (Table 1). Still, some of the consequences of socio-economic development trajectories that will work through to measurable changes in aspects of climate—such as those for land use—will be traceable and hence can be deduced. We presume that the cautious matching between SRES and KNMI scenarios of [36] for the region encompassing the Netherlands is sufficiently robust to apply in the North-West lowlands of Europe. This ranges roughly from the Seine upwards to the Elbe, because multi-annual discharge patterns among these adjacent rivers correlate well [37].

3 Downscaling SRES-Scenarios for the Scheldt River Basin

The SRES-scenarios were downscaled for the Scheldt river basin as an input to the development of an integrated modelling suite including land use, groundwater and surface water flows and nitrogen dynamics within the framework of a European cooperative research project, SPICOSA.⁵ Comparable integrating work has been carried out for the Seine [38], Rhine, Elbe and Po [39–43]. The Scheldt will serve as an illustrative example.

Downscaling of scenarios may detail aspects of the distribution of wealth, the intensity of agriculture, types and distribution of recreation, the planning and regulation of urban sprawl and the adopted lifestyles by the population at large, including health and demographic aspects as well as governance styles. In the Scheldt case, the focus was on agricultural land use and other sources of the plant nutrient nitrogen (N) as it moves through the river basin via river and estuary to the sea. The time horizon was 2030. We have limited ourselves to societal aspects that may affect the intensity of land use and N cycling. We base ourselves on earlier articulations [9–12, 22, 29–32]. All is brought together in Table 3. Despite heterogeneity in culture, governance and occupation density of the Scheldt basin, we have taken Belgium to be representative and made it lead our deliberations on societal aspects.

Clearly, these downscaling deductions for the four SRES-scenarios lead to contrasts in (1) land use and water management as a consequence of postulated differences in the worldwide economic situation as well as (2) contrasts in policy trends and governance styles in Europe and this specific river basin. For example, A1 and B1 are thought to lead to a reduced demand for land for agriculture compared to A2 and B2, although for different reasons.

Also, marked differences in governance style will be reflected in the way society deals with river corridors. These could be dealt with either as mainly utilitarian, transport ways of water and goods or as multifunctional elements of a more sustainably planned and used landscape. Articulations similar to Table 3 can be used to formalise scenarios in terms of hydrology, land allocation, agricultural markets, land use intensity and demography as input tables and decision rules for quantitative modelling efforts. Vermaat et al. [44] report on this modelling effort and conclude that over the 30-year modelled time period until 2038, climate scenarios (G versus W+, Table 2) have considerably less impact on nitrogen load of the Scheldt than measures that can be taken at the level of farming nutrient use systems, land use and domestic sewage treatment. Most cost-effective measures for complying to the Water Framework Directive quality standard for nitrogen appear to be (1) enhancing nutrient retention in buffer strips, that form a buffer between agricultural land and the river and (2) improving domestic sewage treatment. Our scenarios differ greatly in the probability that these different packages of measures are being included in river basin management plans.

⁵ See <http://www.spicosa.org>

Table 3 Articulation of SRES-scenarios (Table 1) for land use and water management in the Scheldt basin

Scenario	Agricultural					Type and intensity of recreation	Nature conservation
	World economic influences on local economy	Global and regional demography	Policy and governance styles	Land requirements and intensity	Spatial planning and urban sprawl		
A1	Rapid global economic growth; export subsidies, import tariffs, farm payments and interventions	World population increases to a peak in 2050, to decline afterwards. Total population stable in North-West Europe, age distribution skews to longer life expectancies	Rapid introduction of new, clean and efficient technologies; otherwise small bureaucracies and liberal legislation	Land requirements will decline drastically due to more efficient production technologies in North-West Europe and elsewhere	Agriculture is focused on high-productivity soils and disappears from less-favoured areas; otherwise no restrictive regulation and spatial planning; attractive landscapes suburbanise	Surface water is functional: transport, irrigation and recreation; flood risks are covered rationally by using economically efficient solutions	Existing areas remain protected; land abandoned by agriculture can be allocated to nature conservation, particularly near existing reserves
A2	Slower global economic growth, regionally variable development; export subsidies kept but reduced, import tariffs, farm payments and	World population continues to increase. Age distribution in NW Europe as in A1, but limited immigration	Cultural divergence	Current farming practices remain mainstream, few embrace innovative technologies and ecological agriculture; keen	Agriculture remains in less-favoured areas. No restrictions on urban sprawl. Villages for the rich emerge in attractive areas, whereas	As A1: flood risks are dealt with technically or devolved to downstream	Nature is primarily for recreation; existing protected areas will serve this

B1	<p>intervention prices kept</p> <p>Stable global economic growth; export subsidies, import tariffs and intervention prices are phased out; farm payments decline to 50 % in 2030</p>	<p>from the South</p> <p>As A1</p>	<p>competition from America and elsewhere</p> <p>Land requirements will decline slightly due to lower stress on export agriculture; high-tech ecological farming is widespread throughout the EU</p> <p>Structural EC-wide implementation of environmental regulation; improved equity and social cohesion, clean technologies and zero spillage farming; pro-active GHG policy</p>	<p>urban conglomerates witness a decline in wealth</p> <p>Agriculture in less-favoured areas is intensified and joined with the Natura2000 network as extensive grazing land. Compact cities. Provincial towns will grow as a successful mix of green and social services</p>	<p>The WFD is successful and timely implemented across the EC. Flooding are carefully combated combining economic efficiency and ecological ratio</p>	<p>Demand for mass recreation is dampened by prudent policy, social behaviour and the location of services. Holidays are mainly spent in the safe and familiar EU</p>	<p>Land abandoned by agriculture is added to Natura 2000 network. Biodiversity decline in the EC is turned to a modest increase</p>
B2	<p>Slow and variable global economic growth, the EU does comparatively well; export subsidies are phased out; import tariffs are kept; agricultural environmental farm payments and intervention prices increase</p>	<p>World population continues to increase, though at first slower than in A2. Otherwise as A2</p>	<p>Cultural divergence; limited modernisation</p> <p>Similar to A2, larger portion of ecological farming, notably in combination with regional specificity of products; less competition from America</p>	<p>Agriculture as A2; urban sprawl is restricted and evolution is practised; incentives to sustain or revive smaller, rural villages and towns</p>	<p>As A2, but upstream-downstream conflicts are negotiated among neighbours</p>	<p>No regulation, still limited long-distance mass recreation</p>	<p>Nature conservation only through private NGOs. Nature is for people; hence typical landscapes are conserved</p>

4 Conclusions and Recommendations

It is very likely that considerable changes will occur in the major socio-economic driving forces in our river basins. This will also affect risk source-pathway-receptor linkages. However, we are uncertain of the direction and magnitude of these changes. We can use scenarios to chart several plausible trajectories of change leading to different societies inhabiting landscapes that are not as they look now. Scenarios should be seen as sets of contrasting but internally consistent, plausible descriptions of how the world could look in the distant or near future. For river basin management purposes, the Intergovernmental Panel on Climate Change (IPCC) Special Report on Emissions Scenarios (SRES) may be recommended. These scenarios are well articulated, have developed into a benchmark in the current literature and will remain so for the next generation of scenarios [45].

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Informing River Basin Management on Flood and Drought Risks Taking Future Uncertainties into Account

Aline te Linde, Hans de Moel, and Jeroen Aerts

Abstract Severe dry periods (droughts) receive less attention than floods and storms in natural hazard risk assessments. This is remarkable as droughts (heat waves) cause the most casualties of all natural hazards in Europe. The combination of socio-economic developments and climate change poses a challenge to water managers to not only mitigate flood and drought probabilities but also consider measures that a priori alleviate the damage of extreme floods and droughts. This chapter describes how climate scenarios and socio-economic scenarios can be combined to assess future flood risks. Furthermore, several ways are explained of dealing with uncertainty in an approach that may inform river basin managers on flood and drought risks. Finally, an example of drought risk assessment is provided for the Netherlands. The chapter concludes that cooperation with social and economic sciences is a prerequisite for a strategy that best informs river basin management on flood and drought risks. By applying such a strategy, river basin management could involve information on flood and drought risks to assess urgency and priority of a priori risk reduction measures.

Keywords Drought risk • Flood risk • River basin management • Scenarios

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1 Introduction

1.1 *Economic Losses, Natural Hazards and Uncertainty*

The global economic losses caused by natural disasters have increased in recent decades. After adjustment for inflation and increased prosperity, the average overall damage in the 1950s amounts to about 50 billion Euros (converted to 2007 values). This rose to about 700 billion Euros in the 1990s [1, 2]. The insured losses increased to about 140 billion US\$ on average (Fig. 1).

According to the United Nations, more than two thirds of the world's large cities are vulnerable to rising sea levels and extreme river discharges, exposing millions of people to the risk of extreme floods and storms [3]. In Europe, floods constitute to about 38 % of the total losses of all natural hazards. This makes it, together with windstorms, the most costly natural disaster [4]. In England, for example, floods occurred due to extreme precipitation and caused 8 billion Euro damage in the year 2007 [5]. In 2002, floods in the Elbe basin caused nearly 15 billion Euro damage [6, 7].

Severe dry periods receive less attention than floods and storms in risk assessments of natural hazards. This is remarkable as they cause the highest numbers of casualties of all natural hazards in Europe [4].

Figure 2 displays an increase in large natural catastrophes worldwide since 1950. This includes an increase in the number of hydrological and climate-related events. Climate events include heat wave, freeze, wildland fire and drought. According to the United Nations Environment Programme (UNEP: [8]), water stress exists in many places in Europe, resulting in serious water shortages, flooding, pollution and ecosystem damage. Several studies conclude that in the last decades, the drought situation in Europe got more severe, due to an increase in frequency, duration or intensity of low flows (e.g. [9] and see also [10], this volume). The European Environment Agency (EEA: [4]) estimates that the dry and hot summer of 2003 in Europe caused 10 billion Euros of economic losses to farming, livestock and forestry. The heat wave of 2003 also resulted in considerable loss of life in Europe. Some estimates indicate that more than 52,000 people died, although the heat affected mainly elderly people [11].

Fig. 1 Decadal economic losses due to natural hazards in the period 1950–2005. Plotted are the insured losses versus the estimated overall loss (2005 values; source Munich RE, 2008)

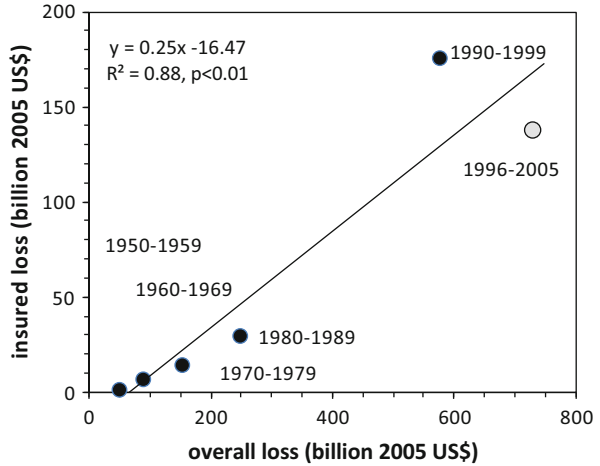
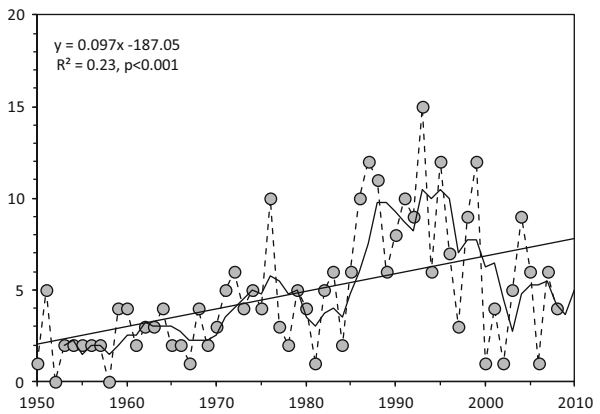


Fig. 2 The number of large (Category 6) natural catastrophes, including a 4-year running mean. Category 6 catastrophes are defined as causing more than 1,000 fatalities, more than 100,000 homeless and/or situations when overall loss and/or insured losses are of exceptional proportions (Adapted from Munich RE, 2008)

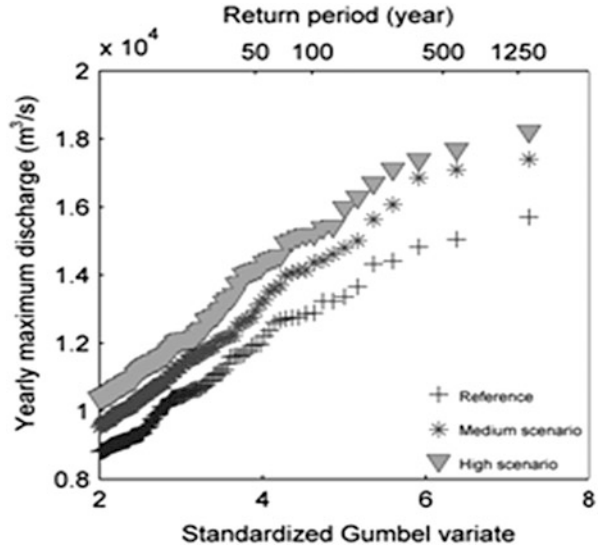


1.2 Frequency of Natural Hazards

The latest report by the Intergovernmental Panel on Climate Change (IPCC) indicates that global warming will intensify the global hydrological cycle ([12], and see Fig. 1 in [13], this volume). As a consequence, the magnitude and frequency of extreme precipitation events are expected to increase. This may lead to an increased flood probability in Europe [14, 15]. Also the frequency and duration of extreme droughts are expected to increase in Europe, in particular in Mediterranean areas [16].

Dankers and Feyen [17] made a pan-European assessment of the changes in flood frequency under changing climate conditions. In several rivers in north-west Europe, the current, flood return period of once every 100 years may in the future be 50 years or even less. Recent findings in the Rhine River basin suggest that—if no

Fig. 3 Extreme value plots of yearly maximum discharges of the Rhine at the German-Dutch border. As input for the rainfall-runoff model served 1,000 years of daily meteorological input data that were obtained by a weather generator (Beersma et al. 2001). Displayed are 1,000 years of simulated yearly maximum discharges for the reference situation and two climate change scenarios for the year 2050 (Source: Te Linde et al. [19])



extra defence measures are implemented—the probability of flooding will increase with a factor of 2.5–5 in 2050 [18] (Fig. 3).

Observed and projected trends of climate impacts, differentiated for four regions in Europe, do show some discrepancy (Table 1). Both observations and projections agree on an increase of river floods in two temperate regions, whereas droughts only increase in the projections of the maritime climate and Mediterranean regions.

Apart from climate change, socio-economic developments play an important role in the vulnerability to extreme flood and drought events. Within the next 30 years, the United Nations predict that the number of people living in cities will increase to 60 % of the world's population, resulting in even more people living in highly exposed areas [20]. Worldwide, water withdrawal intensified in the second half of the twentieth century. This is a direct result from population growth and intensified industry and agriculture [21]. Some scenarios project the world water withdrawal in 2025 to be 1.4 times higher than in 1995. Hence, socio-economic trends further amplify the possible damage of future floods as more people move towards low-lying flood-prone areas and urban areas, and the increase and concentration of people increases water stress.

The combination of socio-economic developments and climate change poses a challenge to water managers to not only mitigate flood and drought probabilities but also consider measures that a priori alleviate the damage of extreme floods and droughts [12]. Examples of a priori measures are early warning, spatial planning, different crop growth and insurance. Furthermore, future projections are highly uncertain and it appears difficult to simulate how flood and drought risks will develop in the future under various combinations of climate change and socio-economic scenarios (conform [22]).

Table 1 Observed (obs) and projected (scen) trends in climate and impacts for northern (Arctic and boreal), temperate (maritime climate, central/eastern) and southern (Mediterranean) regions of Europe (adapted from [4])

Indicator	Northern	Temperate	Southern	
	Arctic and boreal obs/scen	Maritime climate obs/scen	Central/ eastern obs/scen	Mediterranean obs/scen
Atmosphere and climate				
Global and European temperature	+/+	+/+	+/+	+/+
European precipitation	+/+	+/o	o/o	-/-
Temperature extremes in Europe				
Heat waves in Europe	+/+	+/+	+/+	+/+
Precipitation extremes in Europe	+/+	+/+	+/+	+/+
Water quantity, river floods and droughts				
River flow	+/+	o/+	o/+	-/-
River floods (number of events)	o/-	+/+	+/+	o/+
River flow drought	o/-	o/+	o/-	o/+
Economic consequences of climate change				
Direct losses from weather disasters	+/+	+/+	+/+	+/+

+ = increasing; - = decreasing; o = no significant changes (or diverging trends within the region)

The remaining part of this chapter reports on the progress in flood-risk-informed management in Europe, addressing two aspects of future flood management that may deserve additional attention in research and drought-risk assessments: (1) the combined use of climate and socio-economic scenarios in future flood risk assessments and (2) several ways of dealing with uncertainty in a risk-informed approach. Finally, an outlook is provided to the assessment of drought risk, with an example from the Netherlands.

2 Flood Management in Europe

Until recently, flood management in Europe relied strongly on technical engineering capacity [23]. This has a historical basis; water managers have developed the subject area and designed hydro-morphological structures on the basis of flood safety standards [24]. For example, the flood embankments along the river Rhine have been designed to withstand floods that only occur once every 200–500 years in Germany and once every 1,250 years in the Netherlands [19]. While flood defences have provided protection against river flooding, continued socio-economic developments behind these defences resulted in an increase in flood risk. Moreover, not only the amount of economic and social capital increased, but developments

behind flood defences occurred in increasingly more hazardous places because the most safe and obvious places had already been occupied [25].

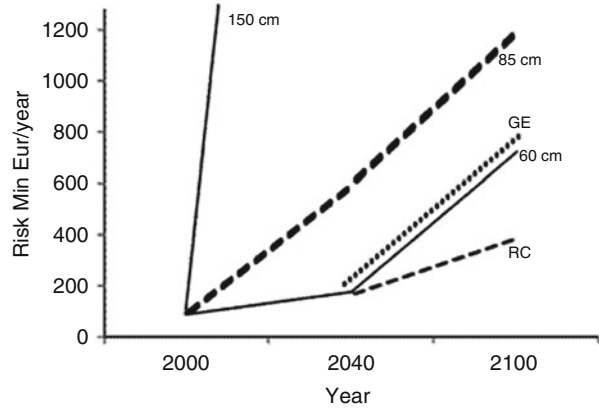
However, since the near floods in 1993 and 1995 along the river Rhine, the 2002 flooding along the Elbe and various floods in the Danube during the last 15 years, flood management has been exploring new approaches that go beyond technical measures such as dikes or canalization. This new way of thinking is referred to as flood-risk-informed management [26, 27]. Flood risk is defined here as the product of probability and consequence (e.g. damaged property and loss of life), i.e. the expected loss per year [28]. The EU supports the transition towards a risk-informed approach in the implementation of the EU Floods Directive (2007/60/EC). This Directive sets out several actions on preliminary flood risk assessment, development of flood risk maps and the preparation of basin-wide risk management plans, to be completed at river basin scale by the end of 2015 (see also [29], this volume). Many European Union member states have already collated information and maps on flood hazard (potentially damaged area), but information and maps on flood risk are still rare [30]. The EU Green Paper on adaptation to climate change [31] also stresses the importance of looking beyond defensive measures. It states that concrete adaptation actions should range from relatively inexpensive soft measures (i.e. raising awareness, public planning) to much more costly defence and relocation measures (i.e. higher dykes, storm surge barriers, relocation of ports and urban centres [31]).

2.1 *Future Flood Risk Assessment*

Infrastructures (buildings, roads, sewer, electricity cables etc.) resulting from spatial developments as well as most flood defence measures are supposed to last for long periods of time. Thus, their planning and management requires insight in likely future necessities. As flood risk is by no means static, it is therefore important to anticipate on expected future changes in flood risk. Feyen et al. [32] estimated future flood risk in Europe, averaged over NUTS2¹ level. They used a hydrological model and inundation maps, depth-damage functions and different levels of flood protection based on the gross domestic product (GDP) per country. The impacts of both climate and land use change were included, and they estimated an average increase in expected annual damage for all European countries from 6.5 billion Euros in 2000 to 18 billion Euros in 2100 (converted to 2006 values). More detail is necessary to be able to map flood risk at the scale of individual countries. Flooding probabilities in the Netherlands, for example, are much lower than the once every

¹The Nomenclature of Territorial Units for Statistics (NUTS) is a geocode standard for referencing the subdivisions of countries for statistical purposes. NUTS regions are based on existing national administrative subdivisions. The population bandwidth for NUTS2 regions is 800,000–3 million, but this is not applied rigidly

Fig. 4 Prognosis of flood risk (expressed as Mln Euro damage/year) for the Netherlands. The relative influence of sea level rise (60, 85 and 150 cm in 100 years) on flood risk is compared with the influence of urban development, according to two scenarios: RC (B2, low economic growth) and GE (A1, high economic growth) (adapted from Aerts et al. [33])



100-year protection level that Feyen et al. [32] adopted. Safety levels in the Netherlands vary from flooding once every 1,250 years in the eastern part to once every 10,000 years in coastal areas.

At a national scale, Aerts et al. [33] have studied the independent influence of both climate change and socio-economic developments on flood risk in the Netherlands. For this purpose, a wide range of scenarios concerning climate change and land use developments were studied. Effects of climate change were modelled using three combined sea level rise and river discharge scenarios, which together influence flood probability. Furthermore, changes in urban development were assessed using two SRES scenarios (see Table 1 in [34], this volume), namely, low economic growth (B2 or Regional Communities) and high growth (A1 or Global Economy) [cf. 22, 34] (Fig. 4).

The results indicate that a moderate rise in sea level of 60 cm results in a similar increase in potential damage as a high economic growth scenario. Climate change effects only dominate for high increases in sea level, which is above 85 cm in the year 2100. These results highlight the importance of flood adaptation policies that limit both flooding probabilities as well as potential damage of flood disasters.

2.2 Dealing with Uncertainty in a Risk-Informed Approach

While clearly important, it is by no means straightforward to take future changes into account in a risk-informed approach. Long-term trends are inherently uncertain, hence difficult to translate into specific or necessary investment demands for daily operational river basin management. The prime source of uncertainty in flood (risk) prediction is the reliability of extreme value statistics [35]. Floods are very rare events, having return periods that are usually far outside the length of the period of the used observed time series data. One of the ways of dealing with this shortage of data is to statistically expand the existing set of observed data from

decades to centuries or even millennia. Te Linde and Aerts [36] used a weather generator model to create 1,000 years of daily precipitation and temperature data [37], in order to simulate daily discharges and annual maximum flood peaks. In this way discharges corresponding to design safety level return periods (i.e. flooding acceptable once every 100 years or once every 1,250 years) can be estimated using interpolation instead of extrapolation.

As the future is inherently unknown, scenarios are often used to explore possible or plausible futures (conform [34] and see also [34], this volume) and make better informed decisions in the present. Such scenarios represent different views on future societal development and provide images of how important driving variables are expected to change. It is of key importance that these variables are internally consistent, meaning that they should logically match within and among scenarios [22, 38, 39]. Furthermore, the scenarios applied would ideally cover the full width of conceivable future situations. Since any single scenario is equally plausible, all scenarios should be included in decision-making. The studies of Te Linde et al. [18] and Aerts [33] are both examples of scenario-based flood risk studies.

A more formal way of dealing with uncertainties is to perform statistical uncertainty analyses of the assessment models. Formal uncertainty analyses are common in integrated assessment models and many approaches exist (see e.g. [40]). In the commonly used Monte Carlo approaches, the assessment model is ran many times using different sets of random input parameter values, which are all varied within their own uncertainty range (see e.g. [41]). This will not result in a single number, but in a distribution of outcomes representing the many possible outcomes of the model given the uncertainties included. While a distribution has clear advantages over a limited set of scenarios, there are also some distinct disadvantages. Most importantly, depending on the model in use, such uncertainty assessments may require large computational capacity, which can become both time consuming and expensive. Furthermore, the uncertainty surrounding the input data must be quantified a priori, which is an inherent complication since the uncertainty ranges are generally estimates [42].

Full uncertainty analyses are used in early warning systems to estimate the bandwidth around water level predictions [43]. Furthermore, many efforts are being put in ensemble studies of climate models (e.g. the PRUDENCE [14] and ENSEMBLES [44] projects²) in order to get a better grip on the uncertainties and distribution of future climate parameters. Combining all these efforts into a full uncertainty analysis of the overall flood risk is a challenging task which is only recently starting to be picked up. Overall, uncertainty can be seen as one of the main remaining challenges in flood management [42], especially in risk-informed approaches that try to account for future changes.

² See: <http://prudence.dmi.dk/>

Table 2 Expected annual damage to agriculture and shipping (Mln Eur/year) (Adapted from: De Bruijn [45])

	Current (2000)	2050 (1 °C rise)	2050 (2 °C rise)
Expected annual damage to agriculture	351	488	624
Expected annual damage to shipping	90	175	280

3 An Outlook to the Assessment of Drought Risk

Just like flood risk, drought risk can be defined as the product of probability and consequence (e.g. damaged property and loss of life) and expressed in expected damage or loss per year. Few studies, though, seem to have attempted this. De Bruijn [45] made a quick scan for the Netherlands. Although densely populated, there is usually sufficient water in the Netherlands to meet the needs of all water use sectors (e.g. drinking water, agriculture, cooling water for power plants, naval transport over the river, industry and nature). The summer of 2003, though, caused severe and costly problems for water management: low river water levels hampered inland shipping; water quality deteriorated due to high temperatures and intruding brackish water; irrigation water for agriculture was restricted and power plants had to cut down their production because of a limited cooling capacity of the remaining river water.

Damages were estimated for two different climate change scenarios (1 or 2 °C temperature rise in 2050; Table 2). It is evident that in the current situation agriculture and shipping already suffer yearly damage. Apparently, both sectors have adapted or seem to cope with these annual damages [46]. Compared to the present situation, the expected damage to agriculture is estimated to increase with a factor of 1.4–1.8 and to shipping with a factor of 2–3. These projections remain inherently uncertain. Economic developments in the agricultural sector, for example, are influenced severely by world market prices, which are highly uncertain. Although estimated drought risk seems substantial, a first mitigation analysis shows that it is feasible and cost-effective to implement adaptation measures to cope with increased drought risk in 2050 in the Netherlands [47].

4 Conclusions and Recommendations

Dealing with natural hazards—like droughts and floods—in a risk-informed approach, which also takes future developments into account, is becoming more and more important in a wealthier and increasingly more densely populated world. Flood management is moving away from being the sole domain of engineers. Cooperation with social and economic sciences is—as further underpinned in book Section C—a prerequisite for successful risk-informed strategies. Integrating these different fields of research is one of the major challenges in informing river

basin management on flood and drought risks taking future uncertainties into account. Such risk-informed river basin management could help basin managers to distinguish between urgency and priority of risk reduction measures. Due to the huge investments, we desire a long lifetime of infrastructure (roads, buildings, sewers, electricity cables, etc.) and defensive structures. Thus, it should be accounted for future trends related to the hazard (e.g. climatic change), damage (infrastructure) and loss (human lives). Furthermore, the uncertainties of assessments, particularly those related to scenario-based future outlooks, should be assessed and communicated adequately in order to enable better informed decisions.

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Future Land Use Patterns in European River Basins: Scenario Trends in Urbanization, Agriculture and Land Use

Fritz A. Hellmann and Hans de Moel

Abstract Land use in the European Union is expected to change significantly during the next decades. This may cause important hydrological impacts. Land use change was modelled in ten different European river basins using the Intergovernmental Panel on Climate Change (IPCC) Special Report on Emission Scenarios (SRES). The model results predict some overall scenario trends that are comparable in most of the basins. The ‘(A1) Global Economy’ and ‘(B1) Global Cooperation’ scenarios predict a large increase in urban area plus abandonment of agricultural land. The ‘(A2) Continental Markets’ scenario predicts an increase both in urban area and in agricultural land. The ‘(B2) Regional Communities’ scenario predicts a relatively small increase in urban area and modest land abandonment. Overall trends can also be discerned for each land use type. Urban area increases in all scenarios and all river basins. Agricultural land use decreases in all scenarios due to urbanization and land abandonment, except in the ‘Continental Markets’ scenario. Natural vegetation and abandoned land increase in most scenarios as a result of the projected abandonment of agricultural land, except in the ‘Continental Markets’ scenario. There are also differences between individual river basins. The Loire, Seine and Tevere basins all show a change from agricultural land towards abandonment and natural vegetation, but magnitude and direction of this change differ between scenarios. The same is predicted, but to a lesser extent, in the Ebro, Elbe, Kokemäenjoki and Tago basins. Land use in the Oder basin remains relatively stable except that some pasture is abandoned in the ‘Regional Communities’

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scenario. The Thames basin consistently shows high levels of urbanization in all scenarios, whereas land use change in the equally urbanized Scheldt basin varies considerably between the scenarios and is rather unpredictable. In general, the distribution of land use within river basins does change—but not dramatically—over the modelled time horizon of 30 years. The magnitude of changes in land use predicted in this study will not likely cause large-scale impacts on the hydrological characteristics of entire river basins, except for the sediment load of the river. Nonetheless, they may cause important local impacts, i.e. at the level of small catchments and sub-catchments such as increased occurrence of low flows, drying up of streams and flash floods.

Keywords Europe • Land use • Land use change • River basins • Scenarios

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1 Introduction

Rural area in the European Union is expected to undergo profound changes in land use during the next decades. Ongoing agricultural intensification and mechanization are thought to lead to a contraction of Europe's agricultural area and depopulation of rural areas with marginal yields [1–4]. Although it is not yet clear how the abandoned land areas will develop, substantial parts will probably get occupied with natural vegetation such as forest [4, 5]. Urbanization is another important trend in many rural areas and will significantly change traditional landscapes and land use characteristics in these areas [3, 6].

These land use changes are expected to cause a great impact on the hydrological characteristics of a river basin. Natural vegetation (such as forest) has a considerable retention capacity: it retains water during heavy precipitation and slowly releases the water again afterwards [7, 8]. In addition, natural vegetation has high

transpiration and evaporation rates compared to cropland and it improves the soil infiltration capacity. Hence, a decrease in natural vegetation (i.e. deforestation) in a river basin will cause larger and more unpredictable (peak) discharges [7–10], whereas an increase in natural vegetation or reforestation can result in the opposite [11]. Especially urbanization may cause negative hydrological effects due to the very low infiltration capacity of its paved areas [10].

Because land cover changes are known to cause large impacts on hydrological processes, more insight into the future development of Europe's rural area will help in understanding what potential changes in hydrological processes can be expected. Several studies have addressed the effects of rural land use changes within the European Union on hydrological processes [10, 12–14]. However, many of these studies have described land use changes at the local catchment scale and not at multiple river basin scales simultaneously. This makes it difficult to compare changes in land use between different basins and discerns main trends from local changes. Therefore, the aim of this study is to determine land use changes in a range of European river basins and to compare trends in land use changes based on land use change scenarios derived from a previous study [15]. Scenarios based on the well-known Intergovernmental Panel on Climate Change (IPCC) Special Report on Emission Scenarios (SRES) that describe different societal development directions (see Table 1 in [16], this volume) were used to derive assumptions regarding key drivers of land use change. Using these assumptions, the scenarios are translated with the EURURALIS¹ model ensemble into local land use changes at a spatial grid of 1 km² over the period 2000–2030. Subsequently the predicted changes in land use in different European river basins were analysed and compared to each other.

2 Methodology

2.1 Outline

The data used in this study have been generated within the EURURALIS project [15]. This project predicted the potential development directions of the rural areas in the (at that time) 25 European Union member states (hereafter EU25) using the IPCC-SRES scenarios A1, A2, B1 and B2 [16–18]. The EURURALIS consortium attached its own label to these scenarios: A1, Global Economy; A2, Continental Markets; B1, Global Cooperation; and B2, Regional Communities (see [2] and see Table 1 in [16]). Assumptions regarding key drivers of land use change (e.g. macroeconomic and demographic developments) were derived from the scenario interpretations [18]. These key drivers were used as input for the macroeconomic

¹ See <http://www.eururalis.eu/>

model ‘Global Trade Analysis Project’ (GTAP)² and the ‘Integrated Model to Assess the Global Environment’ (IMAGE)³ to estimate the aggregated agricultural land use changes per EU25 member state over the period 2000–2030 [19]. The aggregated land use changes were then translated into local changes at a spatial resolution of 1 km² by using the land use change model ‘Dyna-CLUE’ ([5]).⁴

2.2 Models

GTAP is a macroeconomic model that estimates international trade between world regions using input-export tables for each world region [19]. Production systems within each region are modelled through input–output tables that include the whole production chain. At each step in the production chain, the most profitable use of resources is determined. The consequences for the production chain and other economic sectors are determined. Scenario assumptions regarding key drivers of land use (e.g. macroeconomic and demographic developments) have been considered as preconditions within the model. In this model structure, agricultural production was included as a production chain with some adjustments to account for the fact that land use types are not perfectly interchangeable and that there is a limited land supply [20].

The integrated assessment model IMAGE was used to adjust the estimations of GTAP regarding agricultural production by taking into account the effects of climate change and land use change [21]. It determines the effects of climate change and of the land use changes as estimated by GTAP on the agricultural yields used in GTAP. The modified yields are returned to GTAP, which renews its estimations with the modified yields. The looping between GTAP and IMAGE continues until the models converge. The resulting output simulates the production and trade of goods and services between different world regions and includes the land area dedicated to arable and livestock production per EU25 member state.

Spatial land use changes have been simulated by Dyna-CLUE using competition among land use types [3]. This competition is based on the comparative suitability of a location for the various land use types. The comparative suitability of a location for the various land use types is based on local biophysical and socioeconomic factors. At each location the model allocates a land use type with the highest suitability and then compares the allocated changes in land use with the aggregated national land use changes that have been estimated by GTAP/IMAGE. If the models do not converge, Dyna-CLUE adjusts the comparative suitability of the under- and over-allocated land use types until the models do converge. In this way, Dyna-CLUE translates the aggregated national land use changes estimated by

² See <http://www.gtap.agecon.purdue.edu/>

³ See <http://themasites.pbl.nl/tridion/en/themasites/image/index.html>

⁴ See <http://dyna-clue.software.informer.com/>

GTAP/IMAGE into local land use changes at a spatial resolution of 1 km² [3]. Time steps of 1 year are used and a total period of 30 years is modelled (i.e. 2000–2030). Dyna-CLUE actively models competition between six land use types, and an additional eight land use types are included that are not dynamic (e.g. bare rock, glaciers). Additionally, two land use types are used to model land abandonment (i.e. recently abandoned arable land and recently abandoned pasture). Land abandonment is not actively modelled in Dyna-CLUE, but is allocated to locations that are literally ‘leftover’ after sufficient area has been allocated to other land use types to meet the area requirements estimated by GTAP/IMAGE. After a set time period, abandoned land is assumed to have developed a full natural vegetation cover and changes automatically into, first, the land use-type ‘natural vegetation’ and, later, into forest depending on the local conditions for forest regrowth. Land use types ‘abandoned land’ and ‘natural vegetation’ are thus coupled.

2.3 River Basins and Land Use Categories Included in the Analysis

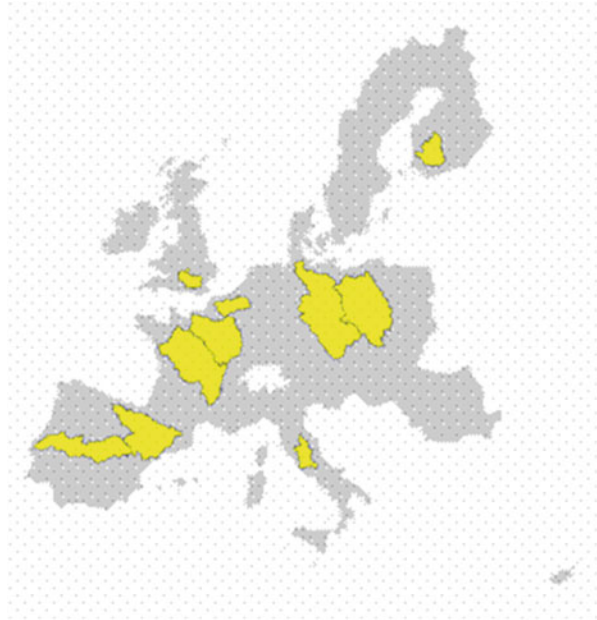
EURURALIS scenario outputs for 2030 are compared with the land use map of the initial year (i.e. the year 2000) to determine which land use changes have occurred in different river basins in each scenario. Land use changes have been assessed for the following ten European river basins: Ebro, Elbe, Kokemäenjoki, Loire, Oder, Scheldt, Seine, Tago, Thames and Tevere (Fig. 1). These basins were chosen because (1) they give a reasonable geographical coverage for EU25, (2) most are relatively large or (economically) important rivers, and (3) EURURALIS outputs cover the entire river basin. The latter is not the case for the Rhine and Danube river basins. That is the reason why these important European basins are not included. The various land use types modelled within the EURURALIS project were aggregated into five broad land use categories in this study: urban area, natural vegetation, abandoned land, pasture and arable land.

3 Results and Discussion

3.1 Scenarios

In general, it is predicted by the modelling exercise that the largest parts of the selected river basins keep the same land use types. However, locally considerable landscape changes are sometimes predicted. While land use changes within a scenario can differ between river basins, some overall scenario trends can still be identified after pooling land use changes across river basins per scenario (Fig. 2).

Fig. 1 River basins included in the analysis, with the EU25 as background



The main predicted changes in the ‘(A1) Global Economy’ scenario are a large increase in urban area (the largest increase in urban area of all scenarios) and a large decrease in arable land (but not of pasture) resulting in a substantial increase in natural vegetation and abandoned land. This predicted high urbanization rate is probably due to the scenario’s limited government regulation, strong population growth (including immigration) and relatively high levels of affluence. This would result in a large, wealthy urban population, requiring a lot of area per person. Because this high demand for space per person is not balanced by government restrictions, high urbanization rates are to be expected. The decline in agriculture is probably coupled to the termination of the EU agricultural subsidies and reduction of import tariffs in the ‘(A1) Global Economy’ scenario.

In the ‘(B1) Global Cooperation’ scenario, land use changes are predicted to be comparable to those of the ‘(A1) Global Economy’ scenario. Urbanization rate is somewhat lower and agricultural land abandonment (of both arable land and pasture) is higher. This is expected to result in a large increase in natural vegetation and abandoned land. EU agricultural subsidies and border/domestic support (import tariffs) are phased out as well, which explains the predicted substantial abandonment of agricultural land.

The ‘(A2) Continental Markets’ scenario predicts urbanization rates that are comparable to the previous two scenarios. This is somewhat surprising given the lower population growth and affluence level compared to the ‘(A1) Global Economy’ and ‘(B1) Global Cooperation’ scenarios. A probable explanation can be the absence of restrictive spatial planning regulation. In contrast with the other scenarios, the area of agricultural land is expected to remain relatively stable or

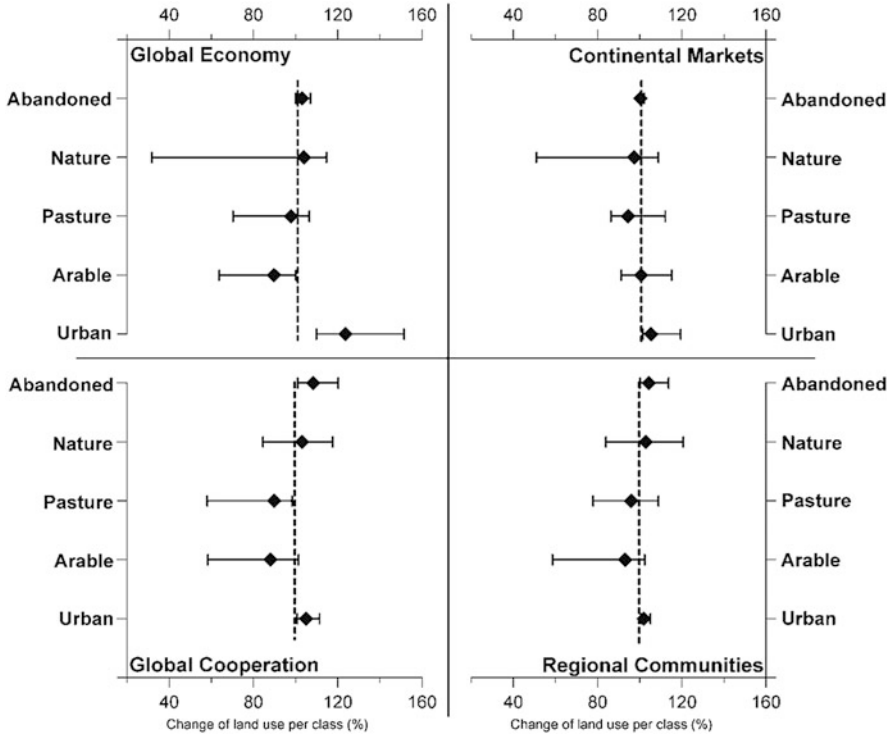


Fig. 2 Median and full range of the land coverage of each land use type among all river basins per scenario (given as a percentage of the area of that specific land use type in the base year 2000). In the case of abandoned land, the area of the abandoned land in each scenario is expressed as a percentage of the combined area of arable land and pasture in the base year because the area of the abandoned land in the base year is not known. The vertical broken line indicates 100 % or no change compared to the base year

will even increase in some river basins. This expansion of agricultural land occurs at the expense of natural vegetation and consequently results in a decrease of natural vegetation and a very small increase in abandoned land in this scenario. Agricultural policy aimed at stimulating production through export subsidies and import barriers is probably the reason for the strong agricultural growth in this scenario.

The '(B2) Regional Communities' scenario predicts relatively low levels of urbanization, which is consistent with the assumptions of low population and economic growth plus governmental regulations on urban sprawl. This scenario also predicts a relatively small decrease in agricultural land and a corresponding modest increase in natural vegetation and abandoned land. This can be attributed to the desire for local self-reliance and associated support for local agricultural production.

3.2 Predicted Trends in Land Use

Overall trends can also be discerned for each land use type (Fig. 2). Urban area is expected to increase in all scenarios and all river basins, but the increase is especially pronounced in the ‘(A1) Global Economy’ scenario. Urbanization is expected to be much lower in the ‘(B2) Regional Communities’ scenario and intermediate in the ‘(A2) Continental Markets’ and ‘(B1) Global Cooperation’ scenarios. This is in line with the scenario assumptions: the ‘(A1) Global Economy’ scenario assumes the highest increase in population and economic growth and ‘(B2) Regional Communities’ the lowest. The difference in variation among river basins is notable (i.e. larger range in Fig. 2) between the ‘(A1) Global Economy’ and ‘(A2) Continental Markets’ versus ‘(B1) Global Cooperation’ and ‘(B2) Regional Communities’ scenarios. The differences in predicted urbanization among river basins are clearly much larger in the first than in the latter two scenarios. Urbanization is predicted to be an especially important land use change process in the Thames, Seine, Tago and Loire river basins.

For the two agricultural land use types (arable land and pasture), a more complicated trend is predicted. Arable land decreases in most scenarios, except for the ‘(A2) Continental Markets’ scenario where in some river basins, a reasonable increase in arable land is expected. Despite the overall decrease in arable land, considerable differences exist among river basins, and for some of them, even a small increase in arable land is expected in some scenarios. Pasture generally decreases in all scenarios, but considerable differences exist among river basins and among scenarios (with some basins actually displaying a small increase in pasture in one or more of the scenarios). Unlike with arable land, the large decreases in pasture are expected in different river basins in the scenarios.

Natural vegetation is expected to generally increase a bit in the ‘(B1) Global Cooperation’, ‘(B2) Regional Communities’ and ‘(A1) Global Economy’ scenarios, but to decrease in the ‘(A2) Continental Markets’ scenario. For the Thames large decreases in natural vegetation were estimated in all scenarios. Abandoned land is expected to increase substantially in all scenarios except for the ‘(A2) Continental Markets’ scenario. Land abandonment occurs in almost all river basins and in all scenarios.

3.3 Predicted Trends per River Basin

Depending on the scenario, the Loire, Seine and Tevere river basins either expect a transition from agricultural land (arable land and pasture) to abandoned land and natural vegetation, or vice versa (Figs. 3 and 4). However, in most scenarios, agricultural land is abandoned and the land use types abandoned land and natural vegetation are expected to increase substantially. Only in the ‘(A2) Continental Markets’ scenario the opposite process may be expected because the area of

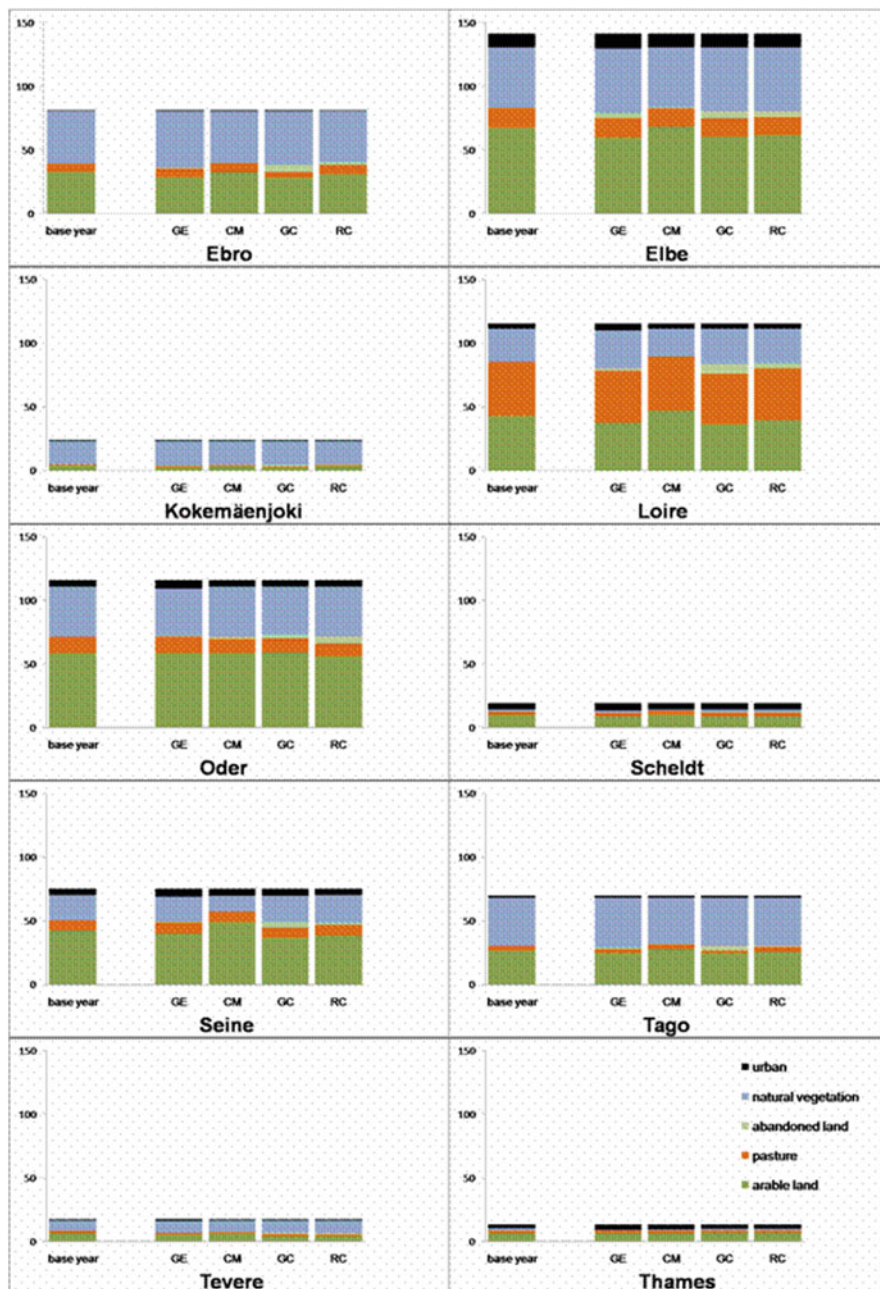


Fig. 3 Predicted absolute land coverage (in 1,000 ha) of the different EURURALIS land use types in 2030 per river basin according to the four scenarios compared to the base year 2000. GE, Global Economy (A1); CM, Continental Markets (A2); GC, Global Cooperation (B1); RC, Regional Communities (B2)

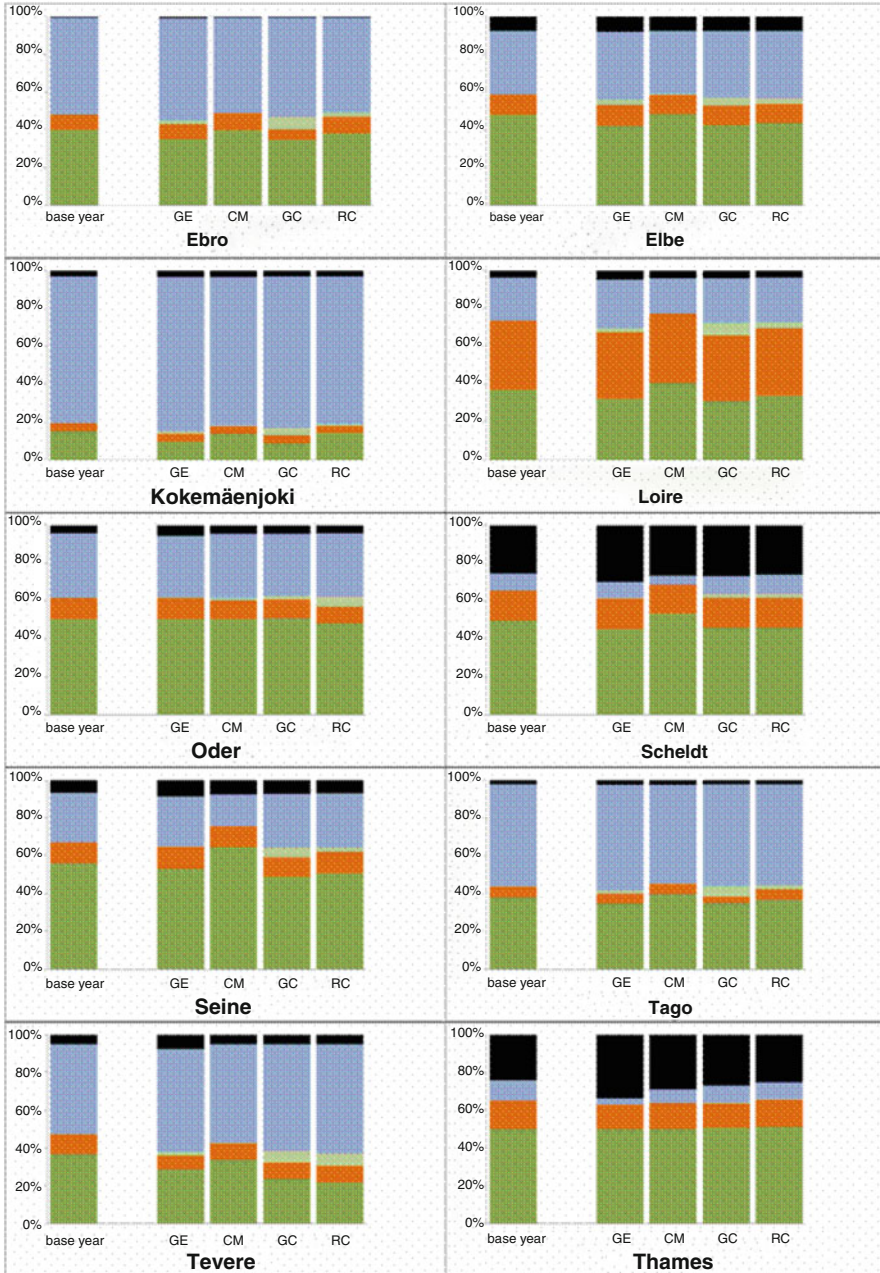


Fig. 4 Proportional land use predictions of EURURALIS per river basin in 2030 according to the four scenarios compared to the base year 2000 (given as a percentage of the catchment's area). Further as Fig. 3

agricultural land is predicted to expand at the expense of natural vegetation. The same interaction between agricultural land versus abandonment and natural vegetation is expected in the Ebro, Elbe, Kokemäenjoki and Tago, though less extensive. Land use in the Oder river basin remains relatively stable except that some pasture is expected to get abandoned in the '(B2) Regional Communities' scenario.

In the Thames river basin, the expected ongoing urbanization makes that natural vegetation disappears and the urban area increases in all scenarios. The urbanization rate is especially high in the '(A1) Global Economy' scenario, which is not surprising as population growth and economic progress are highest in this scenario.

Initially, the Scheldt river basin is equally urbanized as the Thames, but it shows different land use changes and experiences lower urbanization rates in all scenarios. In the '(B1) Global Cooperation' and '(B2) Regional Communities' scenarios, a small increase in urban area and natural vegetation is expected in the Scheldt river basin at the expense of arable land. In the '(A2) Continental Markets' scenario, an increase in urban area and arable land is expected in the Scheldt river basin at the expense of natural vegetation. A high urbanization rate is expected to occur at the expense of arable land and natural vegetation in the '(A1) Global Economy' scenario.

3.4 Predicted Impacts on Hydrology

Large-scale changes in land use are likely to cause a significant impact on discharge and high-flow frequencies [9], for example, on the deforestation of the Meuse river basin [22]. However, modest changes in land use have a minor effect on discharge distribution. Tu [23] showed that historical land use changes in the Meuse river basin over the twentieth century had a marginal impact on the discharge. Similarly, Ward [22] showed for the Meuse that predicted changes in discharge for the twenty-first century can mainly be attributed to climate changes and not to changes in land use. Modest changes in land use can, however, have a large impact on sediment yield [22]: on both local and entire river basin scale. This will affect river morphology, erosion and sedimentation (see also chapters by Négrel et al. [24] and by von der Ohe, Apitz et al. [25], this volume).

While land use changes as modelled in this study are not likely to cause large effects on the discharge distribution of reasonably sized river basins, they can have a considerable impact at the local scale (i.e. small catchments or sub-catchments). For instance, De Moor [26] found that historically, local land use changes considerably influenced the hydrology of the Geul, i.e. a sub-catchment of the Meuse. Likewise, Keesstra [27] showed that natural reforestation (going from 30 % to 70 % during 1954–1985) had a large impact on the hydrology of a small Slovenian catchment (i.e. Dragonja). Discharge dropped dramatically, with especially low flows decreasing by about 85 %, resulting in a much more frequent drying up of the stream. Hence at local scales, especially far upstream in river basins or in very small catchments in general, land use changes, as modelled for the river basins under

investigation, can have substantial effects on river hydrology and sediment yield. River basins with a relative large predicted increase in natural vegetation, like the Tevere (up to 58 % of the river basin area in the (B2) Regional Communities scenario), may respond with a reduced base flow because of the increased evapotranspiration potential due to forest; that potential is much lower at arable land and pasture. This can be exacerbated because of climate change. River basins with a large expected increase in urban area, like the Thames, Seine, Tago and Loire, are likely to face more frequent flooding events after extreme rainfall. This is caused by the reduced infiltration capacity and increased surface run-off of paved urban areas.

4 Conclusions

Based on the scenarios (IPCC-SRES) and models (GTAP, IMAGE and Dyna-CLUE) used in this chapter, we can draw the following main conclusions:

- In general, dramatic land use changes have not been predicted to occur over the period 2000–2030, although locally considerable landscape changes can sometimes be expected.
- When expressed per river basin, relative land use changes can be very different among scenarios, but often remain small in actual area.
- When analysed by scenario, land use changes differ among European river basins: some are expected to urbanize to a large extent, where for others land abandonment is predicted.
- Urban land use is expected to increase in all European river basins in all scenarios. In many basins a transition is expected either from agricultural land use (arable land and pasture) to nature and abandoned land, or vice versa, depending on the scenario. In most cases, agricultural land use is expected to decrease in favour of nature and abandoned land, except for the '(A2) Continental Markets' scenario where agricultural land use even increases slightly in many river basins at the expense of nature.
- The land use changes as predicted in this study are expected to hardly impact the hydrology of the entire river basin. However, they are large enough to impact sediment loads, both on local and on the entire river basin scale. The predicted land use changes can be substantial in sub-catchments that are subject to considerable land use change. Notably, further urbanization in Thames and Scheldt river basins is expected to lead to increase the incidence and volume of peak flows. Furthermore, a larger area of natural vegetation (as a result of land abandonment) may reduce base flow in Mediterranean river basins like the Tevere.

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Framing the Uncertain Future: Articulating IPCC-SRES Scenarios for European River Basins

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Fritz A. Hellmann, Wim Salomons, and Tijs van Maasackers

Abstract Downscaled articulations of Intergovernmental Panel on Climate Change (IPCC) Special Report on Emission Scenarios (SRES) have been outlined qualitatively for a hypothetical Southern, Central and Northern European river basin and a time horizon set at 2030. The purpose was to survey the sensitivity of ecosystem state indicators, to assess which drivers would be within the grasp of river basin management and to make a geographic comparison. Expert workshop debates were structured using a sequence of entries on drivers and the wider geographic setting, on the river basin and its hydrology, on pressures and on a range of ecosystem state indicators. The workshop elaborated IPCC-SRES scenarios A1 (global economy)

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and B2 (regional communities) only, since these are generally considered to be the two most divergent scenarios. Contrasts between these two scenarios in land, resource and energy use as well as in the orientation towards sustainability in governance were thought to lead to distinct contrasts in water and sediment delivery to stream networks, in contaminant loads and their remobilisation and in opportunities for riparian biota to populate available habitat. Also, these contrasts between A1 and B2 are probably most profound in the North and South. In contrast to other scenario assessments in the literature, the workshop found it highly plausible that agricultural land use would expand in the North, notably on deeper soils that had been afforested in previous decades. For the South, uncertainty on the direction of land use change was profound, leading to quite different, sketchy but plausible trajectories. Workshop participants remained cautious in the use of scenarios, because it was felt that adoption of altered lifestyles, transition to a carbon-neutral energy system or a nutrient-balanced, low-external-input agriculture can be charted as scenario elements, but their wholesale assimilation in real societies over the coming decades remains hard to predict. Notably discharge variability was foreseen to be highly responsive to the different scenarios but is considered to be under the influence of a river manager. Major drivers, such as the Common Agricultural Policy of the European Union and world market demand development for dairy or biofuel, will strongly affect land use and soil management. These appear to be largely outside the span of control of river basin authorities. The workshop hoped that risk preparedness in river management would include an identification of such major drivers outside their formal control and of the relevant institutions, both in different sectors and at different levels.

Keywords Expert consultation • River management authority • Scenario divergence • Span of control

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1 Introduction

Faced with risk and uncertainty, environmental policy-makers are increasingly using scenario planning to guide decision-making [1–3]. In the wake of the use of scenarios at the global scale (notably [4, 5]), there is an increasing interest in

Table 1 Working table for three parallel Dahlem-type expert workshops

riskbase scenario articulation for a typical northern/central/southern catchment											
entry	<table border="1" style="display: inline-table; vertical-align: middle;"> <tr> <td>A1</td> <td>B1</td> </tr> <tr> <td>A2</td> <td>B2</td> </tr> </table>		A1	B1	A2	B2	scenario: keywords:	A1	A2	B1	B2
	A1	B1									
A2	B2										
			World markets Growth is good	Provincial Enterprise continental markets pull up the drawbridge	Global sustainability/ global cooperation we've got the whole world	Local stewardship Regional communities Think local, act local					
<i>the wider setting</i>											
(a) economics: markets in and around the river basin											
(b) demography: human population in the river basin					maybe compare with the table in our articulation text						
(c) land use: agriculture and other land use					decide where the differences are						
(d) governance and institutions: style/culture											
(e) recreation: style and intensity											
<i>the catchment and its drainage</i>											
(f) hydrology: rainfall, evapotranspiration, infiltration											
(g) soils and sediment: production, throughput and delivery											
<i>the pressures</i>											
(h) (change in) water quantity: the river discharge pattern changes											
(i) hydromorphological changes: e.g. damming, embankment, goines											
(j) erosion: of soils in the catchment or of banks along the stream											
(k) compaction of arable land											
(l) sealing of the catchment due to urbanisation/roads											
(m) organic matter decline in the soils of the catchment											
(n) contamination (of...)											
(o) salinisation of catchment soils											
(p) acidification of catchment soils											
(q) (successful) settlement of invasive species											
<i>state indicators</i>			change due to pressures (major ones only)								
(r) concentrations of chemicals (any)			h, j, n, o, p								
(s) trophic state (probability of a eutrophic condition: high nutrient availability leading to high productivity)			n								
(t) biota: composition of the communities of living organisms			n, o, p, q								
(u) buffering capacity			m, p								
(v) salinity (concentration of salts in the water)			o								
(w) suspended matter and sediment			h, i, j, k, l, n, q								
(x) water level (in surface waters)			h, i, j, k, l, n								
(y) morphology and pedology of the river basin			h, i, j, k, l, n, q								

Each entry to the table was debated. Pressures and state indicators were adopted from chapters by Négrel et al. [21] and by von der Ohe et al. [22], this volume

applying such concepts at smaller scales to inform regional and local management and policy [6]. Regional land use and water managers as well as policymakers attempt to project what the consequences of global climate change could be in their specific areas; hence, downscaling and articulation of global scenarios to regional and sectoral applications have become increasingly relevant. The spatial articulation of common global environmental change model ensembles is simply too crude [7] when realistic impact estimates of land use pattern change or topography are required at the river basin scale [8, 9].

Downscaling exercises to the river basin scale have been carried out in various ways, depending on focus and purpose [10]. For straightforward numerical indicators, such as gross domestic product (GDP), linear or proportional weighing has been used [11] and population density and economic activity have been gridded using demographic data or even night-time lighting maps [12]. Using the Intergovernmental Panel on Climate Change (IPCC) Special Report on Emission Scenarios (SRES) scenarios A1 (global economy) and B2 (regional communities) (see Table 1 in [5], this volume, i.e. [6]), Döll and Vassolo [13] demonstrated that sectoral water demand estimations in 2025 benefited from downscaled interpretations of the scenario storylines extrapolated from regionally specific rather than

national data, with lower absolute water needs and higher reductions in withdrawal. They concluded these were more plausible and realistic. Often, however, down-scaling by straightforward calculus is not easy and expert judgment is necessary (e.g. [14, 15]).

This chapter reports on a set of three parallel workshop sessions that were carried out to survey possible future outlooks for European river basins based on the IPCC-SRES [16–18]. These scenarios were closely followed, because they are well articulated [9] and have developed into benchmark inputs for a body of quantitative modelling (e.g. [19]) and qualitative exploration, as in this study. An interdisciplinary group of experts worked together to develop three sets of plausible and consistent views on the future drivers, pressures and state, including land use, hydrology, sediment dynamics, nutrients, contaminants and in-stream biota of three typical, but conceptualised river systems draining the landscapes that constitute their basins. Three parallel sessions considered each a hypothetical Northern, Central or Southern European river basin (as far as such a concept is plausible), with a time horizon set to around 2030. The aims were (1) to extrapolate existing scenario articulations to ecosystem state indicators that are likely relevant to the river basin manager and (2) to compare these outcomes across the geographic range spanned by European river basins (e.g. [20]).

2 Approach

Prior to the workshop, baseline information was prepared ([6, 21, 22] i.e. chapters by Vermaat et al. [5], by te Linde [18] and by Helmann and de Moel [23], this volume) and allowed participants a well-informed, equal footing as prescribed in the Dahlem workshop format.¹ Participating experts were recruited from the wider RISKBASE network and represented a range of disciplinary expertise's from science and management.

An overview of the regionalised IPCC climate change predictions for temperature and rainfall for their regions based on Christensen et al. [9] was given to each work group.² Christensen et al. [9] present downscaled maps of temperature and rainfall change between 1990 and 2090 for the A1b scenario (Fig. 1) and predict that global mean temperature as well as precipitation patterns for the other scenarios scales consistently as 0.7 (B1): 1 (A1b): 1.2 (A2). Although the B2 scenario is not explicit in the regionalisation of IPCC's fourth assessment [9], it was assumed here that it would generally offer the widest contrast with A1 (cf. [6, 8, 13]). This strong contrast was helpful in clarifying the projection of scenario consequences on societal values and organisation. Work groups therefore

¹ See <http://www.fu-berlin.de/veranstaltungen/dahlemkonferenzen>

² Swantje Preuschmann (Max Planck Institute for Meteorology, Hamburg) is acknowledged for explaining regionalised climate projections during the workshop

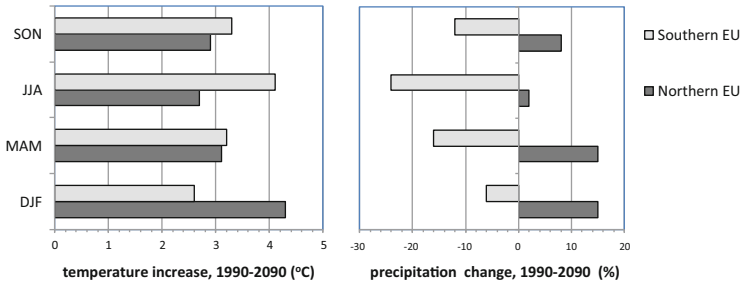


Fig. 1 Estimated seasonal changes in air temperature (*left*) and precipitation (*right*) over the period of 1990–2090 by the IPCC model ensemble for Northern and Southern Europe using the IPCC-SRES A1b scenario (derived from Christensen et al., 2007). North and South is delimited by 48° Northern Latitude; seasons are winter (DJF), spring (MAM), summer (JJA) and autumn (SON). The IPCC foresees a most likely worldwide annual temperature increase of +2.8, +3.4, +1.8 and +2.4 °C for A1b, A2, B1 and B2, respectively, for the same period and +0.6 °C if greenhouse gas concentrations would remain constant at their 2,000 values. For ~2030, the pattern in A1b is similar, but still less extreme

focused on the articulation of A1 and B2. Different interpretations of the SRES storylines exist, notably in the contrast between B1 and B2 in the emphasis on environmental protection (compare [8, 11, 16]), but also in land use allocation [1]. The original interpretation of Lorenzoni et al. [16] was followed here (cf [6], i.e. Table 1 in [5], this volume). IPCC’s global circulation model ensembles were found to generally respond linearly with time [9, 24], with little evidence of robust non-linearities. Thus, the broad scenario-driven trajectories of climate change can be conceived as developing steadily, without additional complexity of major non-linear shifts. This justifies flexibility in setting the time span for this articulation exercise not too narrowly around 2030, since the differences among the scenario patterns magnify with time.

The discussion followed an ordered sequence of entries moving from the wider socio-economic setting as drivers, via the river basin and its drainage, to the resulting pressures and changing state indicators, as guided by a table provided to each work group (Table 1). Articulation of the wider setting was based on the discussion of land use planning styles as worked out for the Scheldt river basin in Vermaat et al. ([6] i.e. chapter by Vermaat et al. [5], this volume). Work groups conceptualised how scenarios (“A1—global economy” and “B2—regional communities”) might affect the interactions between drivers, pressures and states (see Fig. 1 in [25], this volume) and how different economic, regulatory and societal models response to change. A list of relevant pressure and state indicators was adopted from previous RISKBASE workshops (conform [26, 27], i.e. chapters by Négrel et al. [21] and by von der Ohe et al. [22], this volume). Using the template depicted in Table 1, work groups evaluated (1) the scope of influence by a river basin manager, (2) the sensitivity of an entry to the table to among-scenario variation and (3) the timescale over which an entry would affect a detectable change in the river system. Work groups concluded with a gap analysis, focusing on issues

that drove uncertainty, gaps or data needs as well as on tools and/or models that were either needed or available. These aspects of scope, sensitivity, timescale, uncertainty and gaps were reported here jointly for Northern, Central and Southern river basins. In a concluding plenary session, the main findings of each parallel session were reviewed. The output spreadsheets were edited by the reporters and then cross-checked and reviewed by the authors of this chapter.

3 Results

The workshop participants elaborated the wider socio-economic implications of A1 and B2 for their respective, hypothetical river basins (Table 2). Notably for the North and South, vivid changes compared to the present and contrasts between A1 and B2 were foreseen. In the North, the contrasting scenarios were thought to lead to major differences in lifestyle, crop types and land use and energy intensity of society. Previously set-aside land would be returned to agricultural use in the North where soil conditions are favourable. Similarly, the South was perceived as developing into either a tourist-industry-driven narrow coastal strip with a semiarid hinterland or an agroforestry- and water-harvesting-oriented countryside stewardship. In the B2 scenario it is anticipated that local patchwork mosaics may create new landscapes, possibly reconnecting to traditional dry-land agriculture practices. Small networks of neighbouring rural villages could well be highly viable, though at lower overall population density than at present. In the North, population density was foreseen to increase for both A1 and B2 (partially as a result of migration), but less so in the latter scenario, where border and population controls might be more stringent. These stark contrasts in foreseen societal development echoed through in the articulation of ecosystem pressures and state indicators. For the hypothetical Central European river basin, workshop participants conceived less extreme and comparatively intermediate development.

Whereas previous assessments of plausible changes in river discharge pattern could be obtained from the literature (e.g. [9, 28] Fig. 1), extrapolations had to be made on sediment production in the river basin and sediment fluxes through the river network. It was deduced (Table 3) that both in the North and South, soil loss may increase due to the anticipated increased intensity of more frequent high-rainfall events during summer and due to increased wetness during winter in the North. Depending on the scenario, Northern and Southern societies may respond in a more or less adaptive and prudent manner. Under B2, societies were thought to adopt soil conservation and sustainable land husbandry practices, mitigating the impacts of altered hydrology on erosion and sediment dynamics.

For the A1 scenario, the major pressures on the river network in both North and South were related to increased urbanisation of river basins and enhanced engineering of river channels (Table 4). The purpose of these societal responses, however, differs among North and South. In the North, engineering was expected to be designed to cope with higher flow variability and enhanced navigation needs,

Table 2 Anticipated changes in the wider socio-economic setting for a hypothetical Northern, Central and Southern European river basin

Scenario: River basin:	A1, relative to situation in 2009			B2, relative to 2009 and A1		
	North	Central	South	North	Central	South
economics: markets in and around the river basin	continued economic growth across all markets	high tech economic growth (services)	exclusive reliance on tourism, strongly focused on coastal zone	regionally oriented markets; focus on local products requires broad range of (local) industries, markets moderate demand	local markets, less economic growth, less innovation	farmers become land stewards
demography: human population in the river basin	urbanisation, close to the coast, where sea level rise may force inland shifts; population migrates north from the south	increase by migration from the south	mass exodus, intensified urbanization along coast	less climate-induced migration, more regional barriers to movement, greener cities	stable	decrease in population density, small concentrated villages, patchy development pattern, self-organized, local population control
land use: agriculture and other land use	increased demand for agricultural products drives land use change: more crop- and grazing land; Northern Europe becomes more favourable for agriculture	small decrease	present water-intensive agriculture is abandoned	crops differ, agricultural land use increases; agricultural practices more sustainable, focus on local products/crops	slight increase in agriculture, increased demand for local food	Agri-forestry, use of traditional ecological knowledge, water harvesting at household level
governance and institutions: style/culture	deregulation and economic growth, WFD objectives relaxed, derogations pre-dominate	less rules, economic driven	market-controlled, breaking down trade barriers, imports dominate food-markets	huge change in lifestyle, WFD objectives balanced against services, subsidies shifted to sustainability, society makes catchment-sensitive choices	more consensus driven, more rules	local institutions become more autonomous, bottom-up, sense of place, resilience increases
recreation: style and intensity	Scandinavia more attractive in summer, for all of Europe; everyone still flies to holiday destinations	more diverse and intensifying	mass tourism increase, shift towards 'Costas' tourism, hedonic beach life	a reevaluation of what is necessary leads to a life style shift; recreation occurs near home; air and car travel limited	local recreation in countryside	ecotourism on the rise

The anticipation is based on the articulation of the IPCC-SRES scenarios A1 (global economy) and B2 (regional communities). The changes are entered in the table as narrative descriptions of plausible future development trajectories up to ~2030

Table 3 Anticipated changes in river basin hydrology and sediment dynamics for a hypothetical Northern, Central and Southern European river basin

Scenario: River basin:	A1, relative to situation in 2009			B2, relative to 2009 and A1		
	North	Central	South	North	Central	South
hydrology: rainfall, evapo-transpiration, infiltration	increase in winter rain, less rain in summer, less infiltration in summer, more frequent dry rivers, more winter flushing	hydrology more variable, temperature rise affects evapo-transpiration, hence summer run-off will decline	less evapo-transpiration due to lack of water, overall decline of water available	less extreme changes than in A1; land use will be more intensive but management will be more sustainable; some countries already close	see A1	more infiltration due to agroforestry and water harvesting
soils and sediment: production, throughput and delivery	increased soil loss where agriculture increases without improved land husbandry; more sediment production in mountain areas, complex interplay of floodplain sediment delivery and increased damming; possible coastal sediment starvation; potential extremes in deposition/flushing patterns	sediment loading increases since erosion accelerates	soil erosion increase, sediment-production will become more intermittent	as above	less sediment production than in A1	less soil erosion than now, land management practices focused on sustainability

The anticipation is based on the articulation of the IPCC-SRES scenarios A1 (global economy) and B2 (regional communities). The changes are entered in the table as narrative descriptions of plausible future development trajectories up to ~2030

whereas in the South these efforts would attempt to maximise retention of dwindling water resources. Increased urbanisation was thought to enhance the flashiness of flooding in the North. In the South, it concentrates along the coast and enhances salinisation of aquifers due to overuse and seawater intrusion. These pressures together would likely lead to a declining water quality, a reduced diversity in near-natural habitats available in river corridors for biota and a lower biodiversity of European stream networks at large. This is amplified because it is foreseen that the Water Framework Directive (WFD) will be implemented with less rigour in A1 than is presently foreseen, as derogations due to socio-economic needs are expected to be extensive. The patterns for the state indicators discussed generally followed this assumption (Table 5). The stronger environmental commitment of societies

Table 4 Anticipated changes in ecosystem pressures for a hypothetical Northern, Central and Southern European river basin

Scenario:	A1, relative to situation in 2009			B2, relative to 2009 and A1		
River basin:	North	Central	South	North	Central	South
(change in) water quantity: the river discharge pattern changes	more low and flashy flows in summers, more winter flooding	water shortage in summer, flood risk in winter and sometimes in summer	decrease in overall quantity, construction of new dams to regulate discharge	less extreme than A1 but changes will occur	see A1	decrease in overall quantity, natural flooding areas
Hydro-morphological changes: e.g. damming, embankment, groynes	more reservoirs, more navigation, rivers will become more managed, technological solutions for sediment management, damming may affect coastal protection, offset by positive isostasy in Northern Baltic	more storm water basins, reservoirs; less groundwater	more incision in valleys, lifetimes of existing dams decreases as a result of rapid sedimentation	less river regulation; softer engineering; more abstraction still needed but more reuse of water;	more natural rivers	small cisterns, increased reuse of water, reduction of <i>per capita</i> water use
erosion: of soils in the catchment or of banks along the stream	snowmelt events create more erosion; more gravel extraction means changes in erosive potential, sediment imbalance	erosion accelerates; more and higher sand banks	increase	similar to now	less erosion	decrease locally as a result of profound land use change
compaction of arable land	no clear change, depends upon land use change and practices relative to wet-dry cycles	more compaction, less groundwater	no compaction	improvement - better land management	less compaction than A1	less compaction than now
sealing of the catchment due to urbanisation	increasing sealing due to increased urbanisation, intensive in UK, expansive in Scandinavia	increases	localized sealing, especially in coastal zone and deltaic plains	less sealing due to green city approach	less urbanization, sealing	Patchy, local urban sprawl
OM decline in the soils of the river basin	decline in increased arable fields, but less impact than in South	decrease in OM, change in soil biodiversity	strong decrease	improvement due to better soil structure	less decline in OM than A1	limited decrease, because practices to protect the soil will be implemented
contamination (of...)	contaminants increased due to more intensive industry, nutrients and pesticides from farming increase, some offset by technology but not keeping up with increased use, not too much regulation of chemical development and use	more due to increased floods, increased industrial activities (including biofuels), increased run-off and liberal rules	health and safety concerns for individuals will drive policy, increase in industrial pollutants and wastewater, especially in coastal areas as a result of population growth, but decrease in nutrients and pesticides as a result of major decline in agricultural acreage	legacy remains but less inputs due to control and green technologies and products; still a broader range of needs must be fulfilled locally, emissions that cannot be eliminated will be sited to less vulnerable areas	stabilisation at current level	treatment of polluted water will increase, conservation of resources will become focal point
salinisation of catchment soils	not an issue	locally dangers of salinisation of groundwater	intrusion will increase, and the departure of agricultural activities will eventually result in a decrease of inland irrigation practices. But many soils will already be salinised as a result of prior salinisation	not an issue	less salinisation than in A1	reuse of water will increase probability localised salinisation, resulting in high concentrations
acidification of catchment soils	SO ₂ decreasing, will continue to decrease	no increase expected	not relevant in southern catchment	SO ₂ decreasing, will continue to decrease	same as A1	not relevant in southern catchment
(successful settlement of) invasive species	increased temperature and stress as well as globalisation will increase this pressure, also as native species are more stressed	global scenario means more invasive species	lots of invasive species as a result of high mobility	equal or more than now but less than A1; local approach will encourage native species	less than A1	very limited because of combination of awareness and limited mobility

The anticipation is based on the articulation of the IPCC-SRES scenarios A1 (global economy) and B2 (regional communities). The changes are entered in the table as narrative descriptions of plausible future development trajectories up to ~2030 (OM=organic matter)

foreseen in the B2 scenario leads to less extreme prospects in Northern, Central and Southern river basins for pressures as well as state indicators (Tables 4 and 5), though at a short-term cost of environmental investments.

The sensitivity of the indicators to contrasting scenarios along the North–South gradient of river basins was examined. Focus was on projected timescales of impacts and responses and on the scope for river basin managers (Fig. 2). Primarily because all scenarios project change, and the differences among scenarios lie in degree of change, only three pressures were considered to be highly sensitive to specific scenario parameters: (1) changes in the discharge patterns and extent of hydro-morphological engineering (both of which consistently correspond with contrasting governance styles and socio-economic models), (2) the distribution of

Table 5 Anticipated changes in ecosystem state indicators for a hypothetical Northern, Central and Southern European river basin

Scenario: River basin:	A1, relative to situation in 2009			B2, relative to 2009 and A1		
	North	Central	South	North	Central	South
concentrations of chemicals (any)	flood may remobilise historical contamination, market forces may reduce nutrient contamination; new emerging contaminants, lag in policy and management	up	flash flood events will remobilise historical contamination; less agricultural chemical inputs	floods may remobilise historical contamination, but less than in A1, legacy will be better managed; emerging contaminants unclear; better nutrient management on farms but farm intensification; green cities balance nutrients	stable or down	less chemical inputs, and fixation in the landscape due to management and little flow
trophic state (probability of a eutrophic condition: high nutrient availability leading to high productivity)	trends in nutrient management continue, but legacy remains; low productivity lakes may be vulnerable to minor loading increases	slight increase	decline in agricultural activity will result in oligotrophic states of upstream water bodies, but historic pollutants will cause longer term eutrophic state in downstream water bodies	trends in nutrient management continue, but legacy remains, fewer low productivity lakes with vulnerability to smaller increases	stable or down	local increase of eutrophic states and historic pollutants will remain in downstream water bodies
biota: composition of the communities of living organisms	shifts in communities due to climate change, some habitats may increase in biodiversity, net decrease in habitat diversity due to more intensive use, connectivity may be an issue for some communities	down	different species composition, loss of diversity, but there might be compensation as a result of invasives	more microhabitats but net habitat loss; connectivity enhanced; biodiversity shifts from south but more slowly than in A1	better	decline in diversity, but compensation as a result of landscape conservation
buffering capacity	soil buffering stable, estuarine decreasing, rivers improving	down	decrease as a result of erosion	same as A1	stable	small decrease, since practices to prevent loss of soils will be in effect
salinity (concentration of salts in the water)	not an issue	slight / local increase	increase, where water still is present	not an issue	stable	smaller increase than A1
suspended matter and sediment	sediment balance a problem	increase	increase due to erosion, during flash floods; more matter in less water	sediment balance more natural	small increase	strongly controlled by land use practices
water level (in surface waters)	more variable and unstable, more controlled but more abstraction, recharge will differ, more evapo-transpiration in summer, possibly more recharge in winter	more variable, flash floods	decrease and variability will increase; Groundwater table will decrease especially in coastal areas.	hydrographs more variable due to climatic forces but also due to soft engineering - more space for water puts land out of use	more variable but less floods than A1	decrease and control by land use will be less effective
morphology and pedology of the river basin	increased land use, changing landscape changes soil biogeochemistry and source patterns	change due to erosion	incision of valleys; soil quality and depth will decrease	increased recognition of role of riparian zone for ecosystem services so more balanced management; land-river-groundwater connections considered in land and water management	stable	soil quality will increase in places where agricultural activity will increase, local compensation of climate induced changes

The anticipation is based on the articulation of the IPCC-SRES scenarios A1 (global economy) and B2 (regional communities). The changes are entered in the table as narrative descriptions of plausible future development trajectories up to ~2030

new contaminants and (3) the fate of existing stocks of contaminants (i.e. historic contamination). The former two pressures were also expected to have immediate, short-term impacts on river systems and to be most easily within the influence of the river basin manager. Workshop participants were (cautiously) optimistic about the span of control of river basin managers in their conventional sector of influence, but less so outside it, e.g. on land use change. A similar assessment for state indicators (not further described here) was largely consistent with the patterns observed for pressures. Suspended sediment, concentrations of chemicals and the composition of plant and animal communities along the river basin were thought to be highly sensitive to scenario differences and were expected to respond rapidly. Only water quantity was thought to be within the immediate span of control of the managers.

The survey of uncertainty and gaps necessarily met with the bounds of the expertise represented in the workshop. These were notably felt to be present in the economic and demographic aspects of scenario articulation. The workshop participants categorised different types of uncertainty:

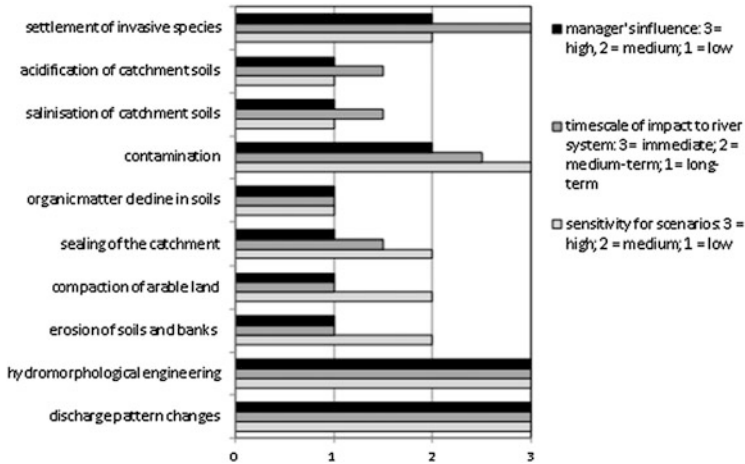


Fig. 2 Three-step ranking of the identified pressures exerted on the river system as a function of possible influence of the river basin manager, timescale of the response in the river system and sensitivity of a particular pressure to the difference between scenarios

1. Some aspects remain inherently uncertain, notably those related to the development and societal adoption of new lifestyles and different technologies across scenarios. Here the implementation of the WFD could be seen as a major experiment on how societal appreciation of environmental quality is translated into practice, with nations rather than river basins as replicates. Similarly, but at a different scale, the effect of invasive alien species (see [22], this volume) will be uncertain in principle. Awareness of the potential problem is common, but prediction and control remain inherently difficult beyond generic but simplistic trends.
2. For other aspects, understanding and availability of modelling tools is sufficient, but spatial resolution is insufficient or patchy across land use types. This is probably the case for precipitation and evapotranspiration modules of climate change models.
3. It was stated that a sufficient mechanistic understanding is probably still lacking, and existing models are crude approximations at best. Workshop participants identified the understanding of the dynamics of soil and sediment generation as such an area (see chapters by Négrel et al. [21] and by von der Ohe et al. [22], this volume). In the area of contaminant toxicology (see chapters by von der Ohe et al. [22], and by von der Ohe et al. [29], this volume), it appears difficult to keep up the pace with emerging pollutants, and scaling up from site to region, catchment and even river basin scale will remain highly uncertain. Also, whereas the WFD is seen as major achievement in Europe, participants noted that the science underlying the chosen indicators is patchy and evidence supporting the major monitoring efforts required will be increasingly challenged (compare [30, 31]).

4. Finally, it was generally felt that cross-disciplinary exchange and understanding (see also Section C in this book) still needs improvement, despite decades of attempts and experience in, e.g. systems analysis, integrated water resources management, integrated assessment and environmental economics. For example, the narrative depiction of contrasting lifestyles adopted by societies in scenarios is colourful and convincing, but causal links via land use to hydrology and water quality remain sketchy at best. Also more solid-integrated modelling efforts of larger projects than the present workshop (e.g. [8, 32, 33]) struggle with inter-disciplinary contrasts in world view and theoretical conceptualisations, data treatment and spatial perspective. The call for better or different inter- or multi-disciplinarity has been made frequently. However, it probably exceeds the absorption capacity of day-to-day practice and what training of managers and scientists can handle.

4 Discussion

The strength of exploratory scenarios is also their limitation: whereas every effort is made to be internally consistent and plausible, predictive power is inherently unknown [34]. Therefore, Berkhout et al. [35] stress the importance of sets of multiple and contrasting scenarios as collective images for learning about the future. It is this learning aspect that is important for well-informed river basin management and has been exercised in this workshop. A common awareness exists of the high probability of considerable changes in major drivers and determinants of water availability and quality in European river basins [28]. Despite the linearity in IPCC climate model responses over time ([24] see above), considerable uncertainty remains about the specific direction and magnitude of these changes in respective European river basins. Scenario articulation has been chosen here to chart plausible trajectories of change leading to vastly different societies inhabiting landscapes quite different from how they look now. Three parallel workshops have extrapolated the consequences of two IPCC-SRES scenarios, A1 (global economy) and B2 (regional communities), down to pressures leading to ecosystem state change in hypothetical Northern, Central and Southern European river basins and in their river networks.

The different institutional, social and economic fabrics of future European societies that were conceptualised using these two scenarios will have a major impact on the manner and extent to which society, and river managers, can adaptively respond to imminent climate change (Tables 2 and 3). Contrasts in land, resource and energy use as well as the orientation towards sustainability in governance will lead to distinct contrasts in water and sediment delivery to stream networks, in contaminant loads and their remobilisation and in opportunities for riparian biota to populate available habitat (Tables 3, 4, and 5). Available area of characteristic stream-accompanying habitat is expected to be greatly reduced under the A1 scenario, where engineered hydro-morphological alterations will be necessary to cope with reduced flow in the South and high or more variable flow

patterns in the North. The contrasts between A1 and B2 are most profound in Southern and Northern river basins, although the patterns are thought to be quite different. Central European river basins were expected to respond in an intermediate manner to scenario-based changes.

The workshop consensus that land in Northern catchments would be re-directed towards an expanding agriculture is not in line with findings of others [22, 36]. Still, the historical analysis of Skånes and Bunce [37] shows large-scale afforestation in previously open, agricultural landscapes in Southern Finland and Sweden in the twentieth century. Here ample space is available for agricultural expansion, given the foreseen improved climatic conditions. Busch [1] demonstrated that different scenario studies would allow for substantial declines as well as increases in agricultural land use across Europe, depending on assumptions of global trade, increases in agricultural productivity and areal needs for biofuel production. The re-use of set-aside land for agriculture thus may well be a plausible future option. To prevent major impacts on stream hydrology and water quality, cautious spatial planning (see [38], this volume) should accompany such schemes. Interference with present hydropower schemes may occur. For the South, uncertainty on the direction of land use change was profound, leading to quite different, sketchy but plausible trajectories.

The analysis of gaps primarily stressed the inherent uncertainty in the projection of societal change in scenario exercises, which is in line with Busch [1]. Adoption of altered lifestyles, transition to a carbon-neutral energy system or a nutrient-balanced, low-external-input agriculture can be charted as scenario elements. However, their wholesale assimilation in real societies over the coming decades remains hard to predict. This remained a continued note of caution throughout the workshop. The second issue is the continued struggle for a maturation of multi-disciplinary research. A fuller and broader integration of social and natural sciences at the scale of individual river basins is lagging behind compared to global-scale assessments such as the IPCC (see [5], this volume) and Millennium Ecosystem Assessment (see [39], this volume). Compared to these two aspects, the limitations in data or model availability may seem trivial.

5 Section B: Anticipating Change

This section has made a case for scenarios as a tool to frame and scope management decisions in the awareness of an uncertain future. Two modelling applications have demonstrated how geophysical and socio-economic system components may interact (chapter by Vermaat et al. [5], this volume). Apparently tangible risks—they are expressed as return periods—of floods and droughts have been offset against the uncertainty of economic development and governance styles (chapter by te Linde et al. [18], this volume). Small differences in land use change coupled to different IPCC-SRES scenarios, that are all too plausible in the near future, may well lead to drastic changes in drainage patterns and riverine sediment load over large stretches of Europe (chapter by Helmann and de Moel [23], this volume). The IPCC-SRES

scenarios appear to work well in downscaling exercises, whether in numerical or conceptual applications. It is fortunate that a new scenario generation [5], see also [18] builds on these scenarios and will not sacrifice these as benchmarks for future modelling and integrated assessment efforts.

The Dahlem workshop was less bound by modelling requirements of data and consistency. Still, the workshop participants managed to grasp the strong contrasts that will develop across the geographic width of Europe's river basins. Here, the workshop probably mimicked the situation of today's river basin managers. With some notion of risks, decisions have to be taken against considerable uncertainty on their future effects in the complexity of real-world river basins that respond to changing drivers and pressures at widely different spatial and temporal scales. Those pressures were identified that would be within the span of control of the river basin management and those that would differ highly among scenarios. It was argued that it is these that should be the focus of river basin management plans. It was also observed that uncertainty is present as much in the geophysical as it is in the societal dimensions of our river basins. Engineering, hydrology and ecology but also economics have gone far in their attempts to engage in the multidisciplinary approach needed to address the complexity of river basin management. The current frontline of multi- or interdisciplinarity lies deeper into social sciences. Balint et al. ([23] and see also Section C in this book) make a case for adaptive management, where stakeholder deliberation is an integral component. Such exercises should be forward looking and hence could benefit from the scenario approaches as described in this book section (Section B). The Dahlem workshop as described in this chapter can serve as evidence that such efforts not necessarily require data-heavy modelling preparations.

Finally, from the sobering notion that river basin management staff has a limited span of control, the Dahlem workshop did endeavour the following recommendations to serve as guidance in contemplating future-oriented actions, priorities and policy drafting:

- Prepare well in advance for changes in hydrology, now that options are still open.
- Near-future risks and probabilities are uncertain, scenario outcomes are uncertain too, but sketch a possible future bandwidth and enhance preparedness to think about future states of society, catchment and river.
- Consider means to influence critical drivers that are outside a river manager's reach but can be affected by national politicians, notably those affecting land use and soil management, such as national or EU agricultural policy.
- Step back from maintaining the status quo and seek new balances between our use of the land and what we want from rivers and their surrounding catchments.

Despite the obvious restrictions of the present qualitative scenario exploration, the Dahlem workshop sketched major differences among plausible future trajectories of European river basins. It included an articulation of highly specific state indicators. In view of the uncertainty facing river basin management, the two scenarios adopted here ("A1—global economy" and "B2—regional communities") could well reflect the margins of an envelope of plausible futures. A recent

assessment [3] suggests that scenarios have less impact if they identify threats for which there is no viable response. Thus, it is important to use these scenarios for formulation of strategies for action. The evaluation of various parameters' sensitivity to scenario conditions, timescale of impacts and susceptibility to river basin management may help in the identification and prioritisation of management actions and approaches. Risk-informed river management will include a focus on future change and hence adopt a scenario perspective similar to what this study argues for.

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Section C
Connecting to the Social System

Integrated River Basin Management and Risk Governance

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Abstract Water management is currently in a sustainable development phase, where pressures like changes in water quantity, erosion and contamination are being seen to interact in a complex pattern. In this phase it is necessary to take a holistic, integrated approach to environmental issues in order to meet sustainable development aspirations. Risk-related management decisions are challenged by complexity, uncertainty and ambiguity. Depending on the challenge, risk management must adapt to the specific circumstances and context. While simple risk problems can usually be addressed through routine-based procedures and measures

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(e.g. waste water treatment), complex risk situations (e.g. environmental quality of river basins) require new governance processes. These processes should be well informed, robust and participatory. Inspiring to this end is the risk governance framework developed by the International Risk Governance Council (IRGC). The use of the best available understanding on how river ecosystems function will resolve some of the complexity, reduce uncertainty and thus will improve river basin management. However, significant uncertainty will remain. This is intrinsic to the complexity of social/ecological systems. We need to accept and may learn to cope with this uncertainty by applying the concept of Adaptive Integrated Water Management (AWM). That AWM works in real-world practice is, for instance, demonstrated by how the Lonjsko Polje Nature Park (a floodplain area of the Sava River in Croatia) is managed to date. Risk-informed management incorporates all these elements as it involves the integrated application of the three key principles: be well informed, manage adaptively and take a participatory approach. The most appropriate scale for implementation of these key principles may be the catchment or landscape scale.

Keywords Adaptive Integrated Water Management • Governance • Integrated river basin management • Risk • Risk-informed management

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1 Water Management Under Transition

During the nineteenth and twentieth century, European societies changed from local communities based on agricultural production towards industrialisation, growing cities and urban populations with intensified productivity of primary production (agriculture, forestry and mining). Accordingly the challenges and development of water management have progressed through phases:

- A *sanitization phase* (eighteenth century up to 1950): when driven by public health problems, the emphasis was on clean water supplies and safe sewage disposal.
- A *pollution control phase* (1950s–1990s): when driven by a degradation of surface waters and aquatic ecosystems, the focus turned to water quality improvement through the control of polluting discharges (in particular point source pollution by urban and industrial waste water).
- To the present *sustainable development phase*, where pressures such as changes in water quantity, erosion and contamination may overlap and interact in complex patterns, and the growing realisation that a holistic approach to environmental issues is necessary to meet sustainable development aspirations through understanding the complexity of catchments and river basins as natural systems.

Traditional water management and its system paradigm during the first two phases have often been characterised as a ‘command-and-control’ approach [28]. Measures focused on engineering or mechanical solutions and the generally intensified land use required rivers to be controlled and their energy harnessed. This was implemented by changing their hydromorphology, for example, by canalising rivers and building dams. Similar approaches have been used to address water quality; the building of improved sewage works to treat effluents, for example, thus protecting the receptor (the river and its ecosystem) rather than dealing with the source—our consumption and the amount of waste disposed.

The choice and design of measures was often driven by implementing specific technical solutions—engineered measures perceived to have the advantage of being controlled systems with low inherent uncertainties. Furthermore the choice and design of measures was often done in isolation without considering the complexity of ecosystems or the interactions and trade-offs at different scales from sites to small catchments and up to entire river basins.

To face the challenges of integration and sustainability, the need for another transition in water management was recognised and, in Europe, was the driving force behind the Water Framework Directive (WFD see [9]; see also [32], this volume). In parallel policies and ideas on Integrated Water Resources Management (IWRM, see Fig. 1) have been developed and refined towards Adaptive Integrated Water Management (AWM, see Sect. 5). Both are management approaches that aim to define frameworks for participatory approaches in water management as well as how to deal with complexity, uncertainty and ambiguities (see Sect. 3) in river basin management (RBM).

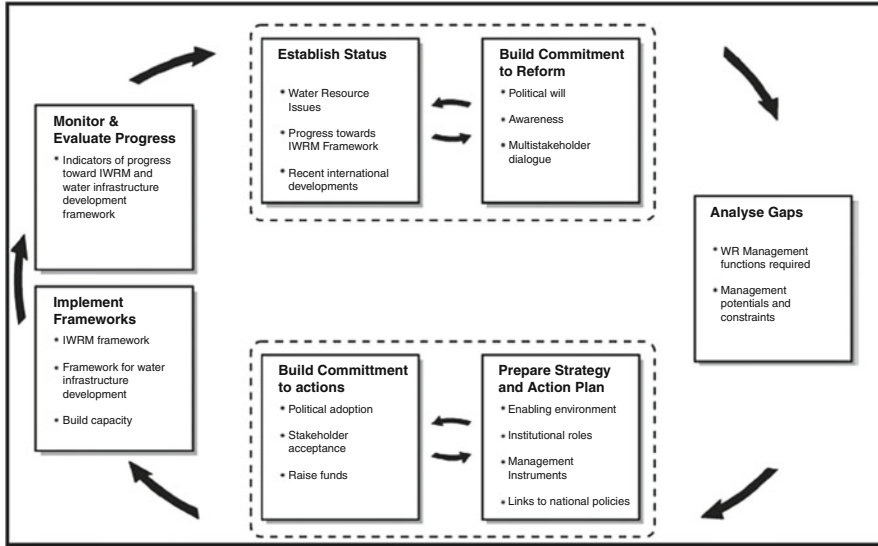


Fig. 1 Integrated water resources management [19]

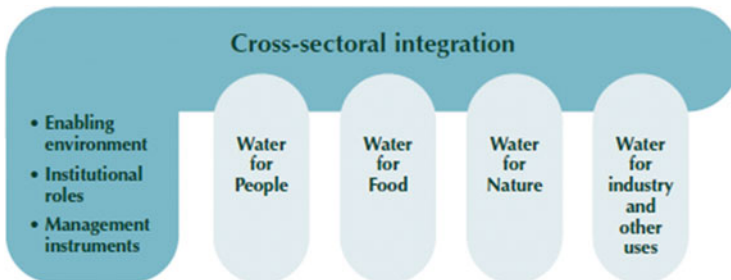


Fig. 2 IWRM and its relation to sub-sectors [12]

IWRM offers a framework for water governance and has been defined by the Technical Committee of the Global Water Partnership [12] as ‘A process which promotes the coordinated development and management of water, land and related resources, in order to maximise the resulting economic and social welfare, in an equitable manner without comprising the sustainability of vital ecosystems’. The IWRM framework emphasises the integration of management interests with the values of stakeholders involved in all sectors influenced by water management (see Fig. 2).

IWRM acknowledges the need for a process which allows involving relevant stakeholders and integrating different interests. In particular where stakeholders are not directly part of the administrative system, competent authorities have not got the resources and often also not the knowledge to successfully implement measures

[24]. Active involvement of central stakeholders is thus of utmost importance to river basin planning. However, organising the interaction with stakeholders and the general public not only at local scales but across river basins and catchments is a significant new challenge (see also [36], this volume).

Experience shows that integrating collaboration into government processes and cultures can only work when government agencies effectively participate in the local partnership processes and coordinate their work with other agencies. Where there are entrenched agency processes, cultural differences, staff without necessary skills or insufficient resources, then collaborative mechanisms may be severely impeded [14]. A review of the claims for IWRM and AWM against evidence from their implementation [23] suggested also that broader societal changes are needed to effectively translate research into practice, science into policy and ambition into achievement. Regarding the latter, the WFD Lille 2010 conference pointed out that now the time has come to ‘let actions speak louder than words’ [3]. Furthermore, this conference concluded that effective connection of research findings to practice and policy depends on four pillars, i.e. right timing, provision of context, linking of levels (EU/National/Regional in both directions) and connecting of people. Further considerations on ‘science-policy’ interfacing are addressed in [37], this volume.

2 Policy Drivers for River Basin Management in Europe

2.1 Water-Related Environmental Policies

Within the European Union, there is a staged process on environmental policy ranging from communications and strategies developed by the European Commission as directives, legislative acts of European Council and the European Parliament. Since 1990, there have been inter alia the Habitats Directive, the Water Framework Directive (WFD see: [9]), the Soil Thematic Strategy [5], the Groundwater Directive (GWD see: [10]) and the Floods Directive (FD see: [11]). These European Directives set the ideologies, priorities, timescales and reporting requirements for the management of our environment, but a much wider variety of European and national policies also contribute to environmental protection of the soil–water–sediment system within river basins (as illustrated by Fig. 3 and see also [32], this volume).

The WFD is the primary legislative driver in Europe that addresses river basin management and is an appropriate vehicle to develop water management towards its third sustainable phase. Its river basin planning process requires a broader and more integrated approach to environmental management. Furthermore the WFD has changed the overall policy context of water management and requires the management of water bodies as part of the biophysical system (i.e. the river ecosystem) at much larger scales than before. It therefore demands a reframing of administrative systems and an adaption of organisational arrangements to pay more

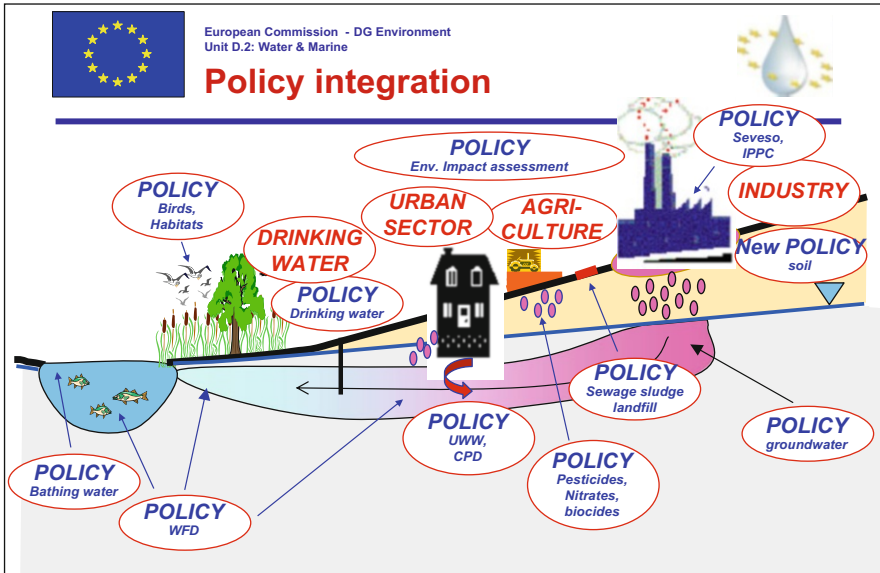


Fig. 3 Environmental management of river basins and policies [29]

attention to natural structures of catchments and river basins. This also provides a strategic driver for land management and, in some areas, land use change at a catchment scale. With its emphasis on cycles (see Fig. 3 in [32], this volume) of river basin management, the WFD is risk oriented and includes an 'ecosystem-centred' approach by defining environmental objectives for water bodies. Measures across whole river basins and catchments will have wider socio-economic consequences than regulatory bodies in most of the member states of the European Union have been previously used to. In the past there was a focus on the management (control) of point source pollution. Now the WFD places a greater emphasis on the wide-scale diffuse pressures across catchments and requires an understanding of the causes for conflicts of interest between catchment stakeholders.

Following the ideas of IWRM, the WFD demands public participation and greater stakeholder involvement. In turn this means that greater transparency in decision-making is required, particularly since larger scale change will bring greater impacts to more people. Therefore, a sound evidence base provided by a robust research environment will be crucial.

Under the WFD, the purpose of river basin management is to ensure the protection and improvement of the water environment using an integrated approach. The approach is—to some extent—also adaptive as every 6 years the river basin management plans are revised in order to improve the effectiveness of these plans (see further in [32], this volume).

Driven by the implementation timescales for establishing the first RBMPs by 2009, a best endeavours' approach has been adopted in many countries using

Table 1 Management spheres according to the transition management literature (van der Brugge and van Raak [31], based on Loorbach and van Raak [22])

Management sphere	Focus	Problem scope	Timescale
Strategic	Culture	Abstract/societal system	Long term (30 years)
Tactical	Structures	Institutions/regime	Midterm (5–15 years)
Operational	Practices	Concrete/project	Short term (0–5 years)

available data, knowledge, systems and structures. As a consequence and to bridge the gap to recent scientific findings, the second and third cycles of river basin planning will need to involve research to transmit new methodologies, models, concepts and approaches into practice.

2.2 *The Appropriate Scale for Management*

A particular challenge in river basin management is finding the right time as well as spatial scale for management.

The ‘window of opportunity’ for proposing and adoption of changes in the WFD river basin management plans is once every 6 years, as described in the previous section and in [32], this volume. Further considerations on the appropriate timescale for strategic, tactic or operational management actions are addressed in Table 1. Regarding timescales it is also important to note that management should integrate with and be timed to other planning efforts such as spatial planning (see [38], this volume). The intermediate and tactical scale (see Table 1) at which the change in culture and strategy induced by the WFD can be translated into changes in practice probably relates best to catchment scale.

However, addressing of the right time and spatial scales, within appropriate institutional settings and integrated with other planning efforts, is still far from common in Europe. A disconnect between the strategic level of a river basin with operational levels at local scales, misunderstandings or conflicts between top-down and bottom-up institutional processes may severely hamper management efforts. Procedures of WFD policy and implementation are well known; the new challenge is to better connect practice to strategy. This urges the need and importance of an intermediate and strategic level of governance at a catchment or landscape scale [30]. This level has several key roles to play:

- Translating national and regional policy and guidance to the people who are implementing activities at the operational level
- Coordinating funding for local activities and developing the so-called win-wins
- Synthesising learnings from local practice and feeding key messages upwards, ensuring that local-scale activities inform the development of new policy and practice

Furthermore, given European land management cultures, the catchment level also provides a good link to spatial planning, which is often well established in the urban and development context, but less so in relation to ‘softer’ rural land uses, such as agriculture and forestry. Finally, the catchment is a tangible and relevant scale for the coordination of knowledge related to the integrated environmental, social and economic system operating within its boundaries. It is recommended that administrations consider carefully how to establish better connections between policy and practice and organise information flow across scales. First practical experiences using the ecosystem services concept are promising (see [35], this volume). Social learning processes can benefit from this concept, as it seems to offer an easier ‘language’ to communicate stakeholder concern and to discover common interests and how to use the natural capacities of the land-water system in a complementary manner.

2.3 The Wider Policy Arena

Anyone involved to river basin management needs to be aware that the policy arena is much wider than the environment. Other policy areas (e.g. economy, industry and energy) may be overriding or creating new incentives that influence the socio-economic system faster and more significantly. High-level policy discussion, like the Europe 2020 reform agenda [8] - aiming for restoring economic growth for jobs and combating unemployment - set out priorities, trickle down through specific common as well as national policies and as a consequence frame decisions on land and water use within our catchments.

Within the European Union, the above-mentioned policy fields are major factors determining the development of employment and economic wealth, which in turn govern the production and consumption of goods and our use of natural resources. There are a number of current sustainable development policy initiatives which may affect river basin management in Europe. The proposed Environment Action Programme to 2020 [7] emphasises to protect, conserve and enhance the EU’s natural capital as well as turning into a resource-efficient, green and competitive low-carbon economy. This corresponds to the ‘20–20–20’ EU climate and energy targets:

- A reduction in EU greenhouse gas emissions of at least 20 % below 1990 levels
- 20 % of EU energy consumption to come from renewable resources
- A 20 % reduction in primary energy use compared with projected levels, to be achieved by improving energy efficiency

In particular the target of covering 20 % of energy consumption through renewable resources is a change in framing conditions and sets a driver which might have relevance for catchment management. For example, more hydropower plants to produce renewable energy will be in a direct conflict to the objectives of the WFD and a philosophy to restore the connectivity of rivers. Changing

agricultural production towards energy plants and biofuels might not be so immediately apparent, but there may be indirect linkages where there is widespread land use change. Cultivating soil differently or using different fertilisers and pesticides may change soil functions with respect to water storage and surface runoff as well as generating diffuse contamination of groundwater by new contaminants. In general the European Common Agricultural Policy (CAP) reform may affect catchments much more and within a shorter time frame than impacts due to climate change and relatively slow reaction natural systems. The strategy for urban environments is also an example of a policy which, by defining new directions on how to control and balance densely populated and intensively used areas, will affect environmental processes and functions within river basins.

Therefore, the challenge to catchment management is not only to understand the natural system (see Section A in this book) and possible future changes within it (see Section B) but also to follow policy debates in general and anticipate possible influential drivers that are likely to change framing and consequently decisions within social systems.

Finally, it is important to note that not all non-water policies are necessarily ‘threats’. Some may actually help to achieve the river basin management goals, for instance, financial or political support. Again this provides opportunities for some adaptive management, considering opportunities provided by other policies and being proactive in policy developments in other areas that may affect river basin management positively or negatively. A more rigorous integration of sectorial policies is needed. The ‘Blueprint to Safeguard Europe’s Water Resources’ ([6]; and see also [32], this volume) attempts to make a significant step here.

3 Risk Governance

3.1 *What Is Risk?*

Risks can be caused by natural phenomena (e.g. an earthquake, flooding) but can also be the result of mental ‘constructions’ [25] (e.g. perceived risks, based on beliefs). Risk accompanies change and changes may induce risks. Thus, it is a permanent and important part of development. The willingness and capacity of societies as well as individuals to take and accept risks is crucial for achieving economic development and introducing new technologies [16].

A technical definition reduces ‘risk’ to a function of the probability of occurrence and the extent of a negative consequence. But more generally speaking, risk is about uncertain consequences of an event or an activity in relation to something that humans value.

Risk itself is different from (actual) harm, which may occur as a consequence of risk and may vary according to the vulnerability of the affected system. Discussing risks involves discussing uncertainties and contingency, as well as beliefs that

human interventions may avoid, prevent, reduce or transfer harm and, consequently, that humans have the ability to design different futures. Thus, risk management is an everyday exercise within modern societies and embedded in a wider context of balancing social, environmental and economic aspects, which means aiming at sustainability and its three pillars (people, planet and profit).

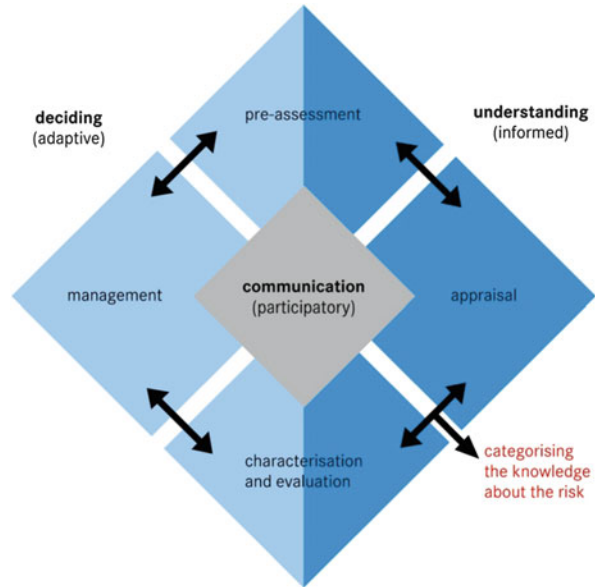
3.2 A Framework for Risk Governance

According to the International Risk Governance Council [17], governance refers to the actions, processes, traditions and institutions by which authority is exercised and decisions are taken and implemented. Risk governance deals with the identification, assessment, management and communication of risks in a broad context. It includes the totality of actors, rules, conventions, processes and mechanisms and is concerned with how relevant risk information is collected, analysed and communicated and how management decisions are taken. It applies the principles of good governance that include transparency, effectiveness and efficiency; accountability; strategic focus; sustainability; equity and fairness; respect for the rule of law and the need for the chosen solution to be politically and legally feasible as well as ethically and socially acceptable.

Many risks, and in particular those arising from emerging technologies, are accompanied by potential benefits and opportunities. Good risk governance should enable societies to benefit from change while minimising its negative consequences [17].

In its White Paper on Risk Governance [16], the IRGC has put forward an innovative and integrative framework of inclusive risk governance (see Fig. 4). ‘Inclusiveness’ is a major premise to applying the framework and concerns the obligation to ensure the early and meaningful involvement of all stakeholders and, in particular, civil society [16]. The process envisaged in the IRGC risk governance framework comprises four main phases: ‘pre-assessment’, ‘appraisal’, ‘characterisation and evaluation’ and ‘management’, as well as a central element: communication. The framework recommends that both a scientific risk assessment (a ‘conventional assessment’ of the risk’s factual, physical and measurable characteristics, including the probability of it happening) and a ‘concern assessment’ (a systematic analysis of the views that stakeholders or individuals may associate with a hazard) be conducted. Understanding the scientific or technical attributes of risk is equally important to understanding stakeholders’ concerns and socio-economic implications.

Fig. 4 Basic model of risk governance (adapted from [17])



3.3 Evaluation and Categorisation of Risk Problems

The appraisal of ‘risks’ against ‘concerns’ needs to be clearly differentiated from judging a risk’s acceptability. In contrast to some understanding and practice, it is important to recognise that ‘risk assessment’ does not produce the final outcome that precedes the management decision. ‘Evaluating’ the risk is often a delicate and controversial phase in the risk governance process and precedes the decision about risk management itself. Risk evaluation balances costs and benefits, assesses broader value- or interest-based issues and involves policy decisions. All relevant evidence for making an informed choice on tolerability or acceptability is collected.

The evaluation of risk is thus an important step in the risk governance process. It provides the necessary basis for deciding whether a risk is acceptable, tolerable or intolerable (see Fig. 5). Acceptable activities offer merits with negligible risks, so risk reduction is unnecessary. Tolerable activities, on the other hand, provide benefits yet require additional risk reduction efforts. Intolerable activities are prohibited or substituted. The latter means that the activity may be changed to an alternative one that is less risky.

The decision on a risk’s tolerability or acceptability is based on the evaluation of the risk. Furthermore, options for dealing with the risk are suggested from a scientific perspective. Scientific risk characterisation therefore establishes the risk profile, judges the seriousness of the risk and draws conclusions as to whether or not and what kinds of risk reduction measures are adequate.

Risk evaluation applies societal values and norms to the judgement on tolerability and acceptability. Risk evaluation also determines the need for risk reduction

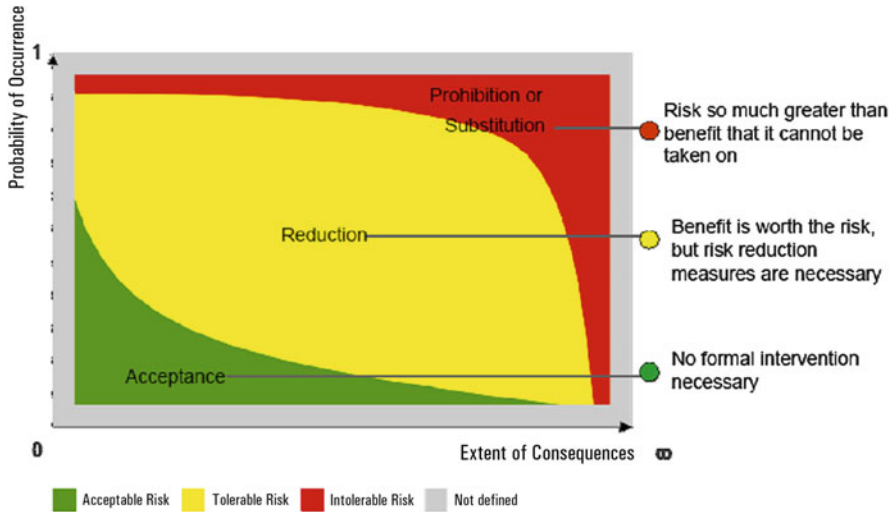


Fig. 5 Acceptable, tolerable and intolerable risks (adapted from [16])

measures. Thus, risk characterisation contributes the evidence-based component towards tolerability and acceptability judgement, whereas risk evaluation determines the value-based component.

The major challenges which are inherent to risk problems and need to be addressed by risk assessors and managers are due to complexity, uncertainty and ambiguity. Complexity refers to the difficulty of identifying and quantifying the relationship between cause and effect. Uncertainty is understood as a deficiency, something that is not known about the system (i.e. river basin), which could eventually be reduced or eliminated by pursuing more research or collecting more data. Such forms of uncertainty can be due to many different factors, for instance, errors in measurements, missing data and lack of understanding about the system's behaviour [4]. Ambiguity denotes the variability of interpretations of evidence. IRGC describes and categorises risks according to the knowledge about them and distinguishes between simple, complex, uncertain and ambiguous risks. It then proposes generic risk management strategies for each risk category.

3.4 Informed Decisions and Risk Management Strategies

Risk management needs to combine all the information gathered in previous steps in order to design and implement the actions and remedies required to avoid, reduce, transfer or retain the risk. Decision making includes identifying, generating and assessing options as well as evaluating and then selecting them. Implementation involves also monitoring and enforcing (including acting on the feedback from monitoring to ensure that the decisions have the desired effect).

Table 2 Risk characteristics and implications for risk management (adopted from [16])

Knowledge Challenge	Management Strategy	Appropriate Instruments
<i>'SIMPLE' RISKS</i> – Low complexity – Cause-effect chains known – Data available – Ambiguity low	<i>ROUTINE-BASED</i> – Judging acceptability / tolerability – Risk reduction	<i>'traditional' decision making</i> e.g. risk-benefit analysis – Technical standards – Economic incentives – Voluntary agreements
<i>COMPLEXITY</i> – High complexity – Cause-effect chains known – Data incomplete – Ambiguity low	<i>ROBUSTNESS-FOCUSED</i> <i>RISK-INFORMED</i>	<i>Improve buffer capacities</i> e.g. additional safety factors – Redundancy and diversity in designing safety devices <i>Characterise the available evidence</i> (expert consensus)
<i>UNCERTAINTY</i> – High complexity – Cause-effect chains known not fully understood – Data available – Ambiguity low	<i>PRECAUTION-BASED</i> <i>RESILIENCE-FOCUSED</i>	<i>Control hazard characteristics</i> tools include, e.g. containment, BAT (best available technology) <i>Improve capabilities to cope with surprises</i> , e.g. – Diversity of means – Avoiding high vulnerability – Preparedness for adaptation
<i>AMBIGUITY</i>	<i>DISCOURSE-BASED</i>	<i>Conflict resolution methods</i> for consensus or tolerance of management option selection – Stakeholder involvement – Emphasis on communication and social discourse

Risk management decisions are challenged by complexity, uncertainty and ambiguity. Depending on the challenge, risk management must be adopted to specific circumstances (see Table 2). While simple risk problems and risk reduction usually can be achieved with routine-based strategies, controlling complex risk situations requires establishing robustness. Strategies to address uncertainty must aim to precaution or seek resilience, i.e. seek to improve the capability to cope with surprises.

Of equal importance to making and implementing the decision is by whom it is made: while some decisions are legitimately made by regulatory bodies and other governmental agencies or departments in a top-down 'government' approach, others are best made through a bottom-up 'governance' approach.

3.5 Risk Communication

Risk communication is needed throughout the whole risk-handling chain, from the framing of the issue through the monitoring of risk management impacts. At each step in the chain, it requires the provision of adequate information to allow stakeholders to make informed choices. It also requires engaging into a dialogue

(see also [36], this volume). The precise type of communication needs to be adopted to the type of risk (simple risk problems may not need more than simple information, whether ambiguous risk problems requires larger societal debates). It also needs to be adopted to potential benefits and risks, i.e. the gains that some stakeholders might get as well as the negative impacts resulting from risks. In all cases, risk communication needs to address contextual issues. Communication has to be a means of ensuring that:

- Those who are central to risk framing, risk appraisal or risk management understand what is happening, how they are to be involved and, where appropriate, what their responsibilities are (internal risk communication).
- Others outside the immediate risk appraisal or risk management process are informed and engaged (external risk communication).
- Effectiveness of the risk governance process is improved, and trust is built between the stakeholders.

4 Risk: Broadening the Notion

Addressing risk in a changing world and at levels ranging from ‘my backyard’ to the ‘global village’ requires the development of a broad perspective to enable complex systems involving non-linear and indirect linkages within and between systems. Our society faces challenges related to climate and socio-economic changes (see Section B in this book), and this is true also for river basin management. Our understanding of risk and the assumptions behind are questioned. Therefore, some additional reflections to those exposed above—i.e. about risk as a concept (Sect. 3.1) and its use in environmental policies (Sect. 2.1)—are necessary, in order to discuss how our societies may frame ‘water risks’ in the future.

4.1 Risk and Concerns

Nowadays environmental policies have to balance multiple requirements. Besides classical concerns like flooding and landslides, societal desires to have more natural rivers and future concerns, such as biodiversity decline and coping with climate change, need to be accommodated. As a consequence, managing risks within river basins needs a different perspective from the classical technological approach. The management framework should not only be ‘ecosystem centred’ but balance societal needs against the capability of river ecosystems to deliver ecosystem services (see [35], this volume). Future river basin management will need to address threats (e.g. erosion, contamination, etc.) to the ecosystem and how these threats lead to real and perceived adverse effects (e.g. harm, losses: see Section A in this book).

The adverse effects we see from floods and landslides are largely risks to people and property in the classical sense. We are now observing biodiversity decline, loss in ecosystem services delivery capacity and loss of resilience in our water systems to cope with climate change. Policy objectives of the WFD for good ecological and chemical status of our waters are not merely ‘nice to have’; they are motivated by our general long-term concerns about ‘planet earth’ which is beginning to find a more coherent language in the ecosystem services concept (see [35], this volume) that is increasingly discussed in a wider public arena.

Managing ecosystems or biophysical systems in general requires goal-driven management options. Thus, aiming towards sustainable development of our societies and sustainable use of natural resources might require going beyond conventions and the specific policy objectives of different sectors towards balancing our needs. This is often referred to as an ecosystem approach.

There is a need for a holistic social/ecological systems approach (see [32], this volume), where threats to natural resources as well as concerns regarding human well-being need to be matched. Applying this could enable to deal with issues that cannot be immediately assessed as ‘probabilities of an adverse effect’. This recognises that certain things cannot be fully scientifically ‘assessed’ but need to be assessed for the perceptions, fears or interests that stakeholders have about them (is the ‘concern assessment’ of IRGC framework). This finally requires a clear separation between scientific questions and societal challenges and calls for transparency on any goals and objectives to prepare decisions.

4.2 A Risk-Informed Approach Towards IWRM

The variety of risks and linkages in river basins when considered as one integrated ecosystem is highly complex, and understanding them is difficult. Following the framework as provided in the chapter by Brils, Barceló et al. [32], this volume, having the risk ‘source-pathway-receptor’ linkage at its core (see Fig. 2 in that chapter) and emphasising the simplified perspectives developed by the Resilience Alliance (see Fig. 4 in [32], this volume), the interaction within a close-knit social/ecological system can be differentiated into the following:

- Direct risks to the social system and human well-being caused by the ecosystem (e.g. landslides, flooding, human health diseases)
- Direct risks to soil, sediment and water as natural resources caused by the anthropogenic sphere, which can generally be understood as stressors (e.g. erosion, pollution; see [33, 34], this volume) resulting from human activities
- Indirect risks linked to the social system as natural resources are depleted and the processes and functions of ecosystems are affected
- The risks resulting from trade-offs between regions, e.g. both spatial (e.g. an upstream stressor is propagated downstream and there poses a risk) and temporal (e.g. remobilization of old, historic contamination by flood and then posing a risk)

From a river basin management perspective, e.g. flooding risks or the risks of landslides would qualify as risks in the classical technological definition of the term. However, due to climate, land use and hydromorphological changes, river basins are less static but increasingly dynamic systems. References expressed by probability distributions derived from past events, that assume simple statistical relationships, may therefore no longer be appropriate as natural river basins may have changed into intensively cultivated (or landscaped) ecological systems governed by complex interactions.

Most approaches developed to characterise risks with regard to the objectives of the Water Framework Directive and the concepts of classifying water bodies into specific chemical, ecological and quantitative status are caught between describing the current situation as well as anticipating environmental quality in future.

So what do we actually mean by risk-informed approaches? A risk-informed approach in current environmental thinking can be considered as an evidence-based approach, which is open to new scientific knowledge, open to public discussion and not based on political and perhaps even other ideological preconceptions. In short, it is transparent decision-making based on scientific evidence with an adequate level of precaution in view of uncertainties.

4.3 Managing Risk and Concerns

IWRM in practice will be a valid and risk-informed approach as long as we realise that we are actually talking about (scientific) evidence-based approaches, which are open to public (stakeholders) debate. In terms of management, the classification of risks in complex, uncertain and ambiguous risks (see Table 2) may also be useful for water managers who are organising stakeholder involvement (see also [36], this volume). IRGC offers a flexible framework for facing the generic challenges of involving stakeholders (see Fig. 6). It offers different routes depending on the categorization of a risk, from simple to complex, uncertain and ambiguous.

Referring to the IRGC escalator and assuming that integrated catchment management is highly complex where uncertainties will always prevail, scientists should be more involved as experts to inform and interact with stakeholders and the interested public.

Simple risk problems at the operational level (see Table 1) of a project or on an individual farm or an industrial site will not generally require scientific support but stay within routine-based management strategies (see Table 2). However, as foreseen by the WFD, strategic questions need to be resolved at the catchment or river basin level and should undergo discussions involving scientists and the wider public.

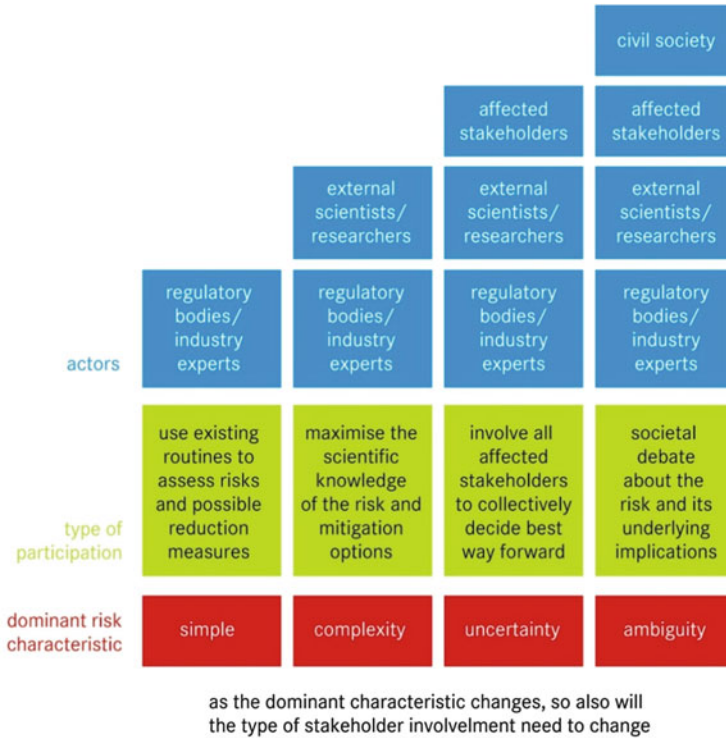


Fig. 6 A structure for stakeholder involvement [17]

5 Adaptive Integrated Water Management

5.1 What Is It?

IWRM challenges water managers with its demands for integration across sectors, scales and interests. In addition to this, Adaptive Integrated Water Management (AWM) puts a stronger focus on the need for learning as part of the management process considering the socioecological system as being under transition. AWM seeks to increase the adaptive capacity of the water system by systematic approach of testing and learning [26]. The project ‘New Approaches to adaptive Water Management under Uncertainty’ (NeWater)¹ 2004–2009 was an integrated project funded by the Global Change and Ecosystem Priority of the 6th European Commission Framework Programme [26]. NeWater developed the concept of AWM further in order to provide more specific guidance to (water) managers as well as to further develop the theory. Based on the AWM framework, case studies were

¹ See also: <http://www.newater.uni-osnabrueck.de/>

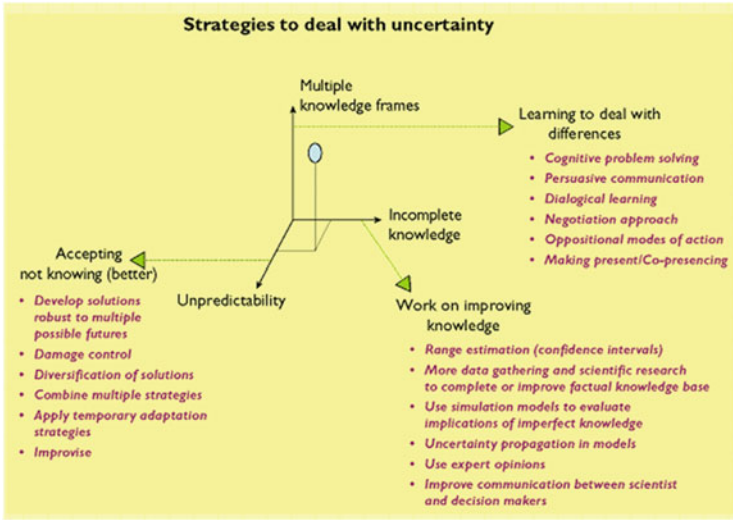


Fig. 7 Strategies to deal with uncertainty [4]

carried out in seven major river basins to test approaches and tools for practical applications.

In the following, two core aspects of the concept, approaching uncertainty and learning in governance structures, will be focused at building mainly on results from the NeWater project.

In NeWater three types of uncertainty were identified: unpredictability, incomplete knowledge and multiple knowledge frames. In most water management problems, all of these types of uncertainty are present [4]. Although water managers are well aware of the different kinds of uncertainty, in general, no explicit structured approaches have been used to deal with them. Isendahl [18] summarised that uncertainties have been dealt with using intuition or based on earlier experience. Also, uncertainties are framed rather diverse, depending on the individual actors, his/her role and the context. Still, for developing strategies dealing with uncertainties, the type or source of uncertainty is central (see Fig. 7). If, e.g. there is the problem of unpredictability, a manager is required to accept this knowledge gap and develop solutions robust enough to different developments. For example, if climate change makes it unclear whether more droughts or more floods are expected, then measures such as wetlands which keep floods and store water for droughts would help in both developments (see also [27]).

In AWM, the main approach towards uncertainty is to ‘create capacity, through learning and adaptation, for responding flexibly and effectively to changing and unknown conditions’ [4]. In the context of AWM, learning has been considered as a central benefit of stakeholder involvement. This social learning has been defined as ‘learning together to manage together’ [15]. Depending on the challenges, different levels of learning are necessary. They have been distinguished as single-loop, double-loop or even triple-loop learning [1, 2, 20, 21]:

- Single-loop learning leads to improvement of practices. It refers to learning *new skills and capabilities* through incremental improvement, doing something better without examining or challenging underlying beliefs and assumptions.
- Double-loop learning [1] has happened if underlying assumptions for action (e.g. agricultural practice is central to nitrate reduction in the Northern Sea) are changed. It changes the problem-solving strategies of the actors [21]. For sustainable development, double-loop learning has been considered vital to achieve the necessary changes.
- Triple-loop learning [20] is expected to induce changes of the background to the underlying assumptions (e.g. ecosystem services may not change in a sustainable world).

Still, social learning is essentially a natural process in which people have to engage themselves. It can be fostered with tools and appropriate process design. Crucial for learning in AWM is that monitoring and evaluation get implemented so that management actions can be reflected on and accordingly revised. The results of monitoring and evaluation are then used to fine-tune or retune policies and practices as the situation changes over time. This also requires the developing solution to have a broad stakeholder participation and negotiation.

Case studies on how to integrate participatory approaches and a broad knowledge base, also supported by eco-hydrological modelling approaches, have been provided by the NeWater project. A typical example of competing ecological and socio-economic interests in the implementation of the WFD was the Upper Guadiana, where the full recovery of groundwater levels is hampered by abstractions and conflicts regarding the restoration of wetlands. In the Elbe basin, options and measures for a climate change adaptation strategy were explored through scenario analysis on climate and land use change. Another inspiring example of AWM in practice is provided in the following section. All these examples show very encouraging results and may thus inspire others to proceed in this direction.

5.2 An Inspiring Example from Practice: Lonjsko Polje Floodplain Management

The Lonjsko Polje is a floodplain area in the Sava River basin in Croatia. It is a highly dynamic ecosystem which is flooded several times annually. By true AWM, the park is managed sustainably in conditions of change and unpredictability [13]. This is further explained hereafter.

The most conventional concept of flood control was embodied in the construction of dikes along rivers wherever floods were causing problems. Under spatial planning schemes, the formerly flooded areas seem safe and ready for agriculture or any other land development. However, recurring floods of recently developed land during recent decades combined with possible climate change impacts and the

accentuation of seasonal effects (e.g. winter flooding and summer droughts) called for rethinking of general approaches.

In the central Sava River basin (Central Posavina, Croatia), a different concept of flood management was designed by the Water Management Plan for the Sava River basin in 1972 already. Although it was primarily planned, in the framework of this project, to reduce the existing natural retention areas by almost six times, the real innovation in flood management was the fact that an area of some 50,000 hectares was to remain a retention capable of being brought into play for flood management. Whereas it had been difficult at this early stage to convince international donors like the World Bank, at the end of the millennium, an Environmental Impact Assessment (EIA) was a mandatory prerequisite to get a new loan. The conclusions of the EIA confirmed requests to consider a flood management system for the entire area of central Sava basin and rejected the constructions of further channels, dikes, dams and floodgates, except where absolutely necessary. One can state that the approach chosen in the case of the central Sava River basin became according to the phases in water management mentioned in Sect. 1 one of the earliest showcases globally where natural retention areas have been used in flood management deliberately and on large scale.

At the same time, almost the entire floodplain of Central Posavina is under nature protection. It is the challenge of both water and conservation management to manage sustainability under conditions of change and unpredictability in a protected area that contains the natural features of a dynamic ecosystem as well as the cultural features of an organically evolved landscape. According to Gugic [13], the pillars of managing sustainability under conditions of change and unpredictability are a sound understanding of key ecological and cultural processes, appropriate concepts of zoning and serial sites as well as integrity and dynamic authenticity of the property. In a dynamic ecosystem like floodplains, dynamic processes result in high ecological complexity and a permanent ongoing change of discontinuous spatial and temporal patterns. The emphasis to understand ecological processes enabled the Lonjsko Polje Nature Park Management to prepare decisions that are more transparent, comprehensible and versatile to stakeholders, the broader public but also the park staff. Zoning and integrity are common concepts in nature conservation but should not ask to freeze ecosystems to a static status. Related to a sound understanding of key ecological processes, these concepts are also appropriate to allow and create flexibility and versatility for managing land uses complementary to conservation aspirations. The concept of serial sites is an upcoming approach improving integrity by connecting habitats through spatial distribution.

Developing a common vision was the most crucial step to integrate demands on flood protection, nature protection, land use and economic wealth in the region of Central Posavina, particularly in Lonjsko Polje Nature Park. Participation was established at different levels at the one hand through a stakeholder committee but as well at a strategic level of the central Sava River basin, to connect two structures of governance of the next higher level.

Experience proved that it is necessary to repeat the process of visioning in an iterative manner, where every feedback makes the vision more accurate, complete

and accepted. During this process and as the nature park was nominated as a World Heritage Site, a comparative analysis was required. Comparison and broader information helped not only stakeholders but also the park management to discover the particular value of the landscape and wetlands.

The new approaches (informed, participatory and adaptive) implemented at the Lonjsko Polje Nature Park have helped demonstrate fundamental benefits and consequences. Besides avoiding unsuitable land uses in areas of inundation, the integrity of floodplain ecosystems can be maintained and allows the continuation of key ecological processes. In terms of retention, it provides a sufficient scope in water management, equally successful with regard to droughts and flooding. It is a low-cost concept and allows local people to continue with traditional land uses.

6 Conclusions

Arriving at a sustainable development phase of water management requires a holistic, integrated approach to environmental issues in order to meet sustainable development aspirations. Risk management decisions are challenged by complexity, uncertainty and ambiguity. While simple risk problems can usually be addressed through routine-based procedures and measures, complex risk situations require new governance processes. These processes should be well informed, robust and participatory. Inspiring to this end is the risk governance framework developed by the International Risk Governance Council (IRGC) that is described in this chapter. The use of the best available understanding on how river ecosystems function (see Section A in this book) and may respond to global change (see Section B) will resolve some of the complexity, reduce uncertainty and thus will certainly improve river basin management. However, significant uncertainty will remain. This is intrinsic to the complexity of social/ecological systems (see [32], this volume). We need to accept and may learn to cope with this uncertainty by applying the concept of Adaptive Integrative Water Management (AWM) as explained in this chapter. The way how the Lonjsko Polje floodplain is managed to date is just one clear-cut example that AWM works in real-world practice.

Risk-informed management incorporates all these elements as it involves the integrated application of the three key principles: be well informed, manage adaptively and take a participatory approach (see [32], this volume). A key challenge will be to implement these key principles at the right scale, within appropriate institutional settings and integrated with other planning efforts. As argued in this chapter, most appropriate might be the catchment or landscape scale.

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Ecosystem Services and River Basin Management

Kate A. Brauman, Suzanne van der Meulen, and Jos Brils

Abstract Ecosystem services are the goods and services provided by ecosystems that maintain and improve human well-being. This framework is inherently anthropocentric, organizing ecological processes by their effects on human beneficiaries and explicitly connecting ecosystem processes to human welfare. The ecosystem services approach facilitates management of a complex system by incorporating important aspects of risk-informed management. Ecosystem services provide a framework for assessing the many stressors to and outputs of a river basin by organizing stakeholders, identifying those whose actions affect the provision of ecosystem services and those whose well-being will be impacted by changes in the provision of ecosystem services, and by delineating the mechanisms by which land-use changes affect stakeholders. The framework is best used to assess changes in ecosystem service delivery and can be useful for considering the trade-offs among services provided by a variety of plausible future land-use scenarios. Within the ecosystem services framework, stakeholder beneficiaries identify the ecosystem services they find valuable. Ecosystem service valuation is not necessarily monetary valuation, but using money as a common currency for comparison can aid decision-making. Economic valuation approaches have a number of problems and pitfalls, but when thoughtfully applied, they allow the possibility of payments for ecosystem services, which can be a useful tool when command-and-control regulation does not work or is politically infeasible. Overall, ecosystem services are a useful tool for river basin managers because they provide a coherent context to incorporate stakeholders and complex biophysical processes into a consistent, learning-based management scheme.

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Keywords Common language • Economic valuation • Ecosystem services • Hydrologic services • Land-use change • Payment • River basin management • Watershed

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1 Introduction

As the authors have emphasized throughout this book, meeting the ambitious goals of the Water Framework Directive (WFD, see [1], this volume) to bring water bodies to good chemical and ecological status will require informed decision-making, adaptive management, and a participatory approach. This chapter addresses the ways in which ecosystem services may provide a powerful yet flexible framework for applying all three principles to a complex social-ecological system characterized by uncertainties in both spheres. The basics of ecosystem services are defined in Sect. 2, and Sect. 3 illustrates how ecosystem services can be used to assess status, trends, and the value of changes in river basins.

The integration of ecosystem services with risk assessment is described in Sect. 4. Section 5 is devoted to examples of practical experiences applying ecosystems services in water management. Finally, Sect. 6 closes this chapter with a brief summary and conclusions.

2 Ecosystem Services Basics

2.1 *Defining Ecosystem Services*

Ecosystems consist of complex and evolving interactions of living organisms and the abiotic environment in which they live, each making use of the production and waste products of others, creating dynamic cycles of nutrients and energy [2]. Humans have always relied on these cycles, gathering and eventually cultivating materials for food, fuel, shelter, and all the other goods and services necessary for well-being [3]. Human ability to supplant and shape natural processes has improved, but people continue to depend on the goods and services supplied by ecosystems [4]. Consider an artificial world, an aquarium, as an example. Filled with rocks and fish, it will thrive for only a short time. To maintain it, the fish must be fed and their waste removed, the water oxygenated, and the glass kept free from algae so the fish inside can be seen and enjoyed. On the scale of an aquarium, these manipulations are possible, but as horizons expand to a field, a watershed, or the whole planet, the limits of technology become more acute and human dependence on earth's life support systems becomes more apparent. The ecosystem services framework clarifies the connections between human well-being and ecosystem function.

Ecosystem services are the goods and services provided by ecosystems that maintain and improve human well-being [5]. For example, people depend on the functioning of soil microbes to maintain soil fertility to grow food crops as well as on the growth of trees to sequester carbon and regulate climate. The ecosystem services framework is inherently anthropocentric, organizing ecological processes by their effects on human beneficiaries [6]. By tying ecosystem function to human well-being and human actions back to ecosystem function, the ecosystem services framework creates a feedback loop that serves to promote both ecosystem and human well-being. In this context, it is apparent that risks to natural resources like soil and water have direct consequences for people [7].

Ecosystem services provide a framework to organize consideration of the ecological processes that affect human lives. The argument is not that processes—plant growth and absorption of nutrients, bird and bat predation on insects, soil formation, pollination by bees, and thousands more—occur for human benefit, it is that people benefit from them. If human actions affect ecosystems so that these goods and services are no longer produced, human well-being suffers in turn [5].

Stakeholders, the beneficiaries of ecosystem services and those who own and manage landscapes that produce them (e.g., in their roles as water managers, land-use managers, or farmers) play a key role in ecosystem service analysis, as illustrated in Fig. 1. They identify the services they receive from a water body

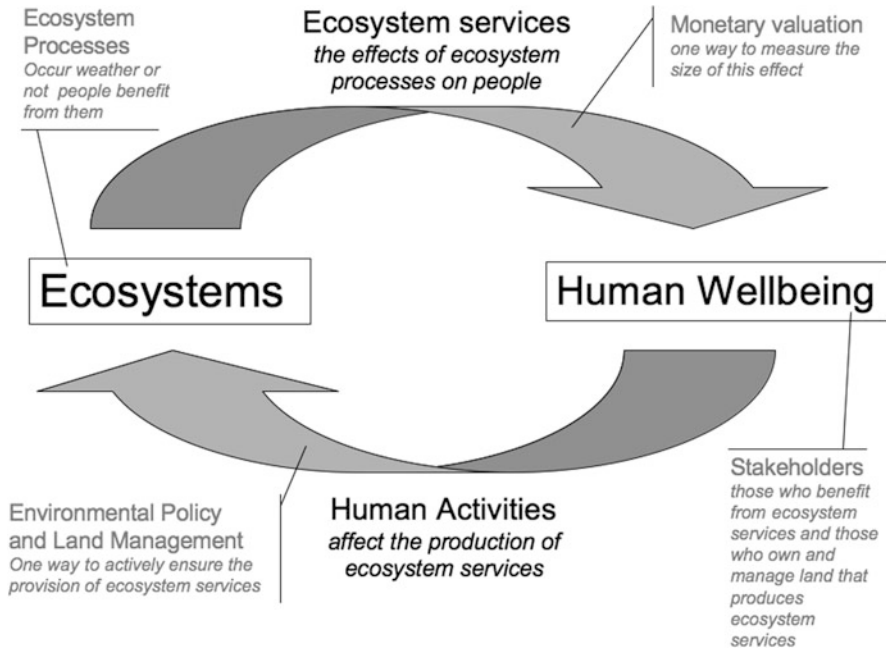


Fig. 1 Ecosystem services can be used as an organizing framework to consider the relationship between stakeholder beneficiaries and ecosystems

and its catchment. This includes services that are considered derogations under the Water Framework Directive, including water extraction for drinking or irrigation. By identifying individuals and groups who benefit from each service and who will be affected if land use and land cover changes, ecosystem services make it possible to assess whether those groups could be provided with services in alternative ways or if different groups will be differentially impacted. Doing so equitably, however, requires care in identifying and including the full suite of beneficiaries [8]. In an era of government regulation often based on cost-benefit analysis, ecosystem services potentially provide a wider lens through which to assess a range of costs and benefits to a diverse set of people [9]. In addition, this explicit linkage may make it possible to establish projects in which beneficiaries of ecosystem services pay upstream residents for their provision [10].

2.2 Ecosystem Services Typology

There are many ways to organize ecosystem services, and categories may need to be refined for specific projects [11]. The Millennium Ecosystem Assessment [5] has divided ecosystem services into four main categories: provisioning, regulating, cultural, and supporting. This categorization, and how ecosystem services link to human well-being, is illustrated in Fig. 2.

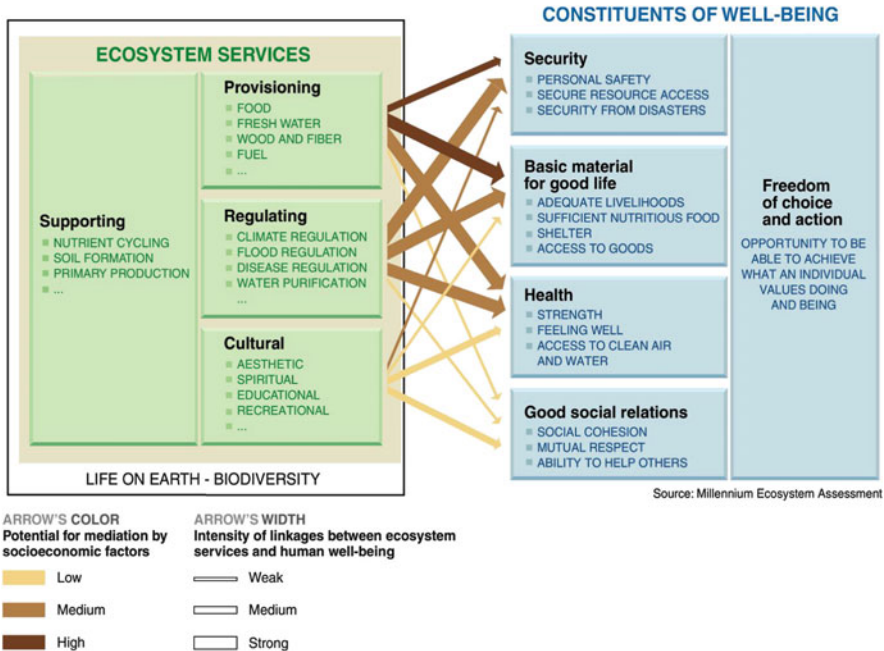


Fig. 2 The Millennium Ecosystem Assessment provides a framework for organizing and categorizing ecosystem services (Figure used by permission of Island Press)

Provisioning services are goods that ecosystems produce and people use. This includes, among others, food crops and biofuels, fish that humans catch for food, forest nuts and berries gathered for food, trees used for timber, and herbs used for medicine. *Regulating services* describe the way ecosystems affect the flow and functioning of larger systems which in turn impact human lives. This includes, among others, vegetation that affects the rate at which rainfall infiltrates into aquifers and flows into rivers, which then influences the timing of water availability; strips of vegetation next to waterways that absorb pollutants, providing people living downstream with improved water quality; erosion prevention by certain types of vegetation to maintain arable land; forests that sequester carbon, providing people worldwide with climate regulation; and trees planted next to highways that remove particulate pollution from the air, providing nearby residents with lower asthma rates and improved health. *Cultural services* describe nonmaterial enjoyment and use of the environment. Cultural services include recreation: activities such as rafting, hiking, and fishing are made possible and more enjoyable by woodlands and waterways that have certain characteristics. It also includes spiritual uses: people wish to be “out in nature” for specific spiritual reasons or for general aesthetic reasons. *Supporting services* are the background processes that people do not use directly but which sustain other services. For example, agriculture depends on nutrient cycling and soil formation, and productive fisheries require specific nutrient and sediment loads in rivers and estuaries that are controlled in part by vegetation reducing stream-bank erosion.

In the context of river basin management, water-related ecosystem services are particularly important. Services produced within a water body, called aquatic ecosystem services, are distinct from those terrestrial processes that affect water, called hydrologic services. Aquatic services include maintenance of fisheries or in-stream pollution assimilation [12]. Hydrologic services describe the way ecosystems within a catchment affect the quantity, quality, location, and timing of water that moves through those ecosystems [13]. Water supply for drinking or for transportation, safety from flooding, and spiritual and esthetic enjoyment of a river, all of these are hydrologic ecosystem services, the water-related goods and services that make people's lives both possible and fulfilling. Hydrologic services provide a lens to translate traditional hydrologic research into the policy world. They also provide a way to identify the most important hydrologic questions for river basin management, thus providing guidelines for future research.

2.3 Biodiversity and Ecosystem Services

Ecosystem services are often invoked to explain the importance of biodiversity (e.g., [14, 15]). While specific processes are often not well described, some new work suggests that increased biodiversity increases ecosystem service provision (e.g., [16, 17]). In addition, cultural ecosystem services enable biodiversity to be regarded as an ecosystem service itself by accounting for the enjoyment people derive from biodiversity or the intrinsic value they attach to it, though in general reports of increased well-being are attached to perception of biodiversity, not to objective measures of biodiversity [18]. Some studies have shown that ecosystem services projects increase biodiversity (e.g., [19, 20]), but, in general, biodiversity conservation and production of ecosystem services have variable overlap [21, 22].

3 Assessing Ecosystem Services in River Basins

3.1 Assessing Ecosystem Services Production and Changes in Their Delivery

3.1.1 Landscape Effects on Water

Because RISKBASE focuses on river basin management, we briefly address the ways that landscapes affect water. Terrestrial ecosystems play a key role in the production of hydrologic services (see Sect. 2.2) because the quantity, quality, location, and timing of water is affected as it moves across the landscape before it comes into contact with users [13]. There are simultaneous, and sometimes conflicting, effects of the landscape on water. Some hydrologic services are closely tied to specific attributes. When water is used for domestic purposes, water quality may be most important; when people are concerned about flooding, timing is the key issue. However, users have requirements in all four attributes—quantity,

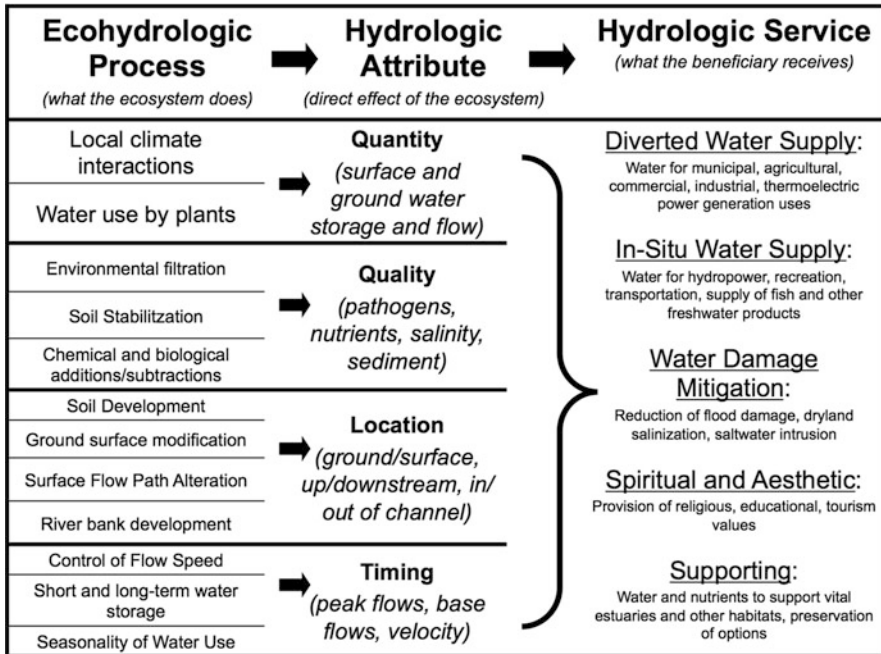


Fig. 3 Ecohydrologic processes can be organized by the hydrologic attribute they affect. Each attribute is related to the hydrologic service that beneficiaries receive. Figure originally published in Brauman et al. [13]

quality, location, and timing—for each hydrologic service, and those requirements may conflict; greater water volume is good for water supply but bad for flooding. As illustrated in Fig. 3, any ecosystem process that affects an attribute, such as local climate interactions affecting water quantity, will affect each service, from reduced availability of drinking water to decreased flood peaks [23]. The way an ecosystem affects each attribute, and thereby each service, depends on the ecosystem type, the quality or health of that ecosystem, and how it is spatially related to other ecosystems and to service users [24].

3.1.2 Whole-System Analysis

No ecosystem services assessment is complete without consideration of the full suite of services that may be delivered by a parcel of land, including carbon sequestration, pollination, habitat for waterfowl, and myriad other services [25]. One of the most powerful aspects of the ecosystem services framework is that it can highlight the full range of services produced by any given parcel. These might be thought of as the co-benefits of the primary service of interest, in this case, water. In one case study in Hawaii, however, Brauman [26] found that the provision of a variety of other ecosystem services provided by a watershed was more valuable than services related to water.

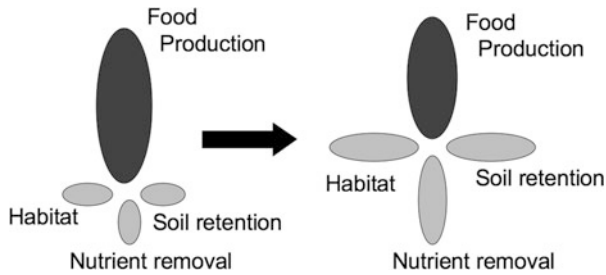


Fig. 4 When the values of a variety of ecosystem services are recognized and accounted for, management choices can be made that increase the aggregate value of the outputs of an ecosystem

By using the ecosystem services framework, the consequences of land management can be anticipated, and unexpected, unwanted outcomes can be reduced. This is particularly important when, due to government mandate or economic incentive, management to optimize one service is likely. For example, while a particularly fast-growing type of tree might be chosen for a forest plantation when the timber yield at 30 years is the only criterion, consideration of downstream water yield may lead instead to the choice of a tree species that uses water more efficiently but grows somewhat more slowly [27]. Historically, agricultural lands have been managed with crop yield as the sole management goal. As illustrated in Fig. 4, consideration of management effects on nutrient load in runoff, on carbon sequestration in soils, and on habitat in buffers between fields, and an explicit accounting of the value of those other services, will likely indicate that management regimes taking these services into account produce greater overall value.

3.1.3 Marginal Impact Assessment

Ecosystem service assessments are most useful in the context of land-use change—providing answers to questions about changes in the delivery of goods and services given changes in a landscape such as a forest harvested, a meadow converted to office towers and parking lots, or best management practices implemented on an agricultural field. Because river basins are complex systems, assessment of the total value of service delivery is both difficult and unlikely to aid in decision-making. Focusing on changes in service delivery is both more practical and more useful [28]. Considering service delivery from different plausible scenarios can identify potential trade-offs (Fig. 5).

Plausible and specific future land-use scenarios are a key to real-world application of the ecosystem services framework for assessing the changes in service delivery. In a study in the Hawaiian Islands, for example, Brauman et al. [29] showed that while the transition from pastureland to native forest increased groundwater recharge, conversion to timber plantation reduced recharge. Land-use change scenarios can be simple and short term, such as a development plan for a single parcel; midterm, such as basin development with and without spatial planning; and long-range, such as future climate scenarios. For example, an ecosystem services analysis could help

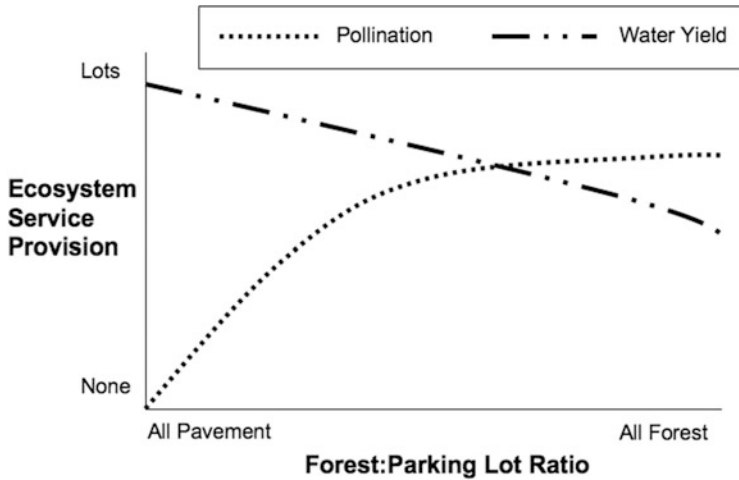


Fig. 5 In a parking lot, no pollination is provided, so the total value of pollination is the same as the change in service provision between parking lot and forest. A parking lot does provide water, however, so an ecosystem service assessment must measure the change in water delivery. Changes to water quality, timing of flows, and location of flows will also be important in the ecosystem services assessment

avoid unintended consequences of a new land-use regulation by predicting how service delivery would change if it was or was not enacted.

3.2 Assessing the Economic Value of Ecosystem Services

3.2.1 Capturing the Value

While people readily acknowledge that ecosystems are valuable, they seldom consider how valuable, either in monetary or nonmonetary terms. In a world of limited land and money, however, where landscape change has real effects on people’s well-being, consideration of the relative value of different ecosystems is important to achieving equitable and efficient distribution of resources [8]. Though it is not a necessary part of an ecosystem services assessment, monetization can be a useful way to quantify value because it provides a common metric, allowing decision makers to compare gains and losses that affect different people at a variety of spatial and temporal scales [30]. Given that cost-benefit analysis has become standard in many decision-making contexts (e.g., <http://www.aquamoney.org>), ignoring the monetary value of ecosystem services means that ecosystems have a de facto value of zero [31]. Ecosystem valuation also sets up a framework for payments for ecosystem services (see Sect. 5.4), which can be a useful tool when command-and-control regulation does not work or is politically infeasible [32].

Criticism of monetary valuation of the environment and of ecosystem services generally falls into two categories: (1) monetary valuation is inherently and ethically inappropriate for representing the value of the environment, and (2) the

methodologies used for determining value incorrectly reflect the true value of ecosystems [33]. For a review of the history, background, and context of environmental valuation, see Farber et al. [34] and Krutilla [35]. They focus on general issues regarding monetary valuation as well as on a number of useful tools for valuation in practice.

Ecosystem services related to water have been valued at least since the 1970s [36]. One common approach in these studies is to look at proxies related to production, for example, using wetland area as a proxy for water purification [37]. More holistic studies look at ecosystem function, such as a study in California in which vegetated buffer strips were shown to be a net benefit to farmers; the improved water quality and reduced soil loss outweighed the costs of removing land from production [38]. A database of nearly 1,500 ecosystem services valuation studies is available via the Ecosystem Services Partnership [39], and a framework for valuing water-quality-related ecosystem services can be found in Keeler et al. [40].

3.2.2 Economic Valuation Concepts

Assessing ecosystem service valuation systematically is important because, historically, when ecosystems have been valued at all, they have been given low prices relative to people's demonstrated dependence on them. Often, the monetary value of ecosystems has been considered narrowly, including only the value of marketable or tradable commodities those ecosystems produce and the value of development that could replace the ecosystem. One explanation for the frequent use of these limited appraisals is the difficulty of valuing non-commodity ecosystem services, such as water purification or recreation value, which are public goods (National Research [41]). Public goods are generally free to users, though they would be costly or impossible to replace. They are also non-rival and non-excludable, meaning that their value is not diminished by multiple users and it is not possible to restrain people from using the good.

In a market, goods are valuable when supply is low and demand is high. When demand increases or supply decreases, prices increase, creating an incentive to conserve. When human populations were small, the supply of ecosystem services was so relatively abundant that these services had little monetary value, despite their integral role in human existence. In medieval Paris, for example, waste dumped directly into the Seine was assimilated by natural processes, leaving the water safe to drink [42]. As population increased, demand for services increased, but because of the public nature of most ecosystem services, their monetary value did not increase along with their real value. By the nineteenth century, the population of Paris had increased beyond the river's ability to assimilate waste, and instead of relying on natural treatment, sewage had to be removed from the city and managed intensively. As with most ecosystem services, there was no direct mechanism to indicate value or to signal scarcity or degradation.

Traditional economics has other characteristics that make valuing ecosystem services problematic [43]. When an ecosystem is functioning, more services are

produced, increasing the supply and thus decreasing the price, so there is little market incentive to protect and improve ecosystem functions. Prices are based on current conditions so that even when future degradation is anticipated, traditional economic measures do not anticipate a reduction in supply. Finally, in a market economy, prices are based on marginal utility—the value of gaining or losing a single additional unit of a good or service. In an ecosystem service valuation, this can be problematic because while the value of a single tree may be higher if it is harvested than if it is left in the woods, the value of the services provided by a whole forest may be much higher than the sum of the parts.

A more complete valuation scheme, either monetary or nonmonetary, takes into account many different kinds of value. Use value can be direct—drinking water filtered through a riparian buffer—or indirect—eating fish that lived in the clean water. Use value also includes option value—the value associated with the possibility of being able to use something in the future. The benefit of simply knowing something exists, called existence or passive use value, as well as the value of being able to pass that existence to future generations, called bequest value, need also be considered.

3.2.3 Scale and Equity

Valuation studies need to carefully consider the distribution of costs and benefits, working to ensure that the value of an ecosystem to all beneficiaries is taken into account [44]. Accounting for beneficiaries at different scales is one example of this—proximity affects the value of an ecosystem to a specific beneficiary, and beneficiaries with different values and interests may be clustered at different distances from a parcel providing services [45]. A study in Madagascar, for example, found that forest conservation was more valuable than logging at local and global scales but that logging was more valuable at a national scale [46]. The generic value of ecosystem services is apparent, but the functionality and value of an ecosystem is likely to be highly variable depending on both the ecosystem and the socioeconomic context [43], so transferring value among sites (i.e., “benefit transfer”) is likely to be misleading and site-specific assessment remains important [47, 48].

The distribution of value between users today and in the future is also important and potentially problematic. Traditional economic methods heavily discount the future, but people may in fact put great value on the availability of ecosystem services for their children or grandchildren [49, 50]. A declining or even negative discount rate (as opposed to the traditionally assumed stable positive one) may be more appropriate when assessing value to future generations, especially when considering resources that are essentially nonrenewable or only renewable on very long time scales [51, 52].

3.2.4 Valuation Methodologies

A variety of methods are available for quantifying value, some measuring attitudes, preferences, and intentions, while others provide economic indicators ([53]; National Research [41, 54, 55]). Nonmonetary methods include ranking of options,

attitude scales, and qualitative summaries. Monetary methods include market and nonmarket techniques. Nonmarket mechanisms include revealed preference methods such as hedonic pricing and travel cost, stated preference methods such as contingent valuation, and second-best options such as avoided cost or replacement cost methods. Thompson and Segerson [55] provide an excellent overview of these methods. Production function approaches explicitly incorporate ecosystem processes into economic studies but do not always include demand in valuation exercises [31, 56]. Economic methods provide imperfect reflections of value; it may be useful to deploy them in conjunction with community- or politically- based methods for ascertaining value [57]. Even ostensibly transparent methods such as market prices for commodities can be problematic because these mechanisms measure willingness-to-pay, which is a function not only of desire but also of ability to pay.

4 Integrating Ecosystem Services with Risk Assessment

In river basin management, as in the management of any complex system with linked social, ecological, and physical processes, there are a variety of uncertainties about the likelihood of both positive and negative outcomes as well as uncertainties about the processes by which those outcomes will occur. Managers must not only determine desired outcomes, they must make decisions about actions to achieve those outcomes without perfect knowledge of what the effects of those actions will be. Given this uncertainty, the ecosystem services approach is one way to help managers determine what information they need and assess whether desired outcomes are being achieved.

4.1 *Using Ecosystem Service Within a Risk Assessment Framework*

Stressors and indicators were introduced in this book's section on "Understanding River Ecosystems" (chapters by Négrel et al. [58], by von der Ohe et al. [59], and by von der Ohe et al. [60]). Here, we tie ecosystem services into the framework as introduced in the first diagram (i.e., Fig. 1) in the chapter by von der Ohe et al. [59] in this volume. In the context of status assessment/surveillance monitoring (top-left box in the diagram), ecosystem services connect indicators and ecological status to water use, providing a guide for goal setting. For example, depending on whether river water will be used for drinking, swimming, irrigation, or fishing, different thresholds would be set for chemical concentrations. Likely there will be multiple benefits from the water, including the knowledge that fish or freshwater plants are thriving. Decision makers will need to assess targets based on stakeholder input.

Ecosystem services link biophysical indicators to land use and land cover. This is causal pathway assessment (top-right box), indicating which land uses cause pressures that have substantial impacts on the delivery of key services. Using

scenarios, it is possible to use the findings of pathway assessment and investigative monitoring to predict how changes in land use and land management will affect the delivery of watershed goods and services. Interventions might involve riparian buffers downstream of fields and towns to reduce nutrient loading in streams [61] or increasing in-stream river flows downstream of a forest plantation by choosing a tree species that grows slowly but requires no irrigation [27].

During the river basin management phase (bottom-right box III), beneficiaries determine the goods and services they wish to manage for and what mitigation measures to take for that. For example, stakeholders might want to maintain agriculture in a catchment but also improve water quality to restore fisheries. Outcomes are assessed (bottom-left box) based on whether desired ecosystem services have been delivered.

4.2 Ecosystem Services and the IRGC Governance Framework

Although it has, to our knowledge, not yet been applied in this context, the International Risk Governance Council (IRGC) governance framework may also be very promising for river basin management as described in [62] of this book (see Fig. 4 in that chapter). The ecosystem services approach fits easily into this framework and can be used to apply the IRGC principles to river basin management in a coherent way.

Characterization of the system occurs during the pre-assessment phase. Ecosystem services provide a straightforward framework for decision makers to determine the functions of the river and the river basin valued by different groups—the basic goals of an IRGC river basin characterization. This is done by ascertaining the services that are important to beneficiaries and identifying and locating those beneficiaries and the owners or managers of ecosystems producing important services. Beneficiaries are the users of ecosystem services within and outside of the basin; owners and managers can be identified because service production is spatial, tied to the parcel where the service is produced. The provision of some ecosystem services, especially those related to water, is not spatially independent—water filtration, for example, is a function of all of the parcels adjacent to a water body, but filtration occurring on an upslope parcel may not be a service if there is no beneficiary [63, 64]. Those who own and manage land producing ecosystem services and those who benefit from ecosystem services are by definition stakeholders—they are invested in the production and enjoyment of river basin ecosystem services.

Determining which ecosystem services are significant is an iterative process. Beginning with generic knowledge of which services are likely to be important in a basin, stakeholders use local knowledge to make the inventory specific and to determine if it is complete. Thus, ecosystem services not only facilitate identification

of the parts of a complex biophysical system that are important to human well-being, they link the biophysical and the social, identifying stakeholders and their connections to the environment. Ecosystem service beneficiaries may not understand the specific ecosystem functions producing the services they value, but by indicating that the final service, such as clean water or a vibrant agricultural economy, is important, they have provided a road map for managers to investigate and protect the production of that service.

For a specific river basin, good status must be defined by managers. Good status is often defined by the achievement of indicators, either physical, sediment loads in the river below a certain threshold, or ecological, certain species present in the river. Ecosystem services facilitate setting standards and thresholds by introducing a social component—depending on who the beneficiaries are, the water body must support those who rely on it for drinking supply, for fishing, or simply for the value of knowing that fish thrive in the river.

During the IRGC's appraisal phase, monitoring for delivery of identified ecosystem services occurs. Ecosystems provide a multitude of services; beneficiaries can identify the most important. Delivery of those services indicates that the system is functioning, a more informative reflection of good ecosystem condition than achievement of a physical indicator. The connection between ecosystem function and ecosystem service delivery is not always well described in the scientific literature, a problem illustrated clearly by Keeler et al. [40] for water-quality-related ecosystem services. The ecosystem services framework provides clear questions to direct future scientific research.

For the IRGC risk characterization and evaluation phase, ecosystem service valuation can help identify the services that are most important to different groups of stakeholders. The valuation need not be monetary—beneficiaries could, for example, rank a variety of services by importance or indicate their willingness to accept a replacement or to go without that service [55]. Valuation provides important information about the tolerance of different groups to service delivery failures and thus what risks they are willing to take.

Ecosystem services are a useful tool in the IRGC management phase. In a world of many demands and limited land and water, trade-offs among land and water uses are a reality. Comparing services delivered by various management scenarios makes trade-offs explicit and identifies affected parties. This allows stakeholders and managers to make decisions with improved information about which groups will be affected and what those effects will be.

Finally, communication and participation is a key part of the IRGC framework. Ecosystem services provide an intuitive language that is open and inclusive, focusing on the connections between people and the environment. While risk often has negative connotations, ecosystem services focuses on benefits and emphasizes working with nature to achieve desired goals.

5 Benefits of Ecosystem Services for River Basin Management

The ecosystem services framework is a tool to be used within the bounds of a larger framework of river basin management and regulation. It can help regulators assess which kinds of land use they wish to promote and which they would like to reduce or forbid by delineating the ways different land uses and management strategies will affect a broad suite of people. The ecosystem services approach does not have to involve monetary valuation, but by providing a framework within which to do so, it expands the tools at a regulator's disposal [11]. Organizing different groups of stakeholders and identifying how they will be affected may be sufficient to illustrate trade-offs from different land-use scenarios.

5.1 *Ecosystem Services Help to Set Priorities*

When performing an ecosystem services assessment within the context of developing river basin protection policy or within the framework of decision-making for the spatial planning of a region, it will be necessary to assess the relevance of ecosystem services in order to prioritize which services and areas receive the most attention, i.e., to set the priorities for river basin management. Different methods could be used to assess the relevance of various services, for example:

- Ecological assessment of ecosystem service provision by an ecosystem. This kind of assessment clarifies the ways ecosystem services are generated and how they are interdependent. It can also help assessment of the resilience of a system [25, 65]. Kremen and Ostfeld [66] provide a framework for assessing the species and communities of species that provide ecosystem services.
- Monetary and nonmonetary valuation (see Sect. 3.2 for techniques). Monetization helps inform cost-benefit analyses, but nonmonetary valuation techniques may be necessary to determine the relative importance of different ecosystem services [47, 55].
- Assessment of potential alternatives for the ecosystem service. Economists have noted that maintenance of an ecosystem service is relevant only if no cost-effective substitute for the service exists [67]. Cost-based decision-making about ecosystem services will thus likely require monetary valuation of alternatives as well as of the ecosystem service in question.
- Ecosystem services area maps. Mapping can provide insight about ecosystem services provision and demand, indicators of ecosystem services, interrelationships among services [68], flows of ecosystem services [69], and the value of those ecosystem service flows [70].

5.2 Ecosystem Services as Common Language Facilitating Participation

The Water Framework Directive (WFD) has developed its own jargon. It enables transboundary communication between river basin managers, as all speak the same WFD language and face the same objectives and same timelines, and it stimulates communication between the policy makers, implementers, and scientists who have a stake in river basin management. However, at the regional water management practitioner scale, jargon can hamper communication with important stakeholder groups such as citizens or farmers. In this context, the few first practical experiences using the ecosystem services concept are more promising (see, e.g., [71] and Sect. 6.2). Social learning processes can benefit from the ecosystem services framework, as it seems to offer an easier language to communicate stakeholder's positions and interests and to discover common interests in the land-water system. Some state that it may even facilitate the common implementation of different environmental policies. However, these very promising benefits have yet to be demonstrated in practice.

Regarding the potential use of ecosystem services as common language, farmers and citizens can easily visualize and then communicate in their own words what “the natural system” offers for their “well-being” (the definition of ecosystem services). This is certainly the case when examples are mentioned to stimulate the thinking process. For the continuation of the stakeholder process, it is not at all relevant if you then call it a function, good, or service that nature offers. It is more important that all stakeholders agree to use to same definition. Thereafter all involved stakeholders can think for themselves and then communicate to the other stakeholders which service is important to them and why. Because farmers and citizens identify services that are important to them through the stakeholder involvement process, ecosystem services may be much more appealing than the abstract WFD goal “good ecological status.” In this way ecosystem services seems to provide a better language to communicate the interests of different stakeholders for jointly exploring a common interest in natural soil-water systems [72].

5.3 Ecosystem Services Management as a European Policy Objective

Over the last three decades, European environmental policy developed from aiming at conservation of single species to halting the loss of biodiversity to a more sustainable use of ecosystem services (see Fig. 6).

At this time, biodiversity protection and sustainable use of ecosystem services are at the core of European policy development and implementation. The European Commission recently adopted a new strategy to halt the loss of biodiversity in the EU by 2020 [73]. One of the main challenges is to translate this strategy into policy and then action. The EC also envisions that by 2050 the EU's economy will have

Year	Policy	Scope (exact extracts from policy text)	
1979	Birds Directive	<ul style="list-style-type: none"> • conservation of all species of naturally occurring birds • measures to maintain the population of the species referred to in Article 1 	conserving single species
1992	Habitats Directive	<ul style="list-style-type: none"> • measures ... to maintain or restore, at favourable conservation status, natural habitats and species of wild fauna and flora 	conserving status of communities of species
2000	Water Framework Directive	<ul style="list-style-type: none"> • prevents further deterioration and protects and enhances the status of aquatic ecosystems • achieve the objective of at least good water status 	enhancing connectivity
2002	Working document Natura 2000 network	<ul style="list-style-type: none"> • community-wide network of nature protection areas • ... to assure the long-term survival of Europe's most valuable and threatened species and habitats. 	
2006	Proposed Soil Framework Directive	<ul style="list-style-type: none"> • preservation of soil functions • current scientific knowledge on soil biodiversity and its behaviour is too limited to allow for specific provisions 	sustaining Ecosystem Services
2008	Marine Strategy Framework Directive	<ul style="list-style-type: none"> • applying an ecosystem-based approach to the management of human activities while enabling a sustainable use of marine goods and services 	

Fig. 6 Development of European environmental policy

grown in a way that “respects resource constraints” and that “all resources are sustainably managed . . . while biodiversity and the ecosystem services it underpins have been protected, valued and sustainably restored” [74]. It is the European ambition to have ecosystem services accurately valued and accounted for by the public authorities and businesses [74].

It is easy to imagine a relationship between the WFD objective of good ecological status and ecosystem services. However, ecosystem services are not mentioned in the WFD. Perhaps this is a matter of timing. In the year 2000, when the WFD came into force, the Millennium Ecosystem Assessment was not yet published. Today, the WFD might have “sustainable use of ecosystem goods and services” as a key objective, as does the Marine Strategy Framework Directive that came into force in the year 2008 (see Fig. 6). Regardless, at the EU policy-making level, the ecosystem services framework is perceived as promising to help WFD implementation. This was confirmed at the “2nd Water Science meets Policy” event held in Brussels, Belgium, on September 2011. The event was entitled “Implementation of the WFD: when ecosystem services come into play.”

In 2012, the Directorate General Environment of the European Commission published the “Blueprint to Safeguard Europe’s Water Resources” [75]. The Blueprint aims to tackle obstacles that hamper action to safeguard Europe’s water resources and is based on an extensive evaluation of existing policy, including the Commission assessment of the Member States WFD River Basin Management Plans. Its long-term aim is to ensure the sustainability of all activities that impact water, thereby securing the availability of good-quality water for sustainable and equitable use. The Blueprint strives to achieve widespread improvement in aquatic

ecosystems. This will contribute to the EU Biodiversity Strategy [73] goal of halting the loss of biodiversity and the degradation of ecosystem services in the EU by 2020 and restoring them in so far as feasible [75]. The Blueprint recognizes the complex reasons for currently insufficient levels of WFD implementation and integration. A primary barrier is the insufficient use of economic instruments. Thus, the Commission will develop a guidance document on methodology to assess the cost effectiveness of water measures and encourage payment for ecosystem services (see Sect. 5.4). This will help identify water efficiency measures and also implement the polluter pays principle [75]. The Blueprint also notes that green infrastructure can help ensure the provision of ecosystem services in line with the EU Biodiversity Strategy [73]. In sum, ecosystem services will be more strongly tied to river basin management from the European (top-down) policy perspective.

Although European policy seems much in favor of applying the ecosystem services framework, it should be noted that there is a considerable difference between policy implementation and the effective realization of policy objectives. Policy implementation can be a paper exercise while actual improvements of the natural system ask for action on the ground. The first generation of WFD River Basin Management Plans is now available. However, plans are only words: only the actual implementation of the selected measures will result in achievement of good ecological and chemical status, i.e., the principle objectives of the WFD [76].

5.4 *Payment for Ecosystem Services*

In a study of functional use of biodiversity in agriculture in the Netherlands, a cost-benefit analyses showed that the provision of ecosystem services such as pest regulation by farmers as a result of alternative agricultural practice provides many societal benefits. However, in many cases the financial benefits of these societal benefits do not flow to the farmer, so the farmer cannot afford to make the necessary investments for implementation [77]. If unequal distribution of costs and benefits is preventing best practices from being applied, payment for ecosystem services could be a solution [32, 78]. Payments can be user financed by individual parties or financed by others (mostly government) on behalf of users.

In most cases, the party that delivers, preserves, or enhances ecosystem services is not being paid by the buyer for a specific ecosystem service delivery but for a specific land use that is assumed to contribute to the ecosystem services target. For example, in Costa Rica, farmers are paid for the ecosystem services they generate indirectly when they employ land-use and forest management practices that preserve biodiversity. The payment program is based on contractual agreements between individual farmers and the government. Payments vary depending on the type of conservation activity, which might include reforestation or forest protection [79].

The Water Protection Fund (FONAG) in Ecuador is a fund, created in 2000, to improve the watershed that provides drinking water to the city of Quito. Water users in Quito (residents, companies from sectors such as agriculture, hydroelectric power, and tourism) contribute to the fund via their monthly water bills. The revenue is

being used to finance watershed management projects, communications and outreach, environmental education, reforestation, and management of protected areas that provide water to the city and to acquire land within the watershed to ensure conservation [80].

In the Netherlands, private parties can receive a payment for providing services that exceed legal obligations and that are aimed at contributing to societal needs related to, for example, nature, landscape, water management, and recreation [81]. European guidelines for state aid in the agriculture and forestry sector [82] and the Common Agriculture Policy (CAP) provide boundary conditions for these types of payments. One example is payment to farmers for cutting vegetation in floodplains; this is beneficial to the fast discharge of water. Another example is payment for maintenance of the wood banks that are part of the characteristic landscape; the topplings are being used as fire wood [83]. In the United States, the Conservation Reserve Program (CRP) provides similar payments to farmers.

Brouwer et al. [84] have performed a meta-analysis of causal relationships between the institutional design and environmental performance of 47 Payments for Watershed Services (PWS). A significant effect was found of the terms and conditions of scheme participation, such as the selection of service providers, community participation, the existence and monitoring of quantifiable objectives, and the number of intermediaries between service providers and buyers on environmental achievement. Direct payments by downstream hydropower companies to upstream land owners for reduced sediment loads were identified as a successful PWS example. Quantitative information on the environmental performance of PWS schemes is however lacking in many of the schemes studied. Brouwer et al. [84] state that international monitoring guidelines are required to enable comparisons, to identify success factors, and to support the future design of cost-effective PWS schemes.

6 Examples of Ecosystem Services Application in Water Management Practice

6.1 Availability of Practical Examples

Ecosystem service-based management projects have been implemented worldwide. Most of the documentation of these projects is in the gray literature; thorough assessments in the scientific literature remain relatively rare. However, the Ecosystem Services Partnership (ESP) is making an attempt to improve this situation. An important aim of the partnership is to mobilize the global but scattered knowledge on ecosystem services to both improve the science and turn that knowledge into action on the ground to show how the concept of ecosystem services can help to achieve conservation and sustainable use of ecosystems, improve local livelihoods, and contribute to the regional and national economy. To facilitate this process, a global network of example projects is needed where scientists work closely together with practitioners and local communities to understand how these social-ecological

systems work and learn how to use the ecosystem services concept for improving ecosystem-based management. These projects can provide lessons learned, test and develop guidelines, and provide an ideal basis for long-term research and education. ESP aims now to collate these projects and make them available via their website.¹

Hereafter we provide a few examples of the practical application of the ecosystem service concept in water management to illustrate points made in previous sections.

6.2 *Dommel River Case Study, The Netherlands*

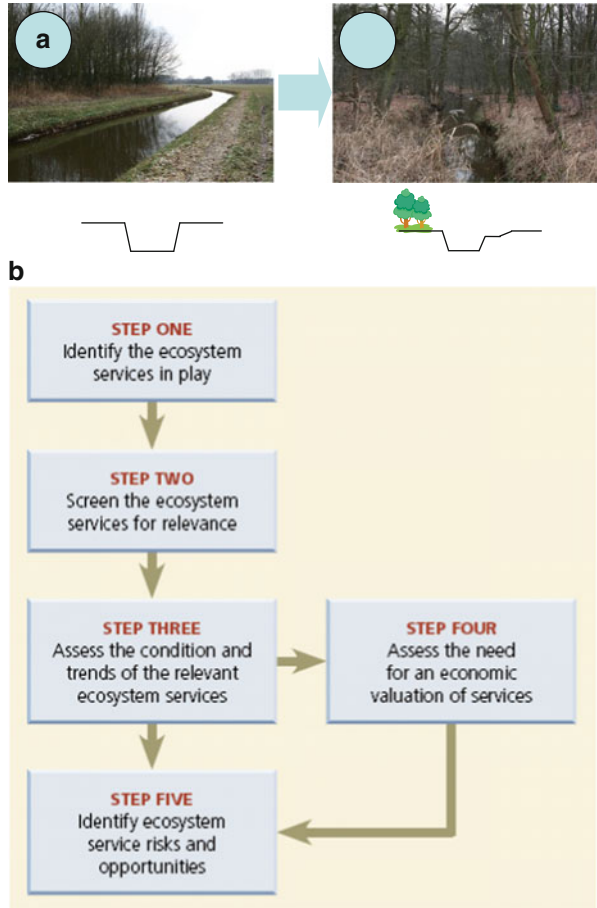
One of the Dutch water boards (responsible for water management at a regional scale) is planning to initiate a river restoration project. The water board has the intention and sees the necessity to cooperate with local stakeholders, mainly the farmers, for successful implementation of this plan. The ecosystem services concept has been tested as a common language in a participatory process with local stakeholders [85]. The study is a rural area of about 500 ha surrounding the small river Dommel and is located upstream of the city of Eindhoven (200,000 inhabitants).

As a general framework for analyses, steps 1, 2, 3, and 5 as proposed by the World Resources Institute (see Fig. 7b) have been conducted in this case study. Information about the social-ecological system was obtained via five individual stakeholder interviews about ecosystem services. The interview included open questions and discussion of a list of approximately 20 ecosystem services. During a workshop with four specialists from the water board and four from a Dutch research institute, the expected effects of an extensive scenario for river restoration on ecosystem services were explored. The changes were related to societal needs and to stakeholders who benefit or who suffer damage because of those changes. The delegates covered the following fields of expertise: hydrology, ecology, communication, ecotoxicology, soil management, water quality, water safety (water quantity), nature development, and physical geography. Based on steps 1–3, risks and opportunities for societal needs in the study area in relation to changes in ecosystem services as a result of river restoration have been identified. Examples of potential measures have been proposed to prevent or minimize the effects of damage to ecosystem services and maximize the use of opportunities of ecosystem services.

Interviewees stated that the interview method broadened their scope. The discussion of a list of ecosystem services provided a lot of extra information on top of the information that interviewees provided during the first part of the interview (open questions). Based on experiences during internal test workshops and experience during other meetings, it was decided not to use the word ecosystem services because stakeholders are in general not familiar with it and the concept is difficult to explain. Instead, the expression “functions of the natural system” was used. It was also

¹ See <http://www.es-partnership.org>

Fig. 7 (a) Assessing the effects of river restoration. (b) World Resources Institute ecosystem services framework ([86], (Figure used by permission of author)



indicated by the first interviewee that the expression benefits was not clear when questions about the beneficiaries were asked; this word has been replaced as well. Ecosystem services that were most difficult to understand are supporting and regulating ecosystem services, especially those that are relevant on a larger spatial scale, such as climate regulation. Also “preventing damage due to natural disasters” needed some explanation to demonstrate that this relates to relative small damages as well. However, specific ecosystem services were generally well understood by the interviewees.

Results show that just the five interviews about ecosystem services—a relatively minor effort—made it possible to gather substantial local knowledge and to come up with a very rich description of the local ecosystem and its link to the socioeconomic system. Effects of river restoration and identified risks and opportunities were related to issues of, for example, water storage, agricultural production, habitat, water purification function of the soil (e.g., to prevent contamination of surface water by groundwater), and recreation. The results of the project have been

discussed with representatives from the water board, some of whom were involved in the project. They concluded:

- Structured analyses of ecosystem services changes due to restoration measures provide a framework that supports thinking.
- The interviews broaden the scope and prevent thinking along specific lines.
- Explaining effects of measures in terms of effects on ecosystem services is expected to support communication with stakeholders and decision makers.
- It is expected that the focus on benefits of measures for several policy goals/societies needs will increase support (public and decision makers, underpinning investment decisions).
- The water board will use the results (risks and opportunities) when developing boundary conditions for river restoration.

6.3 Tamar River Case Study, United Kingdom

The impact on ecosystem services of interventions carried out in the River Tamar catchment by the Westcountry Rivers has been assessed [87]. Valuation of benefits arising from two river restoration programs was determined for provisioning services, regulatory services, cultural services, and supporting services. Some benefits were:

- Increased fresh water quality that is being extracted for public/industrial consumption (result of model study). The estimated annual saving as a result of reduction in the pollutant load of the river is extrapolated from literature.
- Erosion prevention measures will reduce soil loss to the river. The estimated erosion reduction (ton per year, based on literature) is valued based on productivity and transport cost of returning the soil from the estuary to the source.
- Recreational benefits as a result of fish habitat improvement are expressed as increase in farm income from, for example, game fishing and letting cottages.

The analysis was evaluated during a workshop with practitioners and interested parties. One conclusion was that the analysis demonstrates that intended benefits for provisioning services (food and other values contributing to farm incomes) and cultural services (wider contribution to the rural economy) are significant, but the same applies for incidental benefits including regulatory services (such as climate regulation) and supporting services (including habitat provision, nutrient cycling, etc.). A weakness of the analysis as it was undertaken in this study is that key interactions (trade-offs) between services were not adequately identified. Everard [87] advises identifying the scale (i.e., a defined population) for each service since these may differ for different ecosystem services. Carbon sequestration, for example, has a global impact, while benefits of water are generally experienced at a regional scale.

Based on this case study and one in Alkborough Flats, Everard states that the ecosystem services approach helps to identify the full spectrum of stakeholders

affected, or potentially affected, by interventions in environmental (and other) systems. The ecosystem services approach also provides a framework from which to identify novel solutions. Everard concludes by stating “while the ecosystem services concept is quite complicated for a lay audience, the individual services themselves (fresh water, spiritual and religious value, soil formation, climate regulation, and so forth) are readily understood and communicated to the public.”

6.4 Vecht River Case Study, Transboundary Germany: The Netherlands

German and Dutch water managers and local policy makers have experimented with the ecosystem services approach in transboundary river basin management [83]. The study area is part of the river basin of the river Vecht, covers well over 100 km², and includes several villages and a rural area. The main societal challenge is sustainable development of the region in the situation of decreasing population. The general framework of WRI of ecosystem services assessment (see Sect. 6.2) has been applied. Since participants were interested in payment for ecosystem services, attention has also been paid to questions from the UNECE [88] regarding the “process of establishing PES”: (1) Are there any significant water management problems in the river basin? (2) Can ecosystem services help to at least partly address these water management problems? Can these problems be solved or mitigated by means of a project such as change in land use or management practice?

For inventory of the water management problems and to collect information for a description of the social-ecological system, the interview method as described in Sect. 6.2 was used. The list of ecosystem services was tailor made for the study area and included 35 ecosystem services. Ten interviews were conducted: five in the Netherlands and five in Germany from the following sectors/groups: Local water management, Agriculture, Business/industry, Tourism/leisure, Nature protection, Inhabitants. Specific ecosystem services were in general well understood by the interviewees. Some differences were observed between the German and Dutch interviewees with respect to recognition of ecosystem services. This seemed to be related to differences in policy focus over the preceding decades. For example, during some interviews it seemed to be difficult to recognize certain ecosystem services when those functions were traditionally considered human achievements instead of natural processes (this was the case for water regulation). The results of the project have been discussed during a workshop with water managers (regional authority level), policy makers (municipal and national level), and the mayor of one of the municipalities in the study area. They concluded:

- The main added value of the ecosystem services assessment is to find better integrated solutions to the problems in the area, new funding opportunities for measures, and more support by land owners and the general public. It is appreciated that the ecosystem services approach combines the views of different stakeholders. The results of this project, especially the identified risks and

opportunities, provide a broad perspective in which ecology and economy are mingled and which is cross-sectoral, while usually projects focus on one sector.

- Specific ecosystem services are easy to understand and therefore the concept can support communication with stakeholders. The ecosystem services approach might help to explain the aims of water management measures.

6.5 *Catskill-Delaware-New York City Case Study, USA*

The application of the ecosystem services concept in the New York-Catskill-Delaware watershed is probably one of the best examples known of the successful application of this concept in regional water resources management practice ([89, 90]). In the 1990s New York City (NYC) faced rapidly deteriorating surface water quality, mainly caused by an increase in diffuse pollution from upstream agriculture. Thus, urgent action was needed to safeguard the drinking water quality for the nine million people downstream in NYC. However, it was estimated that the logical measure to be taken, i.e., surface water filtration, was going to be extremely costly. This proved to be an enormous stimulus to look for alternative approaches, and ecosystem services took a prominent role. In this case, the involved parties learned together to regard sustainable management of the natural system as a shared responsibility and to regard environment as a profit and not as a cost center. A crucial element to the success of this example was Whole Farm Planning (WFP), which is principally aimed at securing an economically viable farm while minimizing its environmental impacts (for more details, see [91]). Another crucial element to the success of the case was the use of incentives (financial—such as Payment for Ecosystem Services—but also other incentives are possible such as avoidance of onerous regulatory oversight) to make WFP economically viable. Furthermore, farmers were not any longer regarded as enemies of the environment but as environmental stewards that enable ecosystem services (in this case: water filtration/purification by the farmland soil ecosystem) for the benefit of society (i.e., clean surface water for the people downstream in the city of New York).

Brils and Appleton [92] analyzed and identified four key factors that led to the successful application of the ecosystem services concept in this case: (1) the existence of a clear and urgent need by an important stakeholder for action; (2) the taking of an entrepreneurial approach; (3) the definition of, and sticking to, clear targets; and (4) facilitative leadership. This analysis was presented at the event “Implementation of the WFD: when ecosystem services come into play” [92]. At the event, a Dutch and a UK case study were presented that showed remarkable resemblance—although at a considerably smaller spatial scale—to the US case study as presented by Brils and Appleton. Thus it was discussed after the presentations whether the key factors above also applied in the two European cases. It was concluded that the four key factors above applied were confirmed by the European experiences. Maybe this is the case because application of these key principles is just a matter of common sense. Furthermore, it was concluded that ecosystem services tell us why a good status of our ecosystems matters. However, it is contested what “good” means from human welfare point of view.

7 Summary and Conclusions

7.1 Summary

River basins are complex systems in which uncertainty about future status stems from both myriad interacting drivers and from ambiguity about the propagation of the effects of these drivers in time and space. Informed decision-making reduces uncertainty, but many unknowns persist, and thus the risk of arriving at negative outcomes or of failing to achieve positive outcomes is a reality.

The ecosystem services framework aids informed decision-making by explicitly linking the goods and services produced by functioning ecosystems to human well-being, illustrating the broad impacts of various land-use scenarios. Assessing the delivery of ecosystem services, and correcting management if desired services are not delivered, is one way to operationalize adaptive management. Furthermore, using the framework encourages stakeholder participation because ecosystem services provide an open and intuitive language for connecting human well-being to ecological function.

The ecosystem services approach provides a framework for assessing multiple stressors and multiple outputs of a river basin, facilitating management of a complex system. While bringing water bodies up to good chemical and biological status as indicated by the Water Framework Directive is potentially straightforward for a single status indicator, actual water bodies require goal setting for multiple indicators and action on many fronts. Ecosystem services are a useful tool for understanding and managing the manifold physical and social drivers that affect a range of indicators in a variety of ways.

7.2 Conclusions

The ecosystem services framework is useful for river basin managers because it provides a coherent context to incorporate stakeholders and complex biophysical processes into a consistent, learning-based management scheme. The framework links ecosystem processes to people, creating a feedback loop that promotes both ecosystem and human well-being. Because value to stakeholder beneficiaries is explicitly considered, trade-offs can be assessed and addressed.

Some of the key benefits of using the ecosystem services framework in river basin management are:

- The framework can account for multiple services provided by a single ecosystem.
- The framework supports communication among stakeholders across different sectors.
- The framework makes the feedback loop connecting ecosystem services suppliers to users explicit, thus encouraging the protection of the ecosystems that supply important goods and services.

- Because valuation is part of the framework, ecosystem services allow stakeholders to assess trade-offs and eventually—if desired—create the possibility of payments from users to suppliers, a potential supplement to regulation.
- Ecosystem services are flexible. Using the framework does not necessitate payment schemes, but it does allow people to opt in to environmental protection.

Thus, the added value of the ecosystem services framework is to find better integral solutions to the problems in river basins, new funding opportunities for measures, and more support by land owners and the public.

Possible key factors for the successful application of the ecosystem services framework in river basin management are:

- The existence of a clear and urgent need by an important stakeholder for action.
- The taking of an entrepreneurial approach.
- The definition of and sticking to clear targets.
- Facilitative leadership.

While the ecosystem services framework is starting to come into more widespread use in river basin management, there is still a paucity of practical, down-to-earth example projects, and many of those projects have not been systematically evaluated. While the ecosystem services approach shows great potential for all kinds of decision-making about land use and land management, many more projects and assessments will need to be done.

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Participatory Approaches and the Role of Facilitative Leadership

Mattijs van Maasakkers, Michael Duijn, and Britta Kastens

Abstract Involving a broad range of stakeholders in the management of river basins is one of the three key principles of risk-informed river basin management. This chapter describes the role of the facilitative leader in involving stakeholders. He or she guides and assists actors in designing the participatory process and thus helps to improve the process of decision-making, as opposed to other actors, who are primarily focused on influencing the content of the decision. Three examples from practice are described of the work of the facilitative leader in water management related cases. The first case addresses the collaborative development of a ground-water model in the four Northern provinces of the Netherlands, i.e., Groningen, Friesland, Overijssel, and Drenthe. Prerequisite in that development was that the model had to be usable by a variety of governmental actors. The second case addresses the development of a set of policy recommendations for the accommodation of water in the spatial planning of the town of Arnemuiden and its immediate surroundings (province Zeeland in the southwest of the Netherlands). The third case addresses a duo facilitative leadership in the river Dhünn, i.e., a small river in the valley of the German river Wupper that is a tributary of the river Rhine. In this case, stakeholder participation aimed to facilitate the connection between science and regional policy making and to facilitate participation in water management. All three examples demonstrated the important and central role the facilitative leader plays in (1) designing and implementing the collaborative process, (2) assisting the stakeholders in choosing the participatory methodology, (3) translating between

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professional and other kinds of jargon, and (4) forming a stable element in contentious and complex water management related decision-making processes.

Keywords Collaborative water management • Leadership • Participatory approaches • Process management • River basin • Stakeholders

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1 Facilitative Leadership and River Basin Management

River basins are complex “social/ecological systems” (see Fig. 4 in [1], this volume). The “social” elements can consist of a variety of economic, social, and political communities and activities (see also [2], this volume). The result of this complexity and diversity is that in most river basins, no single government entity can claim comprehensive authority nor can a single social scientific discipline claim a holistic understanding of all relevant activities and developments. In addition, understanding of “the ecological” elements of the system involves scientific disciplines varying from chemistry and hydrology to ecology and geology (see book Section A). Therefore,

river basin management (RBM) requires the coordination of, and negotiation between, a broad range of individuals, organizations, and political institutions across legal, political, social, and scientific disciplinary boundaries. Leading such a coordination and negotiation process can be viewed as a distinct role, which requires a particular set of skills [3], responsibilities [4], and tasks. In this chapter, it is argued that facilitation of the river basin management process requires facilitative leadership. A facilitative leader “focuses almost entirely on getting the right people to the table, or at least getting their ideas represented in some meaningful way. The leader as convener gets the process going, and then steps back and lets the group assume the responsibility for finding the right solution” [5]. Within the context of RBM, this means that bringing together the range of available knowledge about risks (see book Sections A and B), and the range of preferences regarding the relative importance of those risks has to be viewed as a distinct and crucially important role. Merely bringing together the right people, representing the range of viewpoints and sources of expertise does not suffice. For RBM to be successful, the participants will have to develop a sophisticated shared understanding of the risks facing the river basin, the relative prioritization of those risks, and reach some level of agreement on how to manage or reduce those risks (see [2], this volume).

A basic assumption underlying this chapter is that RBM should seek to produce not only technically competent and financially efficient but also fair and stable outcomes (see [5]) if it is to be considered successful. There is no single “one-size-fits-all” answer to the question of how a facilitative leader can work to increase the potential for fair, efficient, stable, and wise risk-informed RBM. However, a central tenet of this role is that it is different from those who seek to promote a specific, substantive course of action related to risks in river basins. To further clarify the role of the facilitative leader, it should be emphasized that it differs from the role of policy entrepreneur. The facilitative leader is an actor focused on bringing together the experts, stakeholders, and decision-makers in a particular decision-making process rather than an actor who seeks to promote a specific solution, intervention, or management strategy to deal with a certain risk. In his famous work “Agendas, Alternatives, and Public Policies,” Kingdon [6] described how policy making was consistently and often decisively influenced by promoters of a specific idea or proposal. Kingdon’s work shows that a policy process can be understood as a set of independent streams of problems, policy proposals, and politics. The advocates for specific proposals or for the prominence of a specific idea, he referred to as “policy entrepreneurs.” At first sight, the role of the policy entrepreneur and the facilitative leader are related, since the latter is part of the same “proposal generating” stream. However, there is a difference between (1) proposing a particular financial, physical, or technical intervention to manage a certain risk and (2) seeking to create a basic framework that allows for a more satisfactory discussion of policy options. Therefore, we describe the facilitative leader as an actor in RBM who can propose, facilitate, and lead the process through which risk is managed in a river basin, but who is basically neutral regarding the characterization and prioritization of risk in the river basin. Sabatier [7] describes this type of actor as a “policy broker” and “as distinguishable from an advocate on a continuum.” This description of the role of the facilitative

leader is different from a traditional description of a neutral intermediary since it includes acting as a “process educator” or “activist mediator,” thus providing leadership for a group of stakeholders. The facilitative leader can play a variety of roles, but we view his or her key tasks as (1) including the analysis of current decision-making structures and policies relevant to river basin management in a specific river basin, (2) the identification of all relevant stakeholders within and potentially outside of the basin, and (3) providing leadership in the implementation of a participatory approach in river basin management.

In this chapter four key elements related to stakeholder participation and facilitative leadership will be further elaborated and illustrated using three case studies. These four key elements are:

1. The identification of a reliable neutral party—here called a facilitative leader—to facilitate the design and implementation of the participatory process.
2. The identification of all relevant stakeholders including “latent” stakeholders; the latter being stakeholders who have a direct interest in the outcome of a process, but do not recognize the issue affects them.
3. The tools and practices used to inform and engage stakeholders need to be adjusted in such a way that they reflect the critical participatory considerations in each situation.
4. The inclusion of all relevant categories of stakeholders, thus being able to include a more encompassing of complex information.

It should be noted, however, that simply applying these elements is no guarantee for a conflict-free, quick decision-making process. Incorporating a broad variety of stakeholders in decision-making processes that require the inclusion of large quantities of technical, often, quantified information is an important challenge in RBM.

The three case studies are used to explore and discuss how stakeholder involvement can improve risk-informed RBM. These case studies use different tools for the design and implementation of an interdisciplinary, inclusive, and participatory decision-making process, with a core focus on the role of the facilitative leader. Before addressing the cases, however, first some more theory will be provided on stakeholder involvement in RBM.

2 Stakeholder Involvement in River Basin Management

2.1 Introduction

There are inevitably multiple actors who hold stakes that will be influenced (positive or negative) by any river basin management effort, i.e., the actual implementation of measures. The development of a river basin management plan is likely to include groups like scientists, government officials, landowners, environmental advocates, and a variety of other organizations that have a direct or indirect interest in

policy making regarding the river basin. Sabatier[7]) described this as a regional “policy subsystem,” which is formed by “the interaction of actors from different institutions interested in a policy area.” It is very challenging to include both scientific experts and a broad variety of other actors in decisions in river basin management as, according to Jasanoff [8], “Scientific rigor and public participation can coexist peacefully only in the catalytic presence of trust and community.” Furthermore, Jasanoff adds that “institutional design is important to establish and ensure the presence of trust and community, since . . . increasing knowledge and increasing participation—in the sense of larger numbers of voices at the table—do not by themselves automatically tell us how to act or how to make good decisions. Participation and science together often produce irreducible discord and confusion.” The facilitative leader can assist in the development of trust and community, but must seek to avoid discord and confusion. This is where decision-support tools, like models and metrics (see [9], this volume), can be invaluable, by facilitating the development of a shared understanding of the river basin.

When applying Kingdon’s [6] framework to river basin management, it can be imagined that the third stream in his model—i.e., that of “politics” which creates the windows of opportunity to actually agree on and implement decisions—is structured by the need to develop river basin management plans under the European Water Framework Directive (WFD). Within each basin, different contextual factors will affect the way in which problems like risk are perceived and placed on the agenda (and thus may or may not be addressed eventually by a dedicated measure). For example, the kind of risk that is deemed most relevant in a densely populated river basin will be different from that in a sparsely populated basin. In addition, the policy subsystem that is active within a particular basin will develop a number of ideas and approaches to improve the status of the basin. In a region where many industries are active, potentially different problems and solutions can be expected to emerge than in a basin where agriculture is the main economic activity. It can be expected that the different stakeholder groups, through their respective “policy entrepreneurs” will attempt to exert influence on the RBM planning process.

2.2 Identifying and Involving All Relevant Stakeholders in a River Basin

In order to assess the level, type, and timing of the required stakeholder involvement in RBM, the facilitative leader must gather information on who the stakeholders are and how they understand or perceive the risks within a given river basin (see also [2], this volume). A common method to do this is through a stakeholder assessment [10], but a variety of different practices to generate this information have emerged [11]. The common key elements in these approaches are the following: (1) the identification of stakeholders, (2) the categorization of stakeholders, and (3) the investigation of links between stakeholders [11]. Stakeholders can be individuals, groups, organizations, entities, and agencies

that (1) have direct authority over parts of the river basin through ownership or public responsibility, (2) are determined to be relevant stakeholders by the facilitative leader, (3) are mentioned by other stakeholders, and (4) come forward and express a direct interest in the matter at hand while the RBM planning process is ongoing.

To gather information from the stakeholders, they are usually interviewed in person, over the phone, or on paper using questionnaires. This information is best kept confidential, thus stimulating that stakeholders speak freely. Based on this information the facilitative leader can evaluate whether or not, and for which kind of decision-making process it is most appropriate to suggest involvement of specific stakeholders. Elected officials—like provincial legislators, city councilors, or local politicians—are an important stakeholder category. They can act as conveners to enhance the legitimacy of the decisions that are reached. To avoid confusion, it is important to clarify the role and responsibility of the convener right at the start of the stakeholder involvement process to all stakeholders participating in that process. The convener is typically the agency or organization with the formal responsibility to develop the RBM plan. It can be suggested that the convener will appoint a facilitative leader not originating from the same organization as the convener and preferably not having any stake in the RBM planning process (thus to ensure independency).

Stakeholder analysis usually focuses on formally or informally organized groups, since interviewing every single individual in a river basin is practically impossible. Organization can be viewed as “the mobilization of bias” [12], which can be understood to mean that any existing organization is fundamentally an expression how to engage a particular problem or issue. Since it is possible to identify a virtually infinite number of problems and potential ways to engage them, the basic insight that results from this view on organizations is that there are always organizations that have not (yet) formed around a particular conception of a problem. A stakeholder assessment does not automatically identify the potential ways to engage stakeholders or identify “latent” stakeholders, whose interests in a river basin have not been mobilized yet. To ensure a legitimate process and avoid or reduce the chance of legal challenges after a decision has been made or implemented, it is important to try to include these kinds of groups and seek potential representatives for them.

In addition to the stakeholder’s assessment as described above, it is suggested to also include an ecosystem services review (for a more detailed description of this concept, see [13], this volume). The ecosystem services review may inform the facilitative leader of any latent stakeholders that could benefit or be adversely impacted by any intervention that alters the provision of goods and services in a river basin. A thorough review of ecosystem services could therefore be engaged in collaboration with known stakeholders. In doing so—as the facilitative leader engages in the early stages of the planning process—the known stakeholders can be given the opportunity to add information and/or perceived services to the ongoing ecosystem services review, to allow for the inclusion of different perspectives on the ecosystem. Ecosystem services can thus be used as a framework to bring socio-ecological challenges into the practical problem perception of the

stakeholders. Often, communicating complex problems in the technological discourse that is employed by ecologists makes these appear highly abstract to stakeholders, especially since the latter group can be comprised of people from a variety of social, educational, and epistemological backgrounds, each with very different ways of accessing and perceiving complex problems.

In summary, the stakeholder assessment will result in a description of the main stakeholders in a certain river basin and the ways in which these stakeholders would like to be (or should be) involved in the RBM planning process.

2.3 Analyzing Processes for Participation

There is a broad range of frameworks and processes available [14], at least in theory, to those seeking to improve participation in decision-making in the public sector in general. When proposing a specific decision-making process, a facilitative leader will have to be able to recognize the existing decision-making structure(s) in a given river basin, whether it is formal, informal, or nonexistent. In addition, he or she will have to articulate to the stakeholders how the new process interferes with the ongoing policy process and how it can generate or enhance trust. The concept of trust, which has a long and complicated history in the social sciences, can be understood as the “expectation that an alter will take ego’s interests into account in exchanges” [15]. The concept of community is understood here as a sense of belonging to a group with a shared set of circumstances and challenges. Decision-making in all European river basins occurs in complex environmental and institutional settings. The diversity of uses present in most basins requires an approach to decision-making that is in accordance with that complexity. One can distinguish between the process of decision-making and the outcomes, both intended and unintended, of that process. Each decision-making processes that is to be considered to allow stakeholder participation in a given river basin can be described in a variety of ways. Following Fung [16] this can be done by answering three straightforward questions: (1) who participates, (2) how do they communicate with each other and make decisions as a group, and (3) what is the connection between their decisions and resulting actions? Which type of decision-making process is more appropriate in a given setting is a key question and depends on existing capacity regarding resources, experience in organizing participatory processes, and on preferences within the policy subsystem as well.

3 Case 1: Collaborative Development of a Groundwater Model for the North Netherlands

3.1 The Policy Context

In 2003, the water management organizations in the Netherlands signed the National Governance Agreement on Water. This agreement called for reaching a better balance between water management and spatial development, and in order to do so, it required the implementation of a so-called desired surface and groundwater regime. This regime helps to identify the appropriate combinations of groundwater levels and desired spatial functions, such as agriculture, housing, nature, and recreation. It also assists in determining the feasible policy measures to influence surface and groundwater levels in the desired direction, i.e., the “desired regime.” To develop those regimes for each water management area, an integrated and detailed groundwater model is needed. This model would be helpful to water management authorities—mainly provinces and water boards—to a priori evaluate (or pretest) the impact of future groundwater measures, before being implemented. Such a detailed model is not available yet in the Netherlands, or at least not one that is accepted as credible by the relevant water management authorities. Thus, a collaborative model development effort was needed that had to meet a range of requirements. As it deals here with a complex social/ecological system, the relevant knowledge about this system is dispersed among a variety of types of expertise. So the first requirement was to make use of all these different expertise’s in the model development. Second, there is a variety of stakeholders interested in the management of this water system, and those stakeholders needed to accept the outcomes of the model as policy-relevant and legitimate. As a third element—given the geographical scope of the modeling effort—multiple provinces, water boards, and municipalities were involved, which meant the model had to integrate various datasets from different jurisdictions. The eventual model that was developed is named the “Methodology for Interactive Planning of WATER management” (acronym: MIPWA). However, equally important as the actual model was the collaborative process through which it was designed and developed.

3.2 The Collaborative Modeling Process

MIPWA was constructed in a collaborative modeling process in which scientists, engineers, and policy professionals from 17 water managing organizations in the Northern part of the Netherlands worked together and “fused” their knowledge about different components of the social/ecological system into a model that is scientifically sound and has practical meaning for regional policy making. These organizations included the four Dutch Northern provinces Overijssel, Drenthe, Groningen, and Friesland, the water boards that fall within the boundaries of

these provinces (six in total), the public-private utilities in charge of water delivery, three municipalities (Smallerland, Assen, and Zwolle), and the Rural Areas Service, which falls under the National Department of Agriculture and Fisheries. The modeling process began with an extensive discussion between scientists and engineers and the intended users of the model, i.e., the representatives of the organizations mentioned above. This discussion was focused on the question what kind of support the model should give for implementing the new policy guidelines aimed at achieving the desired surface and groundwater regime. The discussions lasted for several months and resulted in a detailed, joint description of the model's (technical) specifications, taking into account the relatively high level of preexisting knowledge about water management within the participating organizations. The fact that these specifications have not been seriously challenged later on in the modeling process is a clear indication that it paid off to allow a significant amount of time for that discussion. The actual collaboration in the modeling process itself was organized around the stages of groundwater modeling that need to be covered in order to construct a scientifically sound model. These stages are:

- Gathering and processing of “general” data on soil structures and groundwater systems (structures and dynamics) in the various geographical regions concerned
- Identification of modeling concepts, such as drainage and surface water systems
- Schematization of the deep soil, groundwater discharge, and addition
- Identification of the dynamics, including water use for agriculture and the identification of a “representative weather year”
- Static and dynamic calibration of the initial model
- Testing and deciding on the initial model
- Construction of the user interface

Each of these stages was covered by one or more workshops in which the participants of the organizations were closely involved. In these workshops, the modelers shared the progress of the modeling process with the participants. In addition, the participants had to make choices regarding alternative options of dealing with modeling issues. There are several ways to build the model, i.e., by processing the general data or relate the modeling concepts to one another. Scientists from a Dutch knowledge institute and engineers from two Dutch consultancy firms prepared these decisions by identifying the alternatives and listing the underlying pro's and con's. Then, the participants from the governmental and nongovernmental organizations had to reach an agreement on which way to go. In total, about 15 workshops were organized to implement the collaborative modeling process. Most of the workshops were half-day events, organized in the premises of one water board who more or less served as host for the whole MIPWA process.

At the initiation of the MIPWA process, a team of process experts was asked to facilitate this process, very much along the lines of the role of “facilitative leaders” as described above in this chapter. The team analyzed the objectives, expectations, and wishes of the participants with regard to the intended model through an

extensive stakeholder analysis, which included semi-structured interviews and a review of the available documentation on relevant interest groups and organizations. Among others, this information was used to develop the model specifications and to manage the expectations of all involved in the collaborative modeling process. During this process, the stakeholder analysis was repeated to “monitor” if the process was still on track with regard to the expectations of the participants and the intended support it had to provide for the policy process. This was done by conducting a second round of semi-structured interviews with all participants. This revealed that expectations and policy objectives had not significantly altered and that the model, in the way it was being constructed at that point in the process, would meet both expectations and objectives. In addition, the information gathered in this second stakeholder analysis was used to test the model and its user interface.

3.3 Experimenting with Different Policy Scenarios

The development of the groundwater model was facilitated through the construction of an “impulse-response database,” which allows for tests of the model using a variety of policy scenarios. This database contains the expected impacts of all the predefined groundwater measures in each of the modeling cells (size: 25 by 25 m). In this way, a rapid selection could be made of a number of these measures in a designated region. These combined measures form a policy scenario for groundwater management. The selection of measures was then fed back into the model to evaluate their effectiveness with regard to achieving of the spatial policy objectives in that area. The (estimated) impact of the selected measures could be evaluated through their impact on various land use functions, such as agriculture, nature development, housing, and water storage. For example, if a number of measures are selected to lower the groundwater level in support of agriculture, the use of the impulse–response database enables a rapid estimate as to whether or not these measures are likely to have the desired outcomes.

The analytical support that the model can provide was explored in-depth at the end of the modeling process by means of a policy simulation exercise [17,18]. In this exercise three realistic policy cases were presented to the participants. The cases and policy issues were partially drawn up based on information from the second stakeholder analysis that was executed while the modeling process was ongoing. In the policy exercise, the participants had to solve the policy problems in these cases by using the impulse–response database. In this way, they extensively tested the model for one last time, before applying it to real-world policy challenges in groundwater management. Currently, the model is being used by the water board to experiment with potential policy scenarios for connecting groundwater management to land use options in different regions within their respective jurisdictions. The next challenge is to maintain and update the data underlying the model as well as finding new applications for it.

3.4 Facilitative Leadership in this Case

Facilitative leadership in this MIPWA process was organized and provided through a close cooperation between the overall project manager and the process facilitator. Together they formed the “facilitative leadership” team. The overall project manager was accountable for the substantive progress and results of the modeling process. He had to report on this progress to the funding organizations and managed the contributions of the Dutch knowledge institute and the two consultancy firms. In short, when people wanted to know the current state of affairs of the modeling process and of the administrative aspects of the project, the first person they would turn to, was the overall project manager. Given the number and size of the organizations involved, as well as the fact that there were several technical consultants working on this project, the existence of a single point of contact for the entire project was valuable to keep the effort focused. The facilitative leadership team governed the modeling process by designing, moderating, and evaluating the interaction between the representatives of the organizations involved. This interaction was organized through a series of workshops and meetings. On many occasions the facilitative leadership role was carried out by more than the two persons in the leadership team, because of the need for facilitation of subgroups in most workshops. The facilitative leadership team was responsible for finding ways to connect and translate knowledge between scientists and engineers on one side and policy makers (i.e., the intended users of the groundwater model) on the other side. Prior to the modeling process, a member of the team conducted a thorough inventory of the needs for the groundwater model, through extensive talks with the intended users. This inventory was translated in the specifications for the model, again in close cooperation with the intended users. The specifications referred to the different components of the model, the policy measures that had to be modeled, and the level of detail of the model.

An important feature of the modeling process was the existence of a working group, a small delegation of the group representing the participating water, and land use management organizations (i.e., the group of intended users of the model). This group had formed in the early conception of the project idea and was active in the quest for funding of the collaborative modeling process. The overall project manager frequented this group on regular bases for assessment of the dynamics surrounding the modeling project. The overall project manager then discussed the findings and expectations with the process facilitator, with the objective to keep the process attuned to the external expectations and requirements.

In conclusion, facilitative leadership was provided in this case by a team of experts. This team combined substantive knowledge about water systems and modeling with knowledge about procedural expertise and skills in areas like facilitation and communication. The interaction between formally trained experts and stakeholders was carefully managed to allow for learning and true coproduction of the final, highly sophisticated model. Finally, the repetition of the stakeholder analysis by the facilitative leaders helped to ensure the continued legitimacy of the collaborative modeling

effort, and therefore assisted in building credibility for the outcome, in this case a scientifically sound and broadly accepted groundwater model. This case shows that technically proficient, innovative modeling processes can include meaningful participation by a variety of stakeholders, with and without formal expertise in the field.

4 Case 2: Accommodation of Water in Spatial Planning in the Dutch Town Arnemuiden

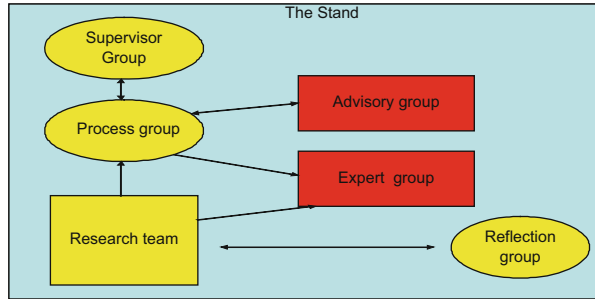
The second case study is located around a small village, called Arnemuiden, located in the southwestern part of the Netherlands, in the province of Zeeland. The province consists of several peninsulas. Many of these used to be islands but are now connected by dams and bridges. However, water remains a dominant spatial feature in the region. Substantial parts of Zeeland are below sea level and its history is marked by a continuous struggle against the sea.

Furthermore, Arnemuiden is situated near the capital of Zeeland, the city of Middelburg, where Arnemuiden as small village is trying to maintain its identity. In 1997, the village lost its status as an independent municipality and became a district of Middelburg. Ever since these administrative reforms, the spatial development of the rural area around Arnemuiden has been a controversial issue. A year before the so-called annexation of Arnemuiden, the voters of Middelburg elected an ambitious city council that was committed to provide an impulse to the city. The spatial development plan for Middelburg (the so-called Quality Atlas) was first presented in 1997 and showed ambitious growth, to be achieved between its publication and 2030. The plan called for inundating agricultural land and building a large number of new houses. This raised opposition in Arnemuiden against any spatial reform initiated by the municipality. The situation entered into a deadlock that lasted for several years, and then turned into a call for an alternative policy. At that time, the province was working on a regional development plan for the entire area around a local water body. In that plan the area around Arnemuiden was still “a blank spot” on the planning map, meaning that a spatial scenario had yet to be developed. All this provided the opportunity to organize the participatory research project as further explained in the next paragraph.

4.1 Organizing a Participatory Research Project

The participatory research project was broad in scope and included a range of stakeholders, making the process design a critical part of the project. Figure 1 shows the process and steps of the interactive planning process conducted. First, a small group of process experts conducted a stakeholder assessment, in which they interviewed stakeholders, sent out a questionnaire to more than 50 stakeholders, and reviewed relevant policies. Then 60 in-depth interviews were conducted with

Fig. 1 Organogram of the groups involved (see text for further explanation)



citizens, farmers, landowners, entrepreneurs, representatives of NGOs, members of expert organizations, and policy- and decision-makers. Thus it was explored what was at stake and for whom, with regard to the future spatial development of the area. These findings were verified in the questionnaire among the participants of an initial scoping meeting to which “all those interested” had been invited. The facilitative leadership team identified different groups of participants, accredited each with their own role and function in the participatory research project to assure embedding at various levels. Most notable are the advisory group, consisting of people with local interests (inhabitants, farmers, land owners, entrepreneurs) and given the task to develop the policy recommendations. Besides this group, an expert group was formed, consisting of technical experts and specialists from the consulting firms, NGOs, and governmental departments involved in the project. This group had the task to support the process group with their specialized knowledge on different (policy) domains. Representatives of governmental agencies were brought together in a supervisor group, which focused on the political, executive, and administrative embedding of the process and its outcomes (see [19]).

Together with the parties involved, the facilitative leaders developed a project plan, including the “rules of the game” or ground rules for the interactive process,¹ and developed ideas on workshops, the development of scenarios, communication, and on reporting of (intermediate) results. The goal of formulating and enforcing these rules of the game was to ensure and maintain support by all organized parties involved. A second, no less important goal was to generate trust within the process. This was done by showing that products were finished in a timely fashion, and guarding that local stakeholders, government agencies, interest groups, and other organizations were respectful and responsive towards each other.

¹ Like restrictions and conditions as posed by laws, appointments on deadlines, end results, rules of behavior, and process management

4.2 *The Participatory Process*

In November 2006, a first session was organized for both the advisory group and the expert group. It was the symbolic start of the research project, having all interested people invited in one room. The provincial executive and the alderman of Middelburg represented the political level and opened the meeting with short speeches, expressing their support and eagerness to receive recommendations directly from the local stakeholders. A member of the facilitative leadership team, after explaining the organization and steps of the collaborative research process, gave a short presentation about the outcomes of the interviews. He explained that the opportunities in the area and the desires of its inhabitants had been evaluated and that four perspectives had become apparent as guiding principles for the future development of the area: (1) water, combined with nature; (2) water, combined with housing; (3) water, combined with recreation; and (4) water, combined with cultural and historic functions. The following months would be used to develop “dream scenarios” regarding these perspectives. These, more fanciful and creative “dream scenarios” would be merged into new scenarios, grounded on the most appreciated spatial elements from the four initial scenarios. In the kick-off meeting, an informative and popularly written booklet about the objectives of the research project and the desired outcomes of the convening assessment was distributed. The booklet also contained the preconditions to be met that the actors involved had put forward for being willing to participate. This provided a shared document that brought together information from different stakeholders, and it showed that the desires and conditions of the different stakeholders were being taken seriously.

In January 2007, the first creative workshop session with the advisory group was organized. To stimulate creativity, the workshop program was kept fairly straightforward. After drinking a cup of coffee, people were asked to join one of the four groups, representing the perspectives on water combined with respectively nature, housing, recreation, and history (see above). The aim was to envisage the area’s potential future development by creating “dream scenarios” on one of the four perspectives. Participants were asked to dream without constraints about an ideal future. The facilitators invited everybody to write down their dreams on small notes. Every group proposed some 50–100 ideas, varying from realizing a new harbor and reopening the Arne (an ancient sea inlet) to developing small-scale housing and nature restoration. The notes were clustered in cooperation with the subgroups by the facilitation team. Next, the team asked which elements should definitely be integrated in the dream stories. This resulted in lists of highly valued “dreams” for each of the four scenarios. The session provided the research team with enough information to start writing a “dream story” that would represent the subgroup’s “ideal future.” The four stories would then be presented at the next workshop session with the advisory group.

Some 6 weeks after the “dreaming session,” the advisory group gathered for the third time and discussed the dream stories that were written by the researchers.

The facilitators asked for additional remarks and suggested changes to the stories. In general, the participants appreciated the way their dreams were put on paper and only had some minor remarks. Next, the dream scenarios were translated to a geographical map. In each subgroup large aerial photographs were mounted on the wall and group members were asked to position the spatial functions of their dream story somewhere in the area. Every subgroup created a spatial representation of their storyline by “moving around” spatial functions (written on self-adhesive notes) on the aerial photograph. These spatial functions, essentially land use types, had been formulated in the previous workshop by the participants and consisted of housing, infrastructure, and recreational facilities. To support the imagination of the participants, draftsmen joined the workshop sessions and sketched the spatial representation of the dream scenarios on a map. Besides this, an inventory was made of questions that the advisory group would like to ask to the expert group. Lastly, four draft maps were made in which the improved dream stories were combined with a geographical representation. Together with a list of 22 questions, the completed stories and maps were sent to the expert group for critical review and for answering of the questions formulated by the advisory group. The four dream scenarios were brought together in a second informative booklet, thus marking the progress of the research project.

The fourth session with the advisory group aimed to identify the most appreciated spatial elements in all four dream scenarios. This session was thoroughly prepared by the research team who analyzed the scenarios for corresponding “building blocks” (spatial elements) and their constituting arguments. For instance, participants had previously indicated that they would like to reopen the Arne (building block) and had given historical, economical, or esthetic reasons for doing so (arguments). For every dream scenario, a table was made linking these “building blocks” to one or more constituting arguments. Large prints of the tables were mounted on the wall. After a plenary presentation of the four dream scenarios, participants were given two stickers for each scenario to mark their preferred element/argument combinations. They also received one red sticker to indicate their disapproval of a spatial element and its constituting argument(s).

The results of the co-valuation (see [20]) were used to determine the elements that were highest appreciated and why they wanted them to be represented in the advice to the spatial development policy makers. The analysis revealed a strong preference for bringing back water in the spatial development area. Yet, some people preferred shallow coves in combination with nature development, while others favored a deep creek suitable for navigation that would attract recreation to the historical village of Arnemuiden. Apart from restoring the historic waterway, the preferences for other functions were more diffuse, revealing diverging degrees of acceptability between extensive and intensive spatial development of the area. It was clear that nondevelopment was not favored by the participants, and neither was an intensive urban expansion.

The analysis of preferences indicated two new scenarios for the area that needed to be further elaborated by the advisory group: a “dark blue” and a “light blue” scenario, referring to the intensity and scale of new water. “Dark blue” refers to relatively more

and deeper water that had to be brought back to the area, reconnecting Arnemuiden to the lake in the northeast. “Light blue” represents the reconnection with the lake as well, but with shallow and smaller canals than in the “dark blue” scenario. The aim of the fourth session was to detail these two new scenarios. After presenting the preferences for the other elements, participants were asked to join either the “dark blue” or the “light blue” scenario for further elaboration. The scenario exercise started by discussing the character of the new waterways in the area and its exact location. Assisted by draftsmen, water was drawn on the map and then all other spatial elements (“building blocks”) were reviewed and reconsidered, describing its function and determining its location in the design of the area. New maps were presented indicating potential spatial development directions for Arnemuiden. A short report was written afterwards for each scenario, describing the elements and arguments behind both spatial development directions.

In September 2007, the advisory group convened again to make the last changes to the two scenarios because the expert group had proposed suggestions that needed to be discussed. As the participants’ values played an important role in the design of the scenarios, every suggestion was thoroughly argued before acceptance or rejection. In both groups (i.e., the “dark-blue” and the “light-blue”), after every decision, the draftsmen would redraw the scenario. The evening was concluded with two refined and more precise drawings, based on which the research team would rewrite the underlying story. At the last workshop session in November 2007, the advisory group performed a final check of the scenarios. Most advisory group members already agreed to the scenarios and did not show up at this meeting. Nonetheless, the process group regarded it as a necessary step allowing people to share their last remarks. Before closing the session, participants were asked to play an active role in the presentation of the spatial plans as advice to the local council and administration. The mayor had agreed to receive the advice in person. Some members of the advisory group volunteered to present the advice and “defend” its underlying ideas and rationale at the final meeting. The spatial planning advice was published in the third and final informative booklet that summarizes the outcome of the research project.

For the final presentation of the spatial planning advice, the process group organized a so-called information market in the civil center of Arnemuiden. The objective of this meeting was the formal presentation of the advice by a representative of the advisory group to the mayor of Middelburg and a delegation of the city council. After the formal presentation, members of the advisory group presented the scenarios and allow the council members to discuss the ideas with their constituents. The information market took place on December 19, 2007.

4.3 The Role of the Facilitative Leader

The Arnemuiden case highlights three aspects of the role of the facilitative leader in water management. First of all, the choice for simple tools (maps, drawings, use of post-its, etc.), combined with careful process design, allowed for new ways of

understanding and actively engaging in the design of spatial development in the region. Being able to “move around” land use types, without requiring technological savvy or specific software skills, enabled engagement of all stakeholders. The second important aspect of facilitative leadership is the ability to develop joint frames for understanding specific policies and interventions. The development of the “dark blue” and “light blue” scenarios functioned as an easy framing device to communicate a complex set of ideas and developments to a broad variety of stakeholders. This type of process allows for the creation of such innovations in language, without “dumbing down” complex knowledge, or simply assuming that most stakeholders will not understand it anyway. The third aspect is the setting and enforcing of ground rules. When done by a facilitative leader, who is considered objective regarding the specific decisions that are made during the process, setting ground rules can prevent allegations that the decision-making process unfairly favors one actor over another. In addition, enforcing ground rules consistently and equally can prevent overly hostile interactions and generate a sense of momentum and interpersonal trust within the process when people are seen promptly following up on commitments.

5 Case 3: Facilitating Participation in Management of the German River Dhünn

5.1 The NeWater project

The main objective of the European Commission funded 6th Framework Program project NeWater² was to understand and facilitate change towards Adaptive and Integrated Water Resources Management (AIWM). AIWM is a management approach that takes into account the complex socio-ecological nature of river basin environments in river basin management policy development and implementation. The goal of AIWM is to do so by increasing and sustaining the capacity to learn while managing the uncertainties associated with management ([21]; see also [2], this volume). A guiding principle in NeWater was codeveloping knowledge and tools with stakeholders and scientists and co-applying of that knowledge and these tools. Stakeholder participation, or more generally public participation, is considered a key issue for AIWM [22, 23]. Within NeWater, stakeholder participation aimed to bridge science and policy and aimed to support the implementation of AIWM. In NeWater, stakeholder participation was, among others, studied and facilitated in the Wupper/Dhünn case study that will be described in the following paragraph.

² See <http://www.newater.uni-osnabrueck.de/>

5.2 *The NeWater Wupper/Dhünn Case Study*

In the NeWater project, academic researchers from the University of Osnabrueck (Germany) supported the implementation of a stakeholder process in the valley of the river Wupper, a tributary of the river Rhine located in the German federal state North Rhine-Westphalia (NRW). NRW is mainly characterized by a high density of population and industry. For decades, the development of the region by humans has influenced the river Wupper in its use and appearance. The river has been polluted and its flow was changed. Dams were constructed in order to secure drinking water provision to the region.

The organization of a stakeholder process took place in a sub-catchment of the Wupper, namely, the river Dhünn. The Dhünn is also a pilot project for implementing the Water Framework Directive (WFD) in NRW (also see [24]). With support of the Wupperverband—i.e., the main water authority, which also coordinates different water uses in the Wupper basin—researchers from the NeWater project initiated a stakeholder process in order to discuss and plan the further steps to be taken towards the implementation of the WFD in the Dhünn catchment. More specifically, the aims were to provide (1) a stakeholder analysis, (2) the methodological background for participatory approaches, and (3) to give support in planning of the implementation and evaluation of the stakeholder process in the context of the WFD.

In NRW, participatory approaches at smaller scales—such as the Dhünn catchment—only began to develop when the process of the NeWater project started in the year 2006. In this context, the team from the University of Osnabrueck, together with the convening Wupperverband organized the follow-up of so-called round tables in the Dhünn. These round tables had already been organized in 2004 in the Wupper basin, but were stopped after a couple of months. The round tables resumed in February 2006 in the Dhünn catchment as a shared initiative between the Wupperverband and, among others, the NeWater project. Within the framework of this cooperative water management effort, several meetings took place between spring 2006 and spring 2008. The main goal of the process was to discuss and develop measures to improve the ecological status of the Dhünn catchment, through a participatory decision-making process. Participatory meaning here that the process had to be designed, implemented, and evaluated in close collaboration with the stakeholders.

5.3 *The Participatory Process Phases*

The participatory process can be separated in three phases: (1) the preparation phase, (2) the dialogue phase, and (3) the wrapping up phase. All phases included participatory elements and followed specific methodologies to support learning within the process (see [24, 25]).

5.3.1 Preparation Phase

The first step in the preparation phase was to define the goal and the basic approach towards the participatory process. This was done in joint discussions between the NeWater team, the Wupperverband, and other research partners. In order to gain the best possible coordination with other projects in the region—and also to avoid redundancies in the investment of work and financial resources—the goals were also coordinated with regional authorities and regional agencies. These are responsible for, or work intensively in, the implementation of the WFD in the case study area. In the second step, the roles of the different participants in the participatory process were defined. The Wupperverband funded the workshops, brought in the local experience, and, as a convener, helped the researchers to win the trust of the stakeholders. The NeWater researchers originally initiated the collaboration and continued as advisors and evaluators of the process particularly regarding the methodological steps and the moderation. A group of professional moderators was part of the NeWater research team. Thus, the NeWater research team as a whole can be seen as the facilitative leadership team. The interdisciplinary NeWater research team and NeWater project approach ensured a systemic approach towards understanding of the Dhünn catchment from the beginning of the participatory process.

The research team conducted interviews and sent out questionnaires to explore the interests of the regional stakeholders. This information provided the basis for deciding which concrete themes to address in the (round table) workshops. The thematic issues that were identified in this way related to water temperature and discharge, structural quality of the infrastructure along the Dhünn and its tributaries, continuous water flow, and additional goals for river bank development. The questionnaires also formed the basis for a comprehensive actor analysis to identify the relevant stakeholders to be involved in the process. Questions to the actors referred to their perceived problems in regional water management, their recommendations for possible solutions, and their expectations regarding the process. Criteria for deciding which stakeholders to involve included (1) the level to which they might be affected by the WFD implementation in the region and (2) their ability to support the discussion on the planning and implementation of measures towards the achievement of the WFD goals. Additional criteria for the selection of stakeholders included interest in participation and a balanced representation of the different stakes.

5.3.2 Dialogue Phase

Within the dialogue phase, three workshops of about half a day each took place. The goals of each workshop were threefold: (1) to generate a joint perception of the most pressing issues related to the implementation of the WFD, (2) to collect and discuss suggestions for suitable measures to tackle these problems, and (3) to jointly agree upon suitable measures and give recommendations for future activities regarding management of the Dhünn river.

Actors from various sectors participated in the workshops, such as water management, nature and landscape conservation, agriculture, forestry, regional planning, and industry. The facilitation followed a modified approach of the “group model building” method by Vennix [26]. After each workshop, a protocol was sent to all participants with a request for comments and/or corrections. In addition, each participant received an evaluation sheet. NeWater researchers assessed the evaluations and the results were used to prepare the next workshop again.

During the first workshop, the participants were asked to suggest all possible measures that could potentially ameliorate the water management problems in the region and to prioritize these measures. Moreover, the participants noted all open questions related to these measures. This allowed all actors to get to know the various perspectives of the different stakeholders in the region. The process also helped to equalize the knowledge base of the stakeholders, which differed significantly in the beginning of the process. After the workshop, the list with all questions was distributed among all participants so they could provide brief written answers. Two months later, the second workshop took place. It started with three presentations by external experts (e.g., from landscape planning and the regional water board). They addressed the open questions that had been identified as particularly relevant during the first meeting. After this, the participants discussed in three moderated breakout groups the open questions and discussed on concrete water management measures and their possible spatial allocation in the region. At the end of the workshop, the breakout group results were discussed plenary. Based on this, a final consensus document was created and distributed for comments among all participants.

The third and final workshop took place 4 months later and was used to discuss the comments that the participants had to the consensus document and to jointly agree upon the content of the document. To this end, the document was projected on screen step-by-step. All comments suggested by the participants prior to the meeting, as well as those that came up during the meeting, were discussed until a joint understanding of all relevant issues was reached. Furthermore, future dissemination and evaluation activities were discussed during this final meeting.

5.3.3 Wrapping Up Phase

During the phase of wrapping up, learning effects for future processes were assessed also in relation to other stakeholder participatory processes conducted in the NeWater project. Another step of this phase was the dissemination of the Dhünn catchment participatory process results and the official signing by all stakeholders of the consensus document that was jointly agreed upon. Next to a description of the process and its methods, the consensus document includes the results and conclusions regarding both the process itself and the proposed water management measures and their possible spatial allocation in the region. This all fed into the official WFD implementation process in NRW.

5.4 Facilitative Leadership in the Dhünn Catchment

The Dhünn case provided another example in which facilitative leadership is provided by a team of process experts. In this case the importance of the role of the convener became apparent. Furthermore, it became apparent that facilitative leadership is a complicated balancing act between providing procedural guidance without substantive pressure in a specific direction. This role needs to be formalized in a contract or working agreement and made clear in participatory process ground rules. In the Dhünn case, the facilitative leaders were not directly funded by one of the regional stakeholders, as the facilitators were funded from the NeWater project budget, so funded by the European Commission, with no direct stake in this case study. This source of funding, combined with the fact that the project was described as a research project, was important in avoiding conflicts of interest. However, the relationship between facilitative leader, convener, and funding source is a complex one, which needs to be transparent to the stakeholders and participants, right from the beginning of the process.

The second aspect of the role of the facilitative leader is the ability to bring in external experts with no direct stake related to the issues at hand, but can bring in specialized knowledge relevant for better understanding of the local situation. By involving the stakeholders in the choice of these external experts, costly conflicts over whose expertise is more credible can likely be avoided. This is where a process like “joint fact-finding” ([27]; and see also [28], this volume) can provide the facilitative leader with a method to prevent the prospect of “dueling experts.”

The third aspect of facilitative leadership that is evident in the Dhünn case is the importance of writing down the agreement in direct consultation with all the participants. The exceptional care with which the final statement was crafted in this case shows that it is possible to collaboratively draft documents, which contain complex information when facilitative leadership is present.

6 Key Findings and Conclusions

6.1 Key Findings from the Three Cases

The three cases described in this chapter provide different, yet thematically overlapping accounts of facilitative leadership in water management. The first case description, i.e., the collaborative groundwater model development case “MIPWA,” highlights three important elements of the work of a facilitative leader.

First Element: The facilitative leader provided communication support between users and developers of the groundwater model. Getting the users and developers to communicate about, and agree on the specifications as the model was developed, was important to ensure its relevance for the users and the scientific soundness as judged by the experts. The translation of demands from the users and possibilities as explained by the developers to each other is a critical process.

Second Element: By choosing to spend significant time on jointly developing and monitoring the specifications of the groundwater model with users and developers, the facilitative leader provided ample opportunity to interact in facilitated meetings. The facilitative leader was able to adjust the speed of the process to the needs of the participants, and this flexibility was acceptable to all involved since the facilitation expertise was deemed valuable.

Third Element: Given the complexity and highly dynamic nature of the process, being a central point of contact and inquiry and being a stable element in the process was another added value of the facilitative leader.

The second case details the work of a facilitative leader, or actually a facilitative leadership provided by process experts, who facilitated a process aimed to accommodate water in spatial planning in the Dutch town Arnhem. This case provides similar insights regarding the potential value of facilitative leaders being actors who can design an integrated process, which includes a variety of different stakeholders in different groups and is sensitive to the changes in the context in which the process takes place. This flexibility is one of the key assets of a facilitative leader who remains engaged throughout the decision-making process, even up to the implementation phase. Facilitation skills played an important role in this process as well, as did the ability to translate between different kinds of jargon by jointly developing new ways to frame the process findings like development of the “light blue” and “dark blue” scenarios. By creation and enforcement of ground rules, a level of trust could be established within the context of the project, which facilitated the communication of a very diverse group of people.

The third case describes the facilitating of stakeholder participation in management of the German River Dhünn. It details a successful intervention by facilitative leaders outside of the formal decision-making structures. This allowed for a higher flexibility in the participatory process approach, which is sometimes considered as one of the preconditions for transformation processes [29], since it allows for the emergence of informal networks facilitating learning processes by creating new information and knowledge flows [30]. The close collaboration between the two facilitative leaders—NeWater and Wupperverband—proved highly beneficial. On the one hand, NeWater’s initiation, scientific steering, and evaluation of stakeholder involvement in an experimental implementation process of the WFD were new for the region. On the other hand, the support by the Wupperverband (a highly respected local stakeholder) was crucial in terms of giving the stakeholders confidence in the NeWater research team. Without being restricted by formal necessities of the official WFD implementation process, this experimental setting allowed the development of a meaningful and creative cooperation between scientists, consultants, and practitioners. This in turn enabled the application of innovative methods and participatory tools to enable active stakeholder involvement. The actual agreement in the Dhünn process was linked to formal WFD requirement of “public participation,” but the WFD does not prescribe how the process should be designed and implemented. This allowed for a high degree of flexibility in that design and implementation and thus in its outcome. The NeWater process design opened the way for participatory experiments outside the boundaries

of existing patterns, e.g., by involving new “methods of recognizing and responding to emerging issues by organizing such an informal participatory platform” [31]. The regular evaluation of the meetings allowed the stakeholders to contribute to the iterative improvement of the process.

6.2 Overall Conclusions

All three cases clearly demonstrate that participatory decision-making is not achieved by simply inviting a range of stakeholders to a meeting. Actually, that is not even the beginning. Properly designed participatory processes start with an explicit attempt to include all relevant stakeholders, which can include the so-called “latent” stakeholders, which are groups that might not be aware that a certain course of action can affect their stakes. To achieve this kind of representation, a sustained effort to bring in all relevant stakeholders is needed, and this is best done through a stakeholder assessment process. This is where a facilitative leader can begin to play an important role.

The second conclusion, emerging from both theory and practice (a.o. the three cases in this chapter) about the role of the facilitative leader in river basin management is that he or she can play a valuable role in the decision-making process itself as well, so beyond its design. This ability hinges on a set of skills, like facilitation and process management, but also on one’s placement in relation to the issues at hand. Obviously, not everybody interested in a river basin can operate as a facilitative leader. One’s expertise, fairness, and focus on process as opposed to content have to be accepted by the other stakeholders. If those preconditions are not met, any procedural recommendation will simply be viewed as an attempt to influence the outcome of the process in an unfair manner.

Finally, it is concluded that the well-thought through selection and use of technical tools, ranging from complicated geospatial models to simple narrative scenarios, is often an important element to help bridge difference between technical experts, local stakeholders, and policy professionals (see also [28], this volume, where it addresses “boundary spanning”). The relative sophistication of those tools should be dictated by the specific circumstances and capacities but their joint development in the context of a participatory process is critical to ensure the legitimacy, relevance, and applicability of the tool. This is the third area in which a facilitative leader can play a crucial role. The development, adjustment, and implementation of these tools can be a lengthy, complicated, and sometimes even frustrating process. To oversee the development of such tools and more generally maintain an overview of the decision-making process as a whole, and to act as a spokesperson for the process, is an incredibly important role. All too often, a desire to develop sophisticated tools can be observed without adequate attention to their connection and relevance to actual decisions that are to be made. Facilitative leaders can establish and protect that connection.

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How Spatial Planning Can Connect to River Basin Management

Geiske Bouma and Adriaan Slob

Abstract Spatial planning essentially involves the development and implementation of strategies and procedures to regulate land use and development in an attempt to manage and balance the numerous pressures placed upon land. Spatial planning can (or should) play an important role in addressing water issues. It is an established mechanism through which some of the river basin management challenges can be addressed. Such challenges are, for instance, flooding and aquatic pollution which are strongly influenced by the nature and location of land use and the changes in that use. However, spatial planning traditions and spatial planning systems within the European Union are diverse. There is no such thing as the common spatial planning system for Europe. In this chapter an analysis is made of how different spatial planning styles in Europe connect to river basin management. From this analysis it appears that especially the style of spatial planning and flexibility in administrative procedures determine whether and how spatial planning can contribute to river basin management. However, there is not one clear preferable spatial planning tradition or type of planning system that has an optimal fit with river basin management. The way the social-ecological systems approach is facilitated by the spatial planning style seems to be an important factor for the match with river basin management. The systems approach matches spatial planning with river basin management as it connects social and ecological systems, and the ecosystem services approach can further enable that connection. Stakeholder involvement, which is linked to area-related interests, addresses the needs and possibilities that will influence river basin management and implementation. The diversity of the spatial planning styles in Europe asks for adaptive and iterative planning in order to be able to implement the connection to river basin management. Adaptive and iterative planning fits well in traditions that rely on flexible organisational planning systems. The connection between spatial planning and river basin management can

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only be made in tailored processes. As the description of spatial planning traditions shows, countries often work along a ‘mix of traditions’; this gives the opportunity to take up the advantages of the different traditions in connecting spatial planning to river basin management.

Keywords Ecosystem services • Land use planning • Planning systems • Planning traditions • River basin management • Spatial planning

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1 Introduction

River basin management entails an integrated approach and links physical planning to water resources planning [1]. There are numerous stressors that induce risks to river basin ecosystems, as dully described in Section A of this book. The Water Framework Directive (WFD; [2, 3]) river basin management plans should address measures to cope with these stressors and risks (see [4] this volume). Developing integrated risk-informed management approaches (see [4, 5], this volume) enables prevention and/or reduction of negative impacts caused by human activities, which are strongly related to land use in the river basins (see Fig. 1 in [4], this volume). Human activities influence each other owing to the competition for land and water resources ([7]; and see chapter by Négrel et al. [8], this volume). Thus, river basin management is a classic example of responding to problems of spatial fit. Lack of fit

causes spatial externalities (activities executed by one user affect others nearby), benefiting free riders and harming others beyond the spatial reach of the responsible institution. Creating better fit involves ‘structuring institutions in ways that maximize compatibility between institutional attributes and bio-geophysical properties’. River basin management is designed to address the interdependencies between, in particular, upstream and downstream effects, water quality and water quantity and water and adjacent land use resources. In doing so, water resources are managed according to the biophysical boundaries of an ecosystem rather than the political-administrative boundaries [9].

Land is a finite natural resource under strain from various often competing demands. These include, for example, social and economic pressures arising from the development of housing estates and transport infrastructures and the need to protect natural resources and land that provides environmental benefits (e.g. well-functioning floodplains). Spatial planning essentially involves the development and implementation of strategies and procedures to regulate land use and development of that use (i.e. land use change: see [10], this volume) in an attempt to manage and balance the numerous pressures placed upon land. Spatial planning, therefore, has an important role to play in addressing water issues such as flooding (see also [11], this volume) and aquatic pollution (see [8, 12], this volume) which are strongly influenced by the nature and location of development. For example, spatial planning policies can help to protect groundwater sites or floodplains through zoning approaches. These approaches, for instance, consist of zoning plans that appoint dedicated zones to a certain land use (e.g. buffer strips) and interest. Planners can also encourage the conservation of water resources via the promotion of water saving technologies as planning conditions attached to permissions to undertake new development [13].

A systems approach within spatial planning gives opportunities to connect the ecological and social system, i.e. the so-called social-ecological system (see Fig. 4 in [4], this volume). Hence, a connection is needed between the river-groundwater-soil-sediment system and spatial planning with its different types of land use. This also calls for bridging the gap between the relevant institutions and organisations involved in spatial planning and river basin management [9]. Stakeholders in river basin management have different geographical and issue-related areas of interest, and they operate at different spatial scales [14].

This chapter addresses the question how spatial planning and land use management can connect to river basin management. It is addressed how spatial planning can facilitate the WFD implementation and the desired coordination between spatial planning, land use and river basin management is elaborated.

2 How Spatial Planning Connects to the WFD

Carter [13] argues that spatial planning is an established mechanism through which the WFD-related water management challenges can be addressed. The WFD and especially the Blueprint to Safeguard Europe’s Water Resources [15] (see also [4],

this volume) state that it is necessary to develop strategies towards a further integration of policy areas. From this point of view, the WFD strives for contribution to other areas of cooperation between European Union member states, specifically the European spatial development perspective (ESDP) is mentioned [2]. In the WFD, direct links are made to land use and the possible pressures on river basins. For instance, land use patterns including areas of urban, industrial and agricultural development will impact the river basins in terms of surface water (see [8, 16], this volume). Also water abstraction for urban, industrial, agricultural and other uses, including seasonal variations and total annual demand, and of loss of water in distribution systems influence the river basins. Concerning groundwater bodies, the land use in the catchment from which the groundwater body receives its recharge is influenced by human activities, for example, through soil sealing, artificial recharge, damming or drainage. It is therefore not a surprise that in the description of the WFD river basin management plans, these pressures and impacts of human activities on the status of surface water and groundwater need to be summarised including the reference to land use.

It is clear that water managers will not be able to implement the WFD on their own. Support will be needed from other policy areas relevant to water use, in particular those with a major influence on land use: agriculture, land use (urban) planning and nature conservation [9]. This also touches on the importance of stakeholder participation (see [17], this volume). Within the water sector there are multiple power centres and scales at which decisions are made. This can be illustrated by the complex system of institutions and actors, needed to deal with water management at the local, national, European and global scale [1]. The implementation of the WFD requires interactive governance between water managers and those responsible for (regulating) land use [9]. The Blueprint [15] also stresses the importance of involving stakeholders to improve implementation of existing legislation, including the WFD.

3 Spatial Planning Traditions Within the European Union

3.1 Spatial Planning on an European Union Level

A central aim of the European Union (EU), as set out in the Treaty of Amsterdam (Article 2) signed in November 2007,¹ is to achieve a balanced and sustainable development, through the strengthening of economic and social cohesion. This implies a balanced territorial development that takes into account the diversity of the territories within the EU. In the absence of corrective policies, the diversity of territories may easily lead to imbalances, often cumulative, in the physical and

¹ See: <http://eur-lex.europa.eu/en/treaties/dat/1997D/htm/1997D.html#0001010001>

human capital of those territories, ultimately resulting in significant divergences in terms of relative wealth, population densities and demographic flows which exacerbate the already divergent economic growth trends.

Several studies confirm that these imbalances can be observed at all scales of territorial analysis. At the European level, the ‘centre-periphery’ model still persists. This model aims to concentrate economic activity in order to promote polycentric development by encouraging the growth of global economic integration zones outside this European core. In this way several larger zones of global economic integration in the EU could be strengthened, equipped with high quality, global functions and services, including the peripheral areas, through transnational spatial development strategies [18]. Also, interactions between the urban and the rural sphere or the disparities existing within the urban sphere are too often disregarded at the intra-regional level. Furthermore, the geographical features of the territory can lead to so-called handicaps as they may represent constraints for development [19, 20], such as remoteness, altitude and slope, insularity as well as population scarcity.

However, territorial cohesion represents an essential precondition for the economic and political success of the building process within the EU community. In this regard, EU enlargement represents a specific challenge for territorial cohesion since it is redrawing the European map, and it is adding further diversified territories (in terms of both development levels and capital endowments). Moreover, the economic convergence achieved by the new EU member states has exacerbated asymmetries within those states between the main urban centres, mostly the capital cities, and the remaining areas [21, 22].

In 1999 the EU developed the European Spatial Development Perspective (ESDP, see: [23]) to anticipate on the agenda of strengthening Europe’s competitiveness. This was adopted in Lisbon in 2000 and reaffirmed in 2005. In some passages the ESDP is attached to the notion of competitiveness (of a region or territory). In other words, regional policy should promote regional competitiveness, but not at the expense of territorial cohesion: future development should be ‘more balanced’ [23]. Emphasis is placed on ‘balance’ at a Europe-wide scale, but ‘the balanced and sustainable development of local entities and regions’ via the promotion of a polycentric settlement structure across the whole EU territory is also a goal of the ESDP [23, 24]. At national and regional level, polycentrism means the promotion of complementary and interdependent networks of towns as alternatives to the large metropolises or capital cities and of small and medium-sized towns which can help integrate the countryside [19].

Territorial cohesion has been under discussion since the early 1990s, and with each new state that joins the EU, the need to pay attention to the evolution of the European territory becomes more acute. These discussions emphasised the importance of territorial cooperation and territorial trends. Main issues concerning territorial cohesion are²:

² See: http://ec.europa.eu/regional_policy/what/cohesion/index_en.cfm

- How can we capitalise on the strengths of each territory so they can best contribute to the sustainable and balanced development of the EU as a whole?
- How can we manage concentration? Cities have at the same time both positive (intensifying innovation and productivity) and negative (pollution and social exclusion) impacts.
- How can we better connect territories? People should be able to live wherever they want, with access to public services, efficient transport, reliable energy networks and broadband internet throughout the territory.
- How can we develop cooperation? The effects of climate change and traffic congestion do not stop at traditional administrative borders, so new forms of cooperation are needed between countries (EU member states) and regions. The 'EU Strategy for the Baltic Sea Region' and the 'EU Strategy for the Danube Region' are examples of new, macro-regional approaches.
- How can we foster 'urban-rural' linkages?

Viewing cohesion from a territorial angle calls attention to themes such as sustainable development and access to services of general interest, such as economic development, public transport, access to healthcare, higher education and training facilities [25]. Furthermore, many issues do not respect administrative boundaries and may require a coordinated response from several regions or states, while others are best addressed at a local or neighbourhood level. It is, therefore, important to improve the cooperation between regions within the EU and with the neighbouring regions outside the EU.

Cooperation between sectors can also bring benefits for territorial cohesion. An integrated place-based approach pursued by Cohesion Policy [26] is ideally suited to respond to complex and strongly embedded issues, such as regional development. However, in order to maximise synergies, better coordination with sectoral policies is necessary. Territorial cohesion also stresses the added value of partnerships with a strong local dimension, which ensures that policies are designed and implemented with local knowledge.

There is, though, according to the 'Compendium of EU spatial planning systems and policies' [27], a common view of planning. It is that of management of spatial development by means of land use regulation or a discretionary land use management system, meaning that approval or denial involves some judgement on the part of decision makers. For interfering with property rights, there must be a legal base, mostly statutory plans adopted by local authorities under national and/or regional planning legislation [18]. The next paragraph describes that this works out quite differently within the EU member states.

3.2 Spatial Planning in European Union Member States

There is no such thing as the common planning system for the EU. The Commission depends highly on the member states in the way the planning system is implemented and executed. Planning systems in EU member states are mirrors of

the cultural, juridical and administrative development of the state and vary in terms of scope, goal setting, processes, and policy setting [28]. The systems also vary in the context of power, e.g. centralisation versus decentralisation, and the relative role of the public and private sector, e.g. planning led versus market led approach.

Even in definitions of the planning approach different views can be recognised. Within the EU there was a separation between ‘land use planning’ (considered as the ‘German model’, statutory planning practice, e.g. laid down in law) and ‘spatial planning’ (considered the ‘French Model’, non-statutory planning strategies, e.g. not laid down in law, more in terms of guidance and policies) [29, 30]. The tradition of the British town and country planning was even defined separately as ‘land use management’ in the EU Compendium of planning systems [27], where planning is more closely associated with the narrower task of controlling land use changes at the strategic and local levels. The traditions noted above give an indication of the different styles observable in the formal systems of planning [27].

3.2.1 Traditions of Spatial Planning

Four major traditions of spatial planning can be identified within the EU and her Member States [27]:

- The *regional economic* planning approach, where spatial planning is used as a policy tool to pursue wide social and economic objectives, especially in relation to disparities in wealth, employment and social conditions between different regions of the country. Central government inevitably plays a strong role. France is associated with this approach.
- The *comprehensive integrated* approach, where spatial planning is conducted through a systematic and formal hierarchy of plans. These are organised in a system of framework control, where plans at lower levels must not contradict planning decisions at higher levels. Denmark and The Netherlands are associated with this approach. In the Nordic countries local authorities play a dominant role, while in federal systems such as Germany, the regional government also plays a very important role.
- The *land use regulation* approach, where planning is a more technical discipline in relation to the control of change of use of land. The United Kingdom tradition of ‘town and country planning’ is the main example of this tradition, where regulation is aiming to ensure sustainable development and growth.
- The *urbanism* approach, where the key focus is on the architectural perspective and concern with urban design, townscape and building control. This tradition is significant in the Mediterranean countries and is exercised through rather rigid zoning and codes and through a wide range of laws and regulations.

Nadin and Stead [31] give an overview of how—in time—the different European states and their planning systems can be classified and how different authors, i.e. studies, made typologies of the planning systems (Table 1).

Table 1 Dynamics in planning system typologies [31]

Davies et al. [50] ^a		Common law England		Napoleonic codes DK, DE, FR, NL	
Newman, Thornley [51]	Nordic DK, FI, SE	British IE, UK	Germanic AT, DE	Napoleonic BE, FR, IT, LU, NL, PT, ES	East European
CEC [27] ^b	Comprehensive integrated AT, DK, FI, DE, NL, SE	Land use regulation IE, UK (and BE)		Regional economic FR, PT (and DE)	Urbanism GR, IT, ES (and PT)
Farinós Dasi [6] ^c	Comprehensive integrated AT, DK, FI, NL, SE, DE (and BE, FR, IE LU, UK) BG, EE, HU, LV, LT PL, RO, SL, SV	Land use reg- ulation BE, IE, LU, UK (and PT, ES) CY, CZ, MT		Regional eco- nomic FR, DE, PT, (and IE, SE, UK) HU, LV, LT, SK	Urbanism GR, IT, ES GY, MT

^aDavies et al. [50] do not give a specific name to the two groups but contrast England and other systems based on their legal frameworks

^bThe EU Compendium identifies ‘ideal types’ of planning traditions. Each country may exhibit combinations of ideal types in different degrees. The ideal types are dominant in the countries indicated here

^cThe ESPON project took the EU Compendium traditions as a starting point and examined how countries, including the transition states of central and eastern Europe, were moving between them

As the typology shows, the planning systems in the states over time borrow and mix elements from the other styles of spatial planning and thus are dynamic (see also Fig. 1). Nadin and Stead [31] argue that Belgium, France, Ireland, Luxembourg and the United Kingdom are taking up elements of the ‘comprehensive integrated’ approach. They also assert that Germany, Ireland, Sweden and the UK are moving towards the ‘regional economic’ planning approach and that Spain and Portugal are moving towards more ‘land use regulation’. Drivers for the dynamics are, for instance, the development of the ESDP and other European policies such as the Structural Funds [22]. The overall movement that took place is mainly towards the ‘comprehensive integrated’ and the ‘regional economic’ approach [6].

Figure 1 shows the distribution of the four styles of spatial planning across Europe. This classification is based on the typologies of planning styles offered by the European Compendium of spatial planning for the 15 ‘old’ EU member states. It is an upgrade of that classification with the inclusion of combined planning style within a state [6].

In summary, the planning system is in part an expression of some fundamental values in a society in relation to, for example, the legitimate scope and aspirations of a government, the use of land and the rights of citizens. Also the impact of the EU and its policy had an influence. The definition of types remains much the same even when different criteria are used.

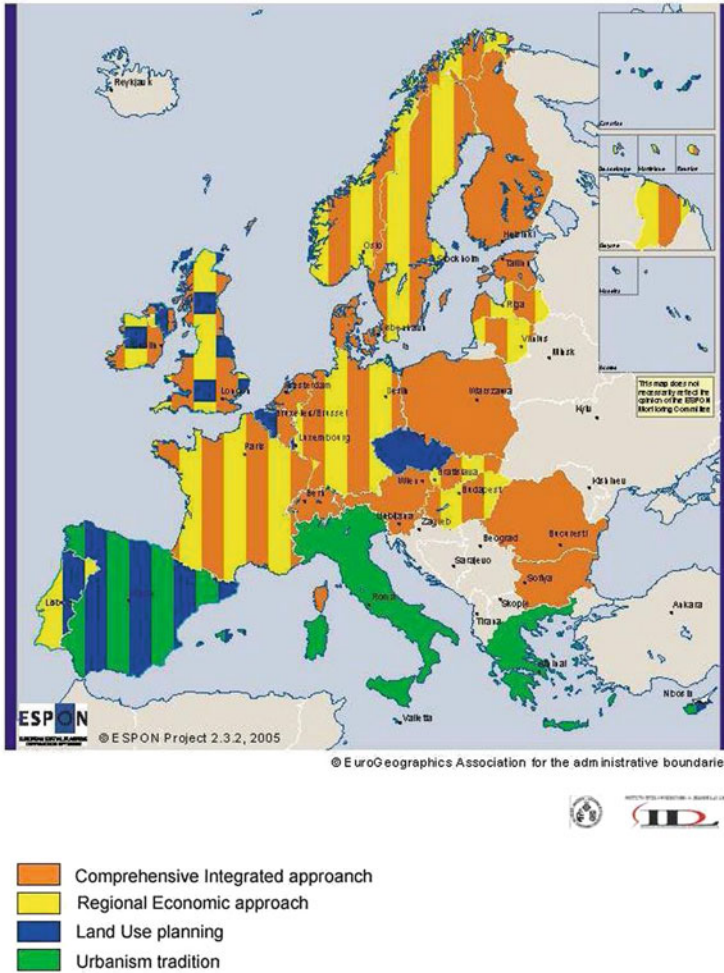


Fig. 1 Styles of spatial planning in Europe: movement within the 15 ‘old’ EU member states and characterisation of new member states and other European states [6]

3.2.2 Operation of Planning Systems

Not only the planning traditions influence the way how spatial planning is operationalised in practice but also the way how planning systems are operated plays an important role. Operation of planning systems relates to organisational and institutional aspects. Thus, two other factors come into play, namely, the extent of flexibility in decision-making and the degree of unauthorised development. In this way planning systems in EU countries can be classified as follows [27]:

- The United Kingdom has a discretionary system—where development proposals are considered on their individual merits, though there is an expectation that the development plan should prevail—and there tends to be a close relationship between objectives of the system and the actual development.
- Denmark, Finland, Ireland and The Netherlands have a moderate degree of flexibility in decision-making, and planning objectives and policies are close to development that takes place.
- France, Germany, Luxemburg and Sweden have systems which have little flexibility in operation, and development is generally in conformity with the planning regulations.
- Belgium and Spain both have rather committed systems while there is only moderate relationship between objectives and reality. Effective control across the whole of the country has not been achieved, and for particular regions or parts of these countries, there are important gaps in the planning framework, or significant levels of unauthorised development.
- Finally there is group of countries, including Greece, Italy and Portugal, where the systems are based upon the principle of committed decisions in plans, but where in practice there has been considerable discrepancy between the planning objectives and reality. High levels of unauthorised development and a plan-making framework with important omissions exist: some areas do not have plans, and existing may be outdated. These systems are, however, undergoing substantial review.

The overall conclusion is that systems differ: they reflect different historical trajectories and legal traditions as well as geographies. Europe is diverse, and policy and administration mirror this diversity [18].

With the overview on spatial planning traditions in mind (previous paragraph), it shows that spatial planning traditions but also the degree of flexibility and organisation are important to consider when talking about spatial planning in Europe. The next paragraph will show how this works out in relation to the WFD implementation.

3.3 How Spatial Planning Styles Connect to the WFD

The way how the WFD interacts in practice with spatial planning and governance of water and land use depends on EU member states' interpretation and implementation of the WFD. The central focus of the WFD is on river basins (see [4], this volume). In this way the EU hopes to encourage a more holistic and territorially integrated approach to solve water-related problems [9]. This touches upon the institutional interplay between water and spatial planning but also has an impact on the interactive (participatory) approach. There are many actors involved in water management, and the WFD requires active public involvement in river basin management planning. The question is whether this fits to the spatial planning traditions and degree of flexibility in the EU member states.

The analysis in this paragraph is aimed to see—taking some EU member states as example—whether the traditions of spatial planning and degree of flexibility and way of organisation of spatial planning match with the goals of the WFD. In the analysis attention is paid to the three main elements for a risk-informed management of river basins: be well informed, manage adaptively and take a participatory approach (see [4], this volume).

3.3.1 France

France is mainly associated with the ‘regional economic’ planning approach. In France the legislative systems and legal arrangements have a strong mandatory component. The administrative arrangements are strongly centralised. ‘Liberty, equality and fraternity’ have been important values since the French Revolution. Given the type of decision-making, France has a system which has little flexibility in operation, and development is generally in conformity with the planning regulations. In France there is a centralised system of procedures, including the French tradition of central government reaching right down to the local level [32]. The French system is centrally directed, with a strong national focus implemented by the central government. The participatory approach seems to need attention in terms of encouraging the stakeholders—for instance, landowners, builders or developers—to take part in the decision-making. This will also have an influence on the way how adaptive management can be applied, since the parties who have a stake in river basin management will need to cooperate in order to bring the defined goals in practice. The social-ecological system approach can be a way to enable adaptive management, connecting physical planning and development to the functioning of river ecosystems. The tendency in France is that a focus is put on territories rather than on political jurisdictions. This makes way for ‘hydro-system’ planning and management and connects to the concept of ecosystem management as well as to the WFD. These developments occurred in France since the middle 1960s and were reinforced by the Water Law of 1992. The result is that the system of zoning and land use regulations can be implemented across more than one commune—and more than one Department—whereas previously this was administratively possible but not systematically practiced [32].

3.3.2 United Kingdom

The United Kingdom (UK) is mainly associated with the ‘land use regulation’ approach. The UK has a common law base, with a strong case law component. Many legislative elements are left open for later Ministerial interpretation. The administrative arrangements are more devolved. Freedom, democracy and rule of law are important values. Given the type of decision-making, the UK has a discretionary system and there tends to be a close relationship between objectives of the system and the actual development.

Within the planning tradition of the UK, the local interests dominate the decision-making over land use. This matches well to the goals of the WFD. The British land use planning system, or spatial planning system, is intended to encompass any relevant issues and factors relevant to decisions about a region and a local authority's area and its land use. The land use planning dominates over water planning including flood risk management [32].

The participatory approach on the local level is strongly embedded within the spatial planning tradition of the UK. This does bring up the question whether the connection to the river basins management is made within the land use planning and if aspects of 'good ecological status' and 'good chemical status' are also taken into account. Next to that, also other policy levels and policy sectors need to be involved in order to be able to operationalise adaptive management, connecting the area development to the functioning of the river ecosystem. Hence, not only an optimum use of all land is important but also connections between land use planning and, for instance, water and environment [32]. The UK is moving from a pure focus on land use planning to a focus on spatial planning [33]. This enables more integration also in terms of a social-ecological system approach.

3.3.3 Germany

The spatial planning tradition in Germany is both associated with the 'comprehensive integrated' and the 'regional economic' approach. In Germany several sectoral policies are involved in implementing the spatial planning policy goals. These include financial equalisation policy (it pursues the distributive objective of establishing 'equivalent living conditions throughout the federal territory'; [34]), economic promotion, large-scale transportation policy, labour market policy, research and higher education, urban development and housing policy and agricultural and environmental policy [6]. Like France, Germany also has a system with little flexibility in operation. Water management in Germany, for instance, is traditionally organised around political-administrative units rather than river basins. From the federal to the municipal level, issues of water management, spatial planning, nature conservation and agriculture are administered by distinct organisational units that are unaccustomed to interacting beyond the scope of formalised planning procedures [9]. Related to the WFD implementation, this means that links will need to be built up between the different policy fields and administrative units. This means openness to new forms of governance relating to water and land use to be able to be adaptive to new circumstances. For member states that traditionally relied on hierarchical, sectoral structures and regulatory instruments to achieve environmental objectives (such as Germany), this will prove to be difficult [9]. A participatory approach—as requested by the WFD—gives possibilities to create more intensive forms of interaction with other stakeholders in order to work on more participatory governance. When building links between the policy fields, this will also give input for enhancement of system understanding, not only looking at land use but also connecting it to the impact on the river ecosystem.

3.3.4 The Netherlands

The Netherlands has a strong planning tradition associated with the ‘comprehensive integrated’ approach. The Netherlands is a decentralised unitary state with a diffuse existence of a regional level, but the structure of the multilevel mechanisms and tools is one of the most developed in Europe. In The Netherlands the local level can be characterised as financially powerful and independent, with an equally strong central state [6].

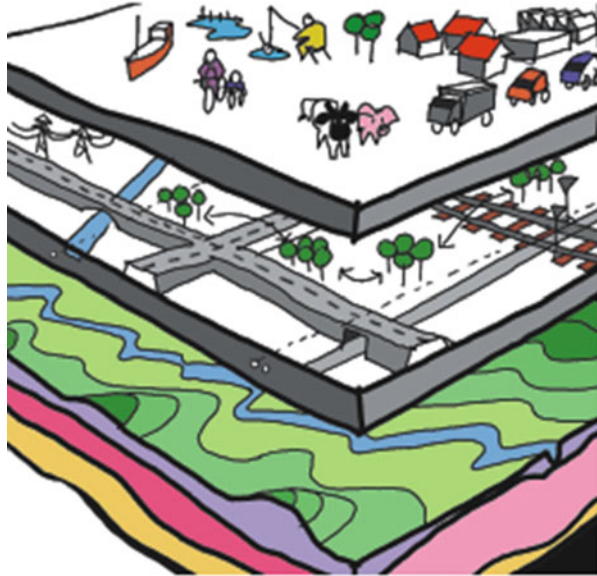
Dutch decision-making has a moderate degree of flexibility, and planning objectives and policies are closely associated to the development that takes place. States with long traditions of government and urban development and administration, such as The Netherlands, can boast a rich and wide spectrum of cooperation arrangements at all territorial levels. The Netherlands, with systems based on consensus principles, can show examples of systematic, regular and institutionalised cooperation between territorial units as well as contractual agreements, linking national, regional and local authorities in integrated policy packages but also simple cooperation of municipalities in the production of joint planning studies [6].

The participatory approach is already part of the spatial planning tradition and organisation in The Netherlands. However, The Netherlands is still working on getting spatial planning and water management together. It is a struggle concerning content, organisation and decision-making [35]. The WFD river basin management plans are a good start, but they also need to be implemented in the spatial visions and regulations for land use management in order to fulfil the spatial fit. Here, the connection between the social and ecological system is also at stake. There is a start, by introducing the ‘layer approach’ (Fig. 2) in the Dutch spatial planning community. This approach connects the human activities (social system) with the water and soil subsurface ecosystem. In order to operationalise adaptiveness, a better combination of the system approach with different organisational (governance) levels is needed.

4 Ecosystem Services, Spatial Planning and River Basin Management

Ecosystems support, often in an invisible way, the development of many spatial elements in urban and rural areas (see, e.g. Fig. 2). Land use influences the development and use of ecosystem services (see [36], this volume) in a positive or negative way (see also Section A in this book). The advantage of the concept of ecosystem services is that the concept explicitly shows the complex feedback loops and mutual influences between ecosystems and human benefits [37]. In this way it is possible to address and consider ecosystems actively within the policy making (including spatial planning) process.

Fig. 2 Layer approach, consisting of three layers (from *bottom to top*): natural system layer (soil and water), network layer (physical infrastructure networks) and occupation layer (urbanisation and spatial functions) (Peter Dauvellier, see: www.ruimtexmilieu.nl)



Brauman et al. [37] put forward that ecosystem services provide a framework for considering the use of natural resources and changes in land use (see also [36], this volume). Tamis et al. [38] acknowledge the added value of ecosystem services in translating it to environmental policies and potential use by stakeholders, thus creating synergy between ecosystem services and indirect benefits. Verhoestraete and Meire [39] argue that ecosystem services as a concept could also be applied for solving spatial problems, to find a balance between the social and ecological system. This should be based on sustainability parameters, where the landscape and planning scale (from micro to macro level) needs to be re-looked at in order to restore the balance. This can lead to defining new ways of land use in favour of the functioning of the ecosystem. Duijn et al. [40] argue that the identification of (subsoil) ecosystem services can promote sustainability: making an explicit link between ecosystem services and spatial development offers the opportunity to make use of the ecosystem in developing urban and rural areas and at the same time preventing negative effects of spatial development on that ecosystem. Furthermore, Colding [41] focuses on the importance of green (infra)structures in urban areas in order to preserve biodiversity. A more dedicated focus on green infrastructure is also promoted by the Blueprint [15]. In this way, ecosystem processes (e.g. species movement, pollination and seed dispersal) can be maintained and essential ecosystem services in city areas, where these may be critical to support (e.g. production landscapes), can be promoted.

Given the human and non-human complexity of land-water systems, collaboration across the social and natural sciences is necessary and a facet of integrated land-water policy [42]. Ecosystem services make it possible to connect spatial

planning to the management of river ecosystems: who is using the services? How are they using the services? Are there alternatives? The ecosystem services concept allows to recognise the value of ecosystem goods and services [43] and to balance the values (see [36], this volume). In this way the concept of ecosystem services helps to tune the spatial system to the ecosystem and vice versa. It also helps to determine how land use (change) impacts ecosystem services. By using scenarios, the changes at stake within, for instance, the river basins can be assessed (see [10], this volume). In the ecosystem services concept, stakeholders are beneficiaries and enablers of ecosystem services, and thus also their stakes need to be taken into account. From the risk-informed river basin management point of view, it needs to be negotiated with them how land use (change) has to be managed and what ecosystems service-related risks and impacts they are willing to accept.

5 Examples of How WFD Implementation Is Connected to Spatial Planning

After the adoption of the WFD by the European Union in 2000, the member states took up the objectives and started working on its implementation. This paragraph describes a few examples of the WFD implementation practice and how this is connected to spatial planning and land use. The Netherlands, Sweden and Spain are addressed as examples as for these states information was readily available on the connection between spatial planning, land use and the WFD implementation.

5.1 The Netherlands: Four River Basins

The Netherlands is situated at the end of four international river basins: the Ems, Meuse, Rhine and Scheldt. The river basin management plans (RBMP) specify the measures to be taken by the Dutch national government, by the provincial and municipal authorities and by the water boards (Ministry of Transport, Public Works and Water Management, [44]). This shows that quite some governmental levels are involved in the WFD implementation. In The Netherlands, water management is decentralised and each authority has its own duties and responsibilities. When developing the RBMPs, an intensive contact between these authorities was established, and a new way of working was introduced. That way of working includes that the national government sets the national framework, takes care of the international cooperation and formulates goals and measures for the national water ways and main streams. And the water boards were given the role to formulate goals and measures for the regional and local level. In the ‘National Governmental Meeting on Water’ (Landelijk Bestuurlijk Overleg Water, LBOW), representatives of all the governmental levels (national, province, municipalities and water boards) participate.

In The Netherlands the current physical land use and spatial planning is the desired starting point in determining the goals and measures within the WFD [45]. At the same time it is concluded that without changing current land use, or changing dynamics in the spatial development, it will be impossible to achieve ‘good ecological status’ [45]. Land use change changes emissions and thus impacts the water system. Spatial development, therefore, has to take these aspects into account, but the link to the spatial programmes is still to be made operational in The Netherlands. Next to that, it takes also quite some efforts to connect climate change mitigation to spatial development (see also [11], this volume). In some cases the implementation of climate change mitigation measures may be contradictory with the WFD goals. For instance, appointing new locations for water storage and making physical adjustments in the river flows to make this possible will alter hydromorphology and thus will affect ecological status (see [8, 16], this volume). Changing use of river floodplain grazing marshes could influence species composition in the river basin [45].

In order to make water an integral part of spatial planning within the Dutch spatial planning system, it will be necessary to lay down the spatial consequences of river basin management in spatial plans. In the new Water Law that was adopted in December 2009, a link is made to the Spatial Planning Act [46]. This means that the ‘Water Visions’, that are available on the national and province level, are also spatial plans (zoning plans—*structuurvisies*) in terms of the Spatial Planning Act. In this way already a legal connection is established. This is an important step within the Dutch spatial planning system and paves the way to operationalise the connection between river basin management and spatial planning.

5.2 *Sweden: The Motala River Basin*

In Sweden an extensive and strong culture of cooperation between stakeholders and territories can be observed. The local level, i.e. the municipality level, is a strong and powerful level [6]. The implementation of water issues on the municipal level are exclusively dealt with in local planning. However, it seems that water has not yet gained the same importance as land in municipal, physical planning. Furthermore, strategies for public participation in the planning process are still lacking [47]. The WFD provides the opportunity to integrate land and water use in spatial planning and to overcome barriers between actors.

Gullstrand et al. [47] have investigated the contents of the ‘Comprehensive Municipal Plans’ (CMPs) concerning water management for one river basin, i.e. the Motala River Basin. The CMPs serve as guideline for drawing detailed plans, which are legally binding, and as guideline for providing of building permits within the area. The main focus in CMPs is on examining how water resources are described thematically and spatially and on what measures are planned to improve and sustain water quality.

In all 13 investigated CMPs, issues on water quality, water use and water and sewage systems were found. The river basin perspective is not visible in any of

these CMPs yet. This leads to less optimal plans, because impacts from tributaries and upstream water use are not yet taken into account. It was also observed that quite some CMPs had not been updated for a long time (i.e. since 1990). Another interesting observation is that conflicts between different interests regarding water use were hardly taken up. Next to that, the size of the municipality—in terms of expecting substantial benefits from having larger administrative resources—has not played a significant role for better adaptation to new environmental objectives. Gullstrand et al. [47] suggest that this could well be related to difficulties in terms of coordinating water issues in municipalities with more and larger stakeholders groups. It is especially this (WFD requested) stakeholder involvement that will need most of the attention for the future. This is connected to the trend that local community participation in the management of natural resources is increasing in Sweden.

5.3 *Spain: The Muga Basin*

In Spain the planning tradition is strongly associated with the ‘urbanism’ approach, while it is moving towards the ‘land use’ planning approach. This ‘move’ provides the opportunity to work towards land use-related planning rather than rigid zoning and towards laws and regulations to determine land use.

In the Muga Basin, an integrated assessment of water uses and water quality was carried out in 2004 within the European HarmoniCOP project.³ Within this integrated assessment, attention was paid to forms and roles in which public participation in river basin management under the WFD had or could take place [48]. This case gives an interesting insight in how it was tried to understand the stakeholders’ framings of the current situation of the problems affecting the Muga Basin and how it was tried to mutually learn about the supporting knowledge which could help both researchers and stakeholders understand the implementation of the participation provisions of the WFD, from a local bottom-up perspective.

In the Muga Basin a transition towards a new water culture with less infrastructure and more demand-based management is taking place. In this process, water use and quality problems associated to that use were addressed. Furthermore, possible policy measures as well as possibilities for public participation and the role of the WFD were identified. One of the resulting recommendations was to integrate public participation in water management into the wider domain of spatial and land use planning, thus to increase the adaptive capacity to solve environmental problems. The Muga Basin process showed that it was necessary to link public participation in river basin management to spatial and land use planning to prevent narrow problem definitions and the incomplete consideration of policy measures. Moreover, it appeared that the problem perceptions of the different Muga Basin stakeholders differed from the problem perception of the high-level policy makers [14].

³ See: <http://www.harmonicop.uni-osnabrueck.de/>

6 Contribution of Spatial Planning to River Basin Management

The spatial planning practices are quite diverse in the EU member states (see previous paragraphs). This indicates that there can be no one-approach-fits-all match between spatial planning and the WFD. Even within states approaches can differ, depending on the region. It is a fact that member states are struggling to match river basin management with spatial planning. That struggle is influenced by the type of spatial planning traditions and the governance flexibility within the member states.

6.1 *Opportunities of the Social-Ecological System Approach*

When focusing on the opportunities of spatial planning to contribute to river basin management, the social-ecological system approach (see Fig. 4 in [4], this volume) may be the ‘bridge’. This system approach matches spatial planning with river basin management due to the logical connection that is made between the social and ecological system. The ecosystem services approach (see [36], this volume) can further support this bridge. Human activities are the driving forces that results in pressures that change the state and thus impact river basins and land use management (see Fig. 1 in [4], this volume). Human activities also influence each other and thus spatial conflicts, i.e. conflicting land use, can arise that have an effect on the ‘driving forces-pressures-state-impact’ chain and thus affect river basin management. When framing it as the capacity of the ecosystem delivering an economic and/or social benefit, this could give another perspective rather than only framing it as an ecological problem.

Spatial planning is often seen as a holistic approach to shape spatial developments, recognising the interrelations and effects of spatial measures in a long-term perspective [49]. It interconnects problems, aims for integration between policy domains and seeks for adaptive strategies and cooperation between stakeholders. Spatial planning coordinates the different relevant socioeconomic objectives and desires—for example, the development of transportation systems, local economy and housing—and coordinates objectives with a strong environmental component, such as nature development, water management and agriculture [49].

Spatial planning can, however, only provide part of the solutions because sectoral policies have already formulated their own policy strategies, applying their own methods, according to their own objectives and available funds. Often spatial planners take up the role to match, for example, local preferences and stakeholder initiatives, with the (sectoral) policy objectives from the local to the (inter)national level. These local preferences too need to be framed in a wider perspective of socioeconomic and bio-geophysical processes. Successful

‘matching’ to optimise the planning process requires a transdisciplinary approach from scientists, politicians and society. Here also the WFD-endorsed participatory approach enables the integration of knowledge from different perspectives. This will also enable a better, more complete view on the issues that are at stake.

There are several ways to facilitate the connection of spatial planning to the social-ecological system approach. A good example is the use ecosystem services concept that was discussed in this chapter as well as in the chapter by Brauman et al. [36], this volume. But also other concepts like the layer approach (occupation, networks and water and soil) that was developed and is applied in The Netherlands can be used to facilitate the connection. When looking back at the analysis of cases in practice (see Sect. 3), it should however be concluded that a connection between spatial planning and a social-ecological systems approach towards river basin management is not yet common in Europe.

6.2 Opportunities of Adaptive and Iterative Planning

As mentioned earlier, the holistic approach of spatial planning is focused on developing adaptive strategies. Also risk-informed river basin management requests for an approach based on adaptivity (see [4, 5], this volume), i.e. urging for an approach that integrates flexibility and a learning-by-doing (iterative) approach to enhance the sustainability of the plans. Thus spatial planning and risk-informed river basin management go hand in hand. Adaptive spatial planning provides an opportunity for (better) integration of flexibility and learning processes. In planning systems with little flexibility, as are operational in France and Germany, better integration of the participatory approach (see [17], this volume) is one of the points of attention. Here there is a need to adapt and thus work out how to connect several policy and decision-making levels and link to the stakeholders involved. Even in countries with long traditions on government and urban development, like The Netherlands, the realisation of the connections between several disciplines (spatial development, water management, soil management etc.) is still a challenge. For countries where the local interests dominate decision-making, like the United Kingdom, the challenge is to learn how the overall goals of river basin management can be fulfilled. All examples show that this is an iterative process: by using an adaptive planning approach and connecting this to risk-informed river basin management, it is possible to work on this step by step. In this iterative (learning-by-doing) approach, the connection to the stakeholders in the field is crucial. Each stakeholder has a different geographical and issue-related area of interest and often operates at different spatial scales. By involving the stakeholders, the step is made to secure that proposed strategies will be feasible and accepted (see also [17], this volume), and since this is area related, spatial planning can help to make it all more concrete and tangible for the stakeholders. Thus—like the ecosystem services approach—it enables their participation.

7 Discussion and Conclusions

Having the four major traditions of spatial planning and the ‘match’ between the spatial planning styles and the WFD in mind, it is possible to conclude on different perspectives on why and how spatial planning connects or should connect to river basin management. This will be done hereafter, for each spatial planning tradition in a single paragraph.

7.1 *Regional Economic Planning Approach*

In summary, the characteristics of this approach are mainly aimed at reaching social and economic objectives, and planning is centrally organised. The link to the ecosystem is not initially present and thus may hamper the connection to river basin management. However, this connection can be accommodated by adopting the social-ecological systems approach (see Sect. 6.1). However, the focus on central organisation in this planning tradition may hamper a participatory approach, and an adaptive and iterative planning approach is less obvious in this planning tradition. Thus, the question is whether there is sufficient room to adopt new ways of working, like adopting the systems approach. It can be expected that for EU member states working according to this planning tradition, it will be even more (see [4], this volume) challenging to reach the goals of the WFD as their tradition does not match easily to a systems approach and is quite rigid in adopting to new approaches, like an adaptive and iterative planning approach.

7.2 *Comprehensive Integrated Approach*

In summary, the characteristics of this approach are organisation mainly through hierarchy of plans and planning system based on framework of control from national to local level. Hierarchy of plans means that the plans on every level (national, regional, local) are closely related to the ‘real’ developments that take place on these levels. Challenging, however, is that the link between spatial development and water (quality) is not always made. Spatial plans may sometimes even be in contradiction with goals of the WFD, as shown in the Dutch example of creating water retention capacity that could negatively influence river basin hydromorphology and thus achieving of the WFD objectives. In a hierarchical system it is also a challenge to—within the same state—overcome administrative boundaries, as ecosystems do not end at the administratively defined borders of a region (e.g. province) or municipality. Promising, however, is that the framework of control in this planning tradition already takes a participatory approach into account. But this does not yet indicate that stakeholder involvement has already

been successfully operationalised in practice. It can be expected that for EU member states working according to this planning tradition, it will be less challenging to work according to the philosophy of the WFD, but a clear challenge to these states will be to create the connection between different (policy) sectors and levels of government.

7.3 Land Use Regulation Approach

In summary, the characteristics of this approach are as follows: regulation is aimed at ensuring sustainable development and growth and focus mainly on control of land use changes. The focus at sustainable development and growth is an advantage for river basin management as it is an important element in the philosophy behind the WFD. However, as local interests could dominate decision-making about land use and building decisions, attention for the coordination on a river basin level is challenging under this approach. The role of the central government is often more on advising than on enforcement. This may hamper the achievement of goals on a river basin level. The control is mainly set on changes of land use. The question is whether this is linked closely enough to other policy sectors. Often land use still dominates over other uses such as the use of water. The bodies dealing with land use planning and all these forms of planning for the water environment often have different administrative boundaries. This makes integration of spatial and river basin management planning even more challenging, and spatial planning dominates in most cases. In conclusion, EU member states working according to this planning tradition have a good starting point for working in line with the WFD philosophy. However, it will be challenging for these states to give water planning a ‘fair share’ within the land use planning system and to connect to the river basin level.

7.4 Urbanism Approach

In summary, the characteristics of this approach are as follows: focus on urban design and urban development, townscape and building control, mainly organised through zoning and codes, laws and regulation. The focus on ‘urban development’ results in a very narrow focus on spatial issues, and a connection to the ecosystem is mostly lacking. Adoption of the social-ecological systems approach could be a way to overcome this. However, the question is whether such adoption will be easy, given the ways of working that have been established under this tradition that relies heavily on laws and regulations and on the effectuation of zoning and codes. Especially the adoption of a participatory approach is challenging here. This planning approach often leads to incomplete consideration of policy measures—as, for instance, shown in the Spanish case—where decisions are mostly taken on a higher level and hardly after consultation of local stakeholders. Thus the input of

local knowledge is mostly lacking and, hence, the decisions are hardly based on a reflection of the practical situation. For EU member states working according to this planning tradition, it may be very challenging starting up processes that are (1) connected to the 'ecosystem needs' and (2) integrate a participatory approach where local knowledge is consulted before decisions are taken.

7.5 *Resume*

The perspectives described above show that there are under each spatial planning approach challenges to connect the EU member states' spatial planning traditions to river basin management. Thus there is in fact not one preferable spatial planning tradition or organisational type of planning system that matches perfectly with river basin management. However, countries with a spatial planning tradition in the 'comprehensive integrated approach' or the 'land use regulation approach' will be able to make this connection more easily.

There are, however, quite some opportunities for spatial planning to contribute to river basin management as described in Sect. 6. The adoption of the social-ecological system approach within spatial planning enables to connect land use (planning) and ecosystems. The participatory approach and stakeholder involvement provide insight in the area-related interests and address the needs that will influence (both positive and negative) river basin management. It is also linked to developing an integrated approach as land use-related impacts of human activities influence the river basins.

On the other hand, the diversity of the spatial planning styles in Europe asks for adaptive planning and iterative working in order to be able to connect to river basin management. This stresses the importance of learning-by-doing and a participatory approach during the development of the river basin management plans. In this way spatial planning can fit best with the needs of the stakeholders in relation to river basin management.

There is no 'one solution fits all' for connecting spatial planning and risk-informed river management as each EU member state has its own spatial planning tradition and own organisation of the spatial planning system. An effective connection can only be made in tailored processes, as the case descriptions of The Netherlands, Sweden and Spain in this chapter indicate. By interacting on the local level, the strategies and actions become more concrete and tangible. In this chapter it was described that the EU member states often work along a 'mix of traditions', providing the opportunity to take up the advantages of the different traditions when connecting spatial planning to (risk-informed) river basin management.

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Improving the Connection Between Science and Policy for River Basin Management

Adriaan Slob and Michael Duijn

Abstract River basin management is highly complicated as it addresses a complex social–ecological system. It consists of a large area that crosses many administrative borders, it involves different stakeholder’s views on the problem and its solutions, and knowledge about the river basin system is uncertain and fragmented. Many problems in the river basin are of the unstructured type, in which policy objectives are contested and high uncertainty exists. Collaborative knowledge production is especially suited to establish the connection between science and policy for this type of problem. From boundary spanning theory, it is argued that collaborative knowledge production requires (a) sound process management (boundary spanning process), (b) people in both worlds that are willing to cross the boundaries (‘boundary spanners’) and (c) production of joint knowledge objects for instance a model, a map or joint paper (boundary objects). In river basin management roughly four groups with different roles and dynamics can be discerned: (1) scientists from various scientific backgrounds, (2) stakeholders with different interests, (3) policymakers from different policy sectors and (4) politicians from different political parties. This implicates multiple boundaries not only between these groups but also within these groups. In two case studies (both complex research projects), the ‘science–science’ boundary and the ‘science–policy’ boundary are explored. From these cases it is recommended that specific boundary spanning processes should be designed and facilitated by professionals who have preferably an education or training in mediation or process management. ‘Boundary spanners’ are the people who can act at both sides of the boundary, either as a scientist or as a policymaker. They should know both ‘worlds’ very well.

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Keywords Boundary spanning • River basin management • Science–policy interface • Unstructured policy problems

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1 Introduction

The connection between science and policy for river basin management is a complicated one for different reasons. First of all is that risk is not a straightforward concept (see [1], this volume). The concept of risk is embedded in social and cultural contexts. Risk assessment, especially on complex issues, is therefore most of the times a contested issue in which perspectives and values of the people involved play a major role [2]. As a consequence, scientific information about risk is contested and subject to debate. River basins that enclose the whole ‘soil–sediment–groundwater–surface water’ system are complex systems: the relations between the different composing parts are multiple, (highly) dynamical and full of surprises and therefore difficult to understand (see Section A in this book). The ambition to even enlarge this system to the social–ecological system (see Fig. 4 in [3], and see [1], this volume) implies that we are dealing with dynamic, adaptive, complex systems with different kinds of responses and time–place relationships [4]. What does this mean for the connection between science and policy in river basin management? This will be further described and discussed in this chapter.

2 The Role of Science in Complex Policy Problems

In policy analysis a classification of four different types of problems is quite common [5–7] and based on two dimensions: consensus about policy values (objectives and/or underlying values) and consensus (or certainty) about the knowledge base to solve the policy problem (see Table 1).

When consensus exists about policy objectives or values and about the knowledge base to be used, this is called a well-structured problem. Well-structured

Table 1 Four types of policy problems (After [5–7])

Certainty on knowledge base or consensus on means	Consensus on objectives or values	
	Low	High
High	Badly structured problem	Well-structured problem
Low	Unstructured problem	Moderately structured problem

problems are policy problems that are not so much debated and contested and are generally of a technical nature. Dealing with water quantity in water management is an example of a well-structured problem. Data are available or can be made available through measurements, and with these data water can be technically balanced in the river basin. When the allocation of water becomes contested by the river basin actors, though, the consensus about policy objectives erodes, and the problem type changes: it becomes a badly structured problem (see below). This shows the problem type is not rigid and can change over time. In reality many policy objectives or values are contested, and consequently well-structured problems are encountered less frequently than we might think.

When policy objectives or values are contested and no consensus exists about the knowledge to be used to solve the problem, this is called an unstructured problem. This type of problems is also known as ‘wicked’ or ‘ill-defined’. Climate change is an example of an unstructured policy problem. Many of the systems issues in a river basin belong to this type of problem, as actors have different interests and good water systems knowledge is still to be developed.

When there is no consensus on the values or objectives to be reached, but on the other hand the scientists, stakeholders or the public know which knowledge should be used to solve the problem, this is called a badly structured problem. The example in which stakeholders have different claims on water resources but still agree on the knowledge to solve the problem, as in the example above, is a badly structured problem. This type of problem can only be solved if the actors want to seek for a compromise for the conflicting claims on water resources.

When consensus exists on the policy objectives or values but it is debated what knowledge should be used or by what means these objectives should be reached, this type of problem is called a moderately structured problem. The way to solve it is to involve stakeholders in the research process, so they can articulate their research questions, and consensus can be reached on what knowledge to use.

As above is shown, the problem typology for a certain policy issue can change over time; in fact all problem typologies are relevant for water management. In river basin management, the policy objectives are often contested by the stakeholders because of the divergent interests. The complexity of the river basin’s social–ecological system makes outcomes of policy measures—such as the Water Framework Directive programme of measures in the river basin management plans—difficult to predict. The knowledge to be used can therefore be contested too. River basin management has therefore often to deal with unstructured

Table 2 The policy process, role of scientists and use of knowledge in each type of policy problems [7]

Problem type	Well structured	Unstructured	Badly structured	Moderately structured
Policy process	Rule	Learning	Compromise	Negotiation
Role of scientists	Problem solver	Problem signalling	Accommodation	Advocacy
Use of knowledge	Data	Ideas	Concepts	Arguments

problems, which asks for another approach to solve them than the often well-structured problems that are dealt with in water (quantity) management.

With each type of policy problem comes another type of policy process, another role of scientists and use of knowledge. This is depicted in Table 2 [7]. Well-structured problems are quite clear problems, for which scientists collect data and information according to scientific procedures and play the role of problem solver. The policy process is also quite clear and procedures and rules are in place to deal with the problem in a structured way. For unstructured problems no consensus exists on policy objectives and on which scientific knowledge should be used. Science plays a role as a problem signaller and scientific knowledge takes the shape of ideas, as empirical evidence is still in development and not sufficiently available. ‘It may be recognized that problem structuring in unstructured problems should involve policy learning by identifying, confronting, selecting and wherever possible, integrating divergent viewpoints and knowledge’ [7]. This is mainly done by stakeholder involvement (see [8], this volume). Policy development can then become an interactive and participatory process, which includes scientists and stakeholders with different perspectives on the problem [7]. In badly structured problems, there is no consensus on policy objectives, but stakeholders agree on the knowledge that should be used to solve the problem. Policy will try to seek for a compromise for the divergent objectives of the stakeholders, and scientists can play the role of a mediator in this process. In moderately structured problems, there is consensus on the policy objectives, but the involved actors look in a different way to the knowledge that should be used and advocate different solutions for the problem. ‘Willingly or unwillingly, science becomes part of the debate, as the different sides tend to strengthen their position by the use of scientific arguments. Use of knowledge is strategic in that it will be used or rejected depending on the interests at stake. Parties will exploit and even encourage competing scientific findings’ [7].

The main conclusion on the connection between science and policy in dealing with complex river basin issues is that the type of policy problem shapes the role of science and the use of knowledge. As most policy problems on the river basin level are of an unstructured or badly or moderately structured type, stakeholder participation is necessary to bring different viewpoints on objectives and knowledge in the policy process (see also [8], this volume). The relationships between scientific information, policy formulation and implementation and stakeholder participation are context specific and dynamic. The very notion that science and policy are two distinct domains that ‘interface’ with each other is a metaphor

that is not uncontested [9]. Other metaphors and models such as knowledge co-production [10] and co-evolution [11] point to different processes through which information and policies are generated. What these processes have in common, though, is that they look at the connection of scientific information and policymaking as an intertwined process in which scientists, stakeholders and policymakers are interacting to deal with a policy problem. The process of posing policy questions and generating scientific sound answers is a highly collaborative process in which the distinction between the different groups is not that much important. This process of collaborative knowledge production is especially suited to deal with complex unstructured problems, i.e. to deal with river basin management-related issues.

3 Collaborative Knowledge Production

An increasing number of policy scientists and practitioners advocate that the traditional approaches to the production of knowledge to inform policymaking, as developed in policy analysis, are not adequate anymore for dealing with today's complex problems [12]. Instead of a hierarchical and modernistic approach, these scientists and practitioners instigate a more deliberative, collaborative and practice-based way of producing knowledge for policymaking (see [13]). Expert-fed, top-down knowledge production is gradually replaced by practical, bottom-up knowledge production.

Lasswell [14] defined the idea of policy analysis in a way that still accurately refers to today's challenges for organising and implementing it. He claims that policy analysis should be contextualised (situated in a certain policy context) and problem oriented (aimed at solving a specific problem) and should be diversified and multidisciplinary (acknowledging the fact that many societal problems transcend the knowledge disciplines and sources). Lasswell's claims are productive pointers for organising the collaborative production of knowledge to inform policymaking. This indicates processes of collaborative policymaking (or participatory, interactive or communicative policymaking) that will strongly benefit from collaborative processes of knowledge production.

The 'Utilization of Knowledge School' [15, 16] studies the inadequate connectivity between science and policy communities from the viewpoint that each community has different sets of rules, habits, languages and standards. The main results of these studies show that factors like presentation, timing and (dis-)similarities with policymakers' beliefs and values are important for the utilisation and impact of scientific information. The establishment of connections between both communities is especially problematic for complex (i.e. unstructured or badly structured) policy problems as the policy process has to deal with many participants with different framings of the problem [17, 18]. Connectivity relates to the quality of relationships and connections between the actors in the system [19]. Hunt and

Shackley [20] distinguish three types of relationships between policymakers and scientists:

1. *Interaction*: refers to a loose coupling. Coalitions of policymakers, scientists and stakeholders coexist and exchange information but retain clear boundaries and identities, have an independent existence and do not reshape each other.
2. *Integration*: refers to a close fit or an intermingling between the groups. Integrated coalitions of research and practice begin to emerge. Agendas change in response to the perceived needs of the other. Some sharing of tacit (experience-based) knowledge—or cross recognition of the tacit components of the other—takes place.
3. *Hybridisation*: occurs when the association between scientists and policymakers produces knowledge, which is more than the sum of the parts. Petts et al. [21] refer to this term in relation to transdisciplinarity ‘a practice that literally transcend traditional disciplinary boundaries’ that is needed to understand complex societal problems. In hybridisation, boundaries between policymakers and scientists of different disciplinary backgrounds dissolve.

We argue that connectivity requires a joint ownership of the policy process and its supporting knowledge bases. Joint ownership is established through collaborative processes of knowledge production between scientists, policymakers and representatives of stakeholder organisations, with the aim of supporting a designated policy process [12]. In this sense connectivity is aimed at establishing what Jasanoff [22] calls ‘boundary work’ and what is called boundary spanning by others.

The challenge of boundary spanning to facilitate knowledge transfer and integration between separate communities is widely discussed in literature and probed in many knowledge management projects [23, 24]. Boundary spanning will become manifest through people, actively acting as ‘boundary spanners’, also referred to as ‘knowledge brokers’ and through the so-called boundary objects. Most policy activities are characterised through the existence of diverging communities of scientists, policymakers and stakeholders, which indicates boundaries between these communities. Leifer and Delbecq [25] define a boundary as ‘the demarcation line or region between one system and another, that protects the members of the system from extra systemic influences and that regulates the flow of information, material, and people into or out of the system’. Organisational boundaries become manifest in communication boundaries. Boundary spanning refers to the activities that are undertaken to cross these communication boundaries and, thus, organisational boundaries. These activities are essentially difficult and ‘prone to bias and distortion’ [26] mainly because of the excessive specialisation in organisations. The role of specialisation regarding knowledge transfer can be described as follows: ‘Specialization and the existence of organizational boundaries are also associated with the evolution of local norms, values, and languages tailored to the requirements of the unit’s work’ (Ibid). These localised norms, values and languages hinder cross-boundary communication and interaction and thus the transfer of knowledge. Dougherty [27] outlined how different thought worlds hinder communication because ‘individuals use different meanings in their

functional setting' [28]. This refers to the necessity of boundary work, even between individuals within one organisational unit or community. Bogenrieder [29] emphasises the intermediary function of boundary spanners by referring to them as linking pins. Furthermore, Bogenrieder claims that 'as the linking pin is a member of several groups, the linking pin could contribute to bringing in knowledge from another group'. They perform communicative activities between groups. Daft [30] indicates that boundary spanning is performed by people and primarily concerns the exchange of information from an organisation to its external environment. Leifer and Delbecq [25] identify boundary spanners as 'people who operate at the periphery or boundary of an organization, performing organizational relevant tasks, relating the organization with elements outside it'.

Knowledge boundaries can be crossed through the collaborative generation, integration and application of the so-called boundary objects. Star and Griesemer [31] define boundary objects as 'tangible artefacts or object-like forms of communication that inhabit several intersecting social worlds and satisfy the information requirements of each of them'. These boundary objects are generated, integrated and applied by professionals working on either sides of the identified community boundaries. Based on her ethnographic study, Bechky [32] concludes that certain boundary objects are not capable of creating common ground because they 'do not invoke the necessary elements of work context'. Thus, at the boundaries of different types of communities, following different organisational contexts and characteristics, different types of boundary objects do apply. She advocates that boundary objects not only contain knowledge (based on, e.g. [33]) but also 'mobilize action in ways other than sharing understanding' (based on, e.g. [34]). A certain context requires (a) certain boundary object(s) for communities to be able to relate to each other and find common ground for collaborative action (i.e. practice). Duijn [35] paraphrases Bechky by indicating that 'the object must be capable of provoking collaborative practice by invoking the *loci* of practice and conceptualization of a problem (or challenge) that each group has. This will trigger an inescapable process of knowledge transformation' (Table 3).

We argue that collaborative production of knowledge to inform policymaking should integrate three perspectives on boundary spanning, that is, boundary spanning people, boundary objects and boundary spanning processes [36]. Thus far, there is little indication that the role of processes is recognised when it comes to 'organising' boundary spanning. Boundary spanning processes deliver the context, time and place to create boundary objects and allow boundary spanning people to intermediate between the involved communities. So, well-designed boundary spanning processes help to cross the boundaries, facilitate collaborative knowledge production and generate meaningful results for the involved policymakers, scientists and stakeholders.

From this theoretical perspective on boundary spanning, we can derive four conditions that should be met for a productive connection between science and policy:

Table 3 The most important concepts of boundary spanning theory

Premise	Communities are separated through boundaries that hamper communication and joint action
Boundaries	Perceived boundaries between communities that can be of different nature (organisational, cultural, geographical, etc.)
Boundary spanning	Activities that are undertaken to cross the boundaries, like communication or joint activities
Boundary objects	Tangible products of joint activities that satisfy the involved communities, like maps, action plans and policy notes. They contain knowledge and provoke action
Boundary spanners	People who cross the boundaries and intermediate between the communities. They are accepted in this role by the communities involved, for instance, because they are 'part' of the different communities
Boundary spanning processes	The processes that are needed to produce the boundary spanning objects with the communities involved

1. Joint production of documents, models, etc. ('boundary spanning objects')
2. People who can combine different fields of knowledge and can attach to different communities ('boundary spanners')
3. Legitimate and transparent processes to guide boundary spanning activities
4. A joint ownership of the knowledge production process

4 Connecting Science and Policy in River Basin Management

Boundaries between science and policy communities hamper the communication and partly explain why science and policy don't go well along, according to the boundary spanning theory. At first sight this sounds satisfactory, but if we look deeper, we see more boundaries than only the one between the science and policy communities.

There are (roughly) four communities involved in solving complex policy problems: scientists, stakeholders, policymakers and politicians. The latter is often forgotten and mixed up with the policymakers, but they represent a complete different group that has other objectives, tasks and dynamics. If we look still deeper into these communities, then we see that each community consists of several sub-communities. Scientists are divided according to their disciplinary background, politicians according to their political party or beliefs, policymakers according to their specific tasks (most of the time related to a certain law or directive) and stakeholders according to their interests that are involved or affected. Therefore, boundaries between the communities are multiple. To keep it simple, we only look at the boundaries between the science and the policymaker communities. There are three types of boundaries to cross:

1. The 'science–science' boundary: the boundaries between the different scientific disciplines that are hard to cross as they use different concepts and vocabulary.

2. The ‘policy–policy’ boundary: the boundaries between the different involved policy domains. In the case of river basin management, it involves spatial planning, land use, soil policy, fresh water, soil, sediment and groundwater management (see [37], this volume).
3. The ‘science–policy’ boundary: the challenge for the involved scientific disciplines is to come up with meaningful results for the policy domains.

We shall examine two types of boundaries: the science–science boundary and the science–policy boundary. For each boundary we will describe one case. The science–science interaction is illustrated with the AQUATERRA research project, while the science–policy interaction is illustrated with the MIPWA project (see [8], this volume). The policy–policy boundary is left out of our examination because we have gained no personal experiences, but only remote observations, with the difficulties in crossing this boundary within the cases described hereafter.

4.1 The AQUATERRA Integrated Project: A Multidisciplinary Approach to River Basin Management

4.1.1 Introduction to the Case

The AQUATERRA project ([38]¹) was a multidisciplinary, integrated project under the European Commission’s 6th Research Framework Programme (FP6) involving 42 partner organisations from 12 European Union Member States and several other European countries. The project was executed from June 2004 to May 2009. AQUATERRA aimed to ‘provide the scientific basis for improved river basin management through better understanding of the fresh water–sediment–soil–groundwater system as a whole, by integrating both natural and socio-economic aspects at different temporal and spatial scale’ [39]. The project was driven by current and forthcoming environmental policy needs by developing state-of-the-art river basin system understanding. AQUATERRA represented one of the largest and ambitious research projects in Europe that was aimed at improving of the understanding of the functioning of contaminated river basin systems. In Fig. 1 the work package structure of the AQUATERRA project is shown.

This brief project description immediately indicates the prominent presence of knowledge boundaries that have to be crossed in order to meet the project’s objectives. The ‘systems approach’² to the object of study and the ambition to

¹ See also www.eu-aquaterra.de

² Systems approach means that the project aimed for improving of the integrated understanding of the ‘groundwater, (deep) soil, sediment, subsurface and surface water’, i.e. the river basin ecosystem

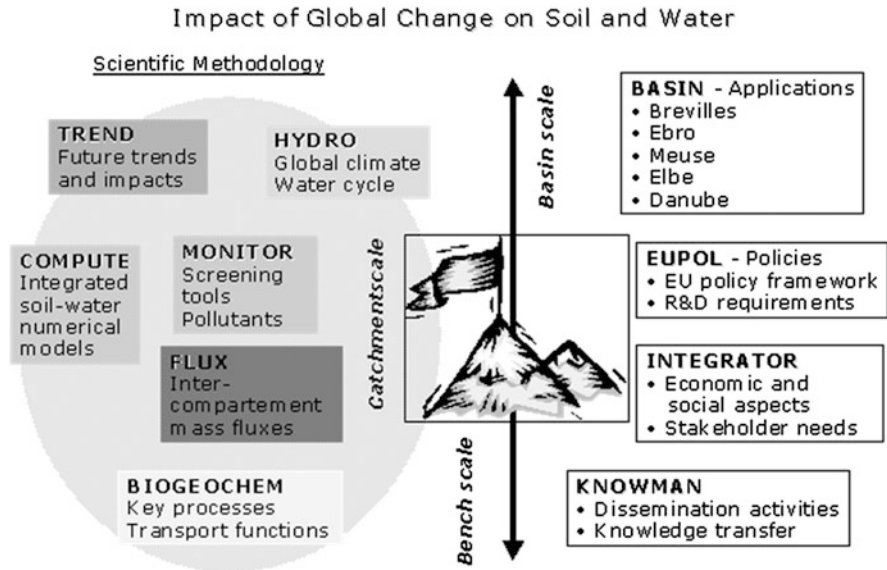


Fig. 1 Schematisation of the European Commission-funded integrated project AQUATERRA. Each *block* indicates a single work package [39]

produce policy-oriented knowledge reveal the knowledge boundaries in this research project. Scientific knowledge has to be integrated to be able to advance the state of the art in system understanding. Therefore, crossing the science–science boundary was very relevant for the AQUATERRA project.

4.1.2 The Science–Science Boundary

One of the challenges in AQUATERRA was to integrate the scientific research and its results, by bringing scientists from different disciplines together. The AQUATERRA research community consisted of many smaller factions, grouped around separate knowledge disciplines in dedicated work packages, as shown in Fig. 1. Although the AQUATERRA researchers shared their scientific background, every discipline has its own practice and language, which hampers integration of research results [40].

There was not a specific work package in the AQUATERRA project dedicated to the process of crossing the disciplinary boundaries and managing the boundary spanning process in the project. The absence of such a boundary spanning work package meant that the integration of scientific knowledge and information was not deliberately organised or coordinated, but was left to the initiative of the work package leaders who were driven by the need to arrive at some integration of scientific information to generate results. This integration was kept within the

boundaries of the work packages as there were no real incentives to integrate results further. As far as boundary spanning processes across disciplines appeared, these were firmly kept within the boundaries of the work packages.

The following boundary spanning activities are worth to mention. A sampling boat trip that lasted several days was organised at the beginning of the project to collect samples for the assessment of the chemical and ecological status of the Danube. This assessment required the input from several disciplines, and as they were confined to work together on the boat, this exercise turned out to be a boundary spanning activity. The scientists from different disciplines worked together on this assessment report, which can be recognised as a boundary object. Another attempt for a boundary spanning activity was a coupled biophysical economic model that was made for one of the river basins studied in AQUATERRA. The developed model could have acted as a boundary object that was established through the collaboration of scientists from diverging disciplines. The coupling of the scientific disciplines with different ways of thinking was not easy, so at the end it turned out that the object was not more than a coupling of two models that were not really integrated.

In hindsight, the science–science boundary was breached in some occasions. With Hunt and Shackley’s [20] theory in mind, we can conclude that boundary spanning was established and organised on the interaction level. Real integration or even hybridisation of knowledge across diverging scientific disciplines was not realised.

4.1.3 The Role of Boundary Spanners

Some people acted in the AQUATERRA community as boundary spanners, who focused on science–science interfacing. Boundary spanners that could connect science and policy in a meaningful way were very rare in the community. There were some scientists who took a leading role in integrating some of the knowledge from various work packages. They did this on a voluntary basis, driven by personal interest in science–policy interfacing, as it was not an obligation they had to fulfil for a specific task in the project. They simply had the experience, visionary power and creativity to look beyond their own disciplinary boundaries. The boundary spanning was expressed in presentations they gave and in some articles that were produced.

Retrospectively, the AQUATERRA project description makes vividly clear that boundary spanning matters in dealing with complex research projects, especially those that are meant to inform policymaking processes. However, it does not seem to emerge spontaneously and should deliberately be organised and implemented in the project. What especially lacked in the AQUATERRA project was a specific incentive or sense of urgency to cross the scientific boundaries. Nevertheless, the observations show the relevance of boundary spanning as a concept for understanding and guiding knowledge transfer in the hybrid science–policy domain.

4.2 Collaborative Groundwater Modelling by Scientists and Policymakers: The MIPWA Project

4.2.1 Introduction to the Case

In the chapter by van Maasakkers et al. [8], this volume, the joint modelling effort for the MIPWA project is already described. As the interaction for this modelling process addressed scientists and stakeholders from governmental bodies, especially the science–policy boundary is of interest.

The participatory modelling process started with discussions on the question what kind of support the model should give to the implementation of the ‘desired surface and groundwater regime’ (i.e. the policy objective). These discussions lasted for several months and resulted in a comprehensive description of the model’s specifications from a technical and a policy perspective. In a series of workshops, the model was constructed. The support that the model should provide was explored in-depth at the end of the modelling process by means of a policy exercise [41]. For this exercise, three realistic policy cases with complex groundwater management issues were prepared by the scientists and engineers. These cases were thereafter presented to the policy professionals. In this way, they extensively tested the model for one last time, before applying it to real policy challenges in groundwater management.

The science–policy boundary was crossed by the continuous and active involvement of scientists and engineers on one side and of policy professionals of the aforementioned water managing organisations on the other side. In a way this boundary was less challenging than one might expect. The reason for this is that most of the professionals involved, on either side of the boundary, had an educational background in (geo-) hydrology. This significantly facilitated the interaction between the participants at least because they understood each other’s lingo. Furthermore, they shared the ambition of constructing a state-of-the-art model.

4.2.2 The Role of Boundary Spanners

Most of the scientists, engineers and policy professionals involved in the MIPWA project can be recognised as boundary spanners as they functioned as ‘linking pins’ [29], who link the modelling process to their home organisations. They conferred with colleagues about the intricacies in the modelling efforts and fed the results of these deliberations back into the process. Especially the policy professionals acted as intermediaries by transferring the (preliminary) results of the modelling process to their constituencies, such as the executive boards, politicians and societal stakeholders. In turn, the policy professionals represented the interests of these actors in the construction of the intended groundwater model. Their intermediary role enhanced the policy relevance of the model considerably because it kept the scientists and engineers ‘awake’ with regard to the objective of the modelling

process, namely, the application of the model in policy processes for groundwater management. Time and again the policy professionals drew attention to the fact that the legitimacy of the outcomes of the model in use would undoubtedly be scrutinised by representatives of stakeholder organisations, such as the agriculture sector organisation and associations for nature conservation. The applicability of the model for policymaking was in this way a very dominant prerequisite in the modelling process.

4.2.3 The Science–Policy Boundary

The collaborative knowledge production process can be described as ‘bricolage’ [42] which can be summarised by the adage that ‘we have to make do with whatever is at hand’. Beforehand ‘everything necessary for building a ground water model that covers the whole of northern Netherlands’ can be thought of, but only in the actual modelling process itself, it will become clear whether all knowledge and data to do so are available or not. In the MIPWA project scientists, engineers and policy professionals learned to discover the actual possibilities of the intended model while constructing it. Together they identified the ‘blank spots’ in the knowledge available to fully live up to the expectations and objectives they and their constituents had, before starting the modelling process. These blank spots had to be filled in ‘on the spot’ by ‘bricolage’: each of the participants had to contribute to a possible solution for filling in the blanks and keep the modelling process going. Duijn and Rijnveld [12] indicate that ‘scientists, consultants and policy professionals negotiated how they should deal with the choices and obstinacies in the data collection, in the steps of the modelling process and in the validation of the model’. These negotiations resulted in ‘small innovations’ that were developed and implemented underway, even without being fully noticed or appreciated. Scientists and engineers had to give their best in finding solutions for what one could call ‘emerging modelling challenges’. The idea of bricolage refers to the collaborative efforts of scientists, engineers and policy professionals to cross each other’s knowledge boundaries and achieve their joint objective. As to date, the current version of the model can be acknowledged as a co-production of scientists and policymakers and as a manifestation of cross-boundary scientific knowledge.

When returning to Hunt and Shackley’s [20] theory, we may conclude that in the MIPWA project, interaction and integration of knowledge is fully achieved. When it comes to hybridisation, we can at least acknowledge that the combination of general modelling steps with knowledge about participatory modelling had resulted in a process of collaborative knowledge production for this specific groundwater model.

Perhaps the MIPWA project illustrates best how boundary spanning might be designed and organised. Compared to the AQUATERRA project, the MIPWA project was far less complex for a substantive perspective, and therefore, deliberate boundary spanning activities were relatively easy to organise. In MIPWA, collaborative knowledge production had a clear defined objective, a well-defined

geographical area and a well-understood necessity for cooperation. This resulted in the condition that all participants, whether scientists, engineers or policy professionals, had a stake in constructing a boundary object, i.e. the groundwater model. The model had to have practical use for policymaking and should be a manifestation of scientific advancement. Both stakes tied the communities of scientists/engineers and policy professionals together in their attempt to meet their ambitions.

It is obvious that the conditions for collaborative knowledge production across the boundaries of scientific disciplines were more difficult in the AQUATERRA project. The collaboration between the scientific disciplines in the AQUATERRA project was less rewarding, and the objectives of boundary work for the involved scientists were not designated enough.

5 Capacity Building to Foster Boundary Spanning Activities

Capacity building is an important factor for maintaining connectivity between the communities involved [43–46]. Three forms of capacity building can be distinguished: (1) individual (professionals involved), (2) relational and (3) organisational. We speak of individual capacity (1) when actors individually have gained the attitude, the knowledge and skills to cooperate with others and can perform boundary spanning activities without ‘outside help’. Cooperation often depends of the human factor, crucial for success or failure in complex projects [43, 44]. Connectivity also leads to building relational capacity (2) through intensifying of relations between actors and the higher degree of trust between the actors [46]. A good relation leads again to stronger connectivity. A strong connection also implies a strong organisational embedding (3) of the cooperation. Capacity building on the organisational level depends on the ways how the organisation is capable of adapting their internal processes (managerial, human resources management, etc.) to external cooperation and of facilitating their representatives to connect to the ‘outside world’. These three forms of capacity building are clearly interrelated.

What kind of knowledge and skills is needed by river basin management professionals for a better connection between science and policy? We make the distinction between the professionals who take part in the process, the ‘boundary spanners’ and the professionals who design and facilitate the process. The professionals who take part in the interaction process should have some knowledge about the main concepts and main dynamics on the other side of the boundary, for instance, on how they are being judged in their own organisation. They should have an open and creative mind, as the ability to ‘think with the other side’ is of utmost importance to cross boundaries. Furthermore, they should have a cooperative attitude, as in the boundary spanning processes, cooperation is needed to produce

knowledge together. In ‘science–policy’ interfacing processes, ideally some people are present that can act as boundary spanners. They should know both ‘worlds’ very well, for instance, because they had jobs in both communities. These people are highly valuable for the process as they can intermediate between both worlds and understand both sides very well.

The people who design the science–policy processes and/or facilitate them should be professionals with preferably an education or training in mediation or process management. Organisations can offer people training for the skills needed in boundary spanning activities, for instance, training in cooperation and listening. They also can offer internships to exchange people for a certain period from science organisations to policy organisations and vice versa. Furthermore, organisations who want to facilitate boundary spanning should facilitate learning processes and should keep the internal boundaries as low as possible.

6 Conclusions

River basins are complex systems as relations between the composing parts of the natural system are multiple and (highly) dynamical (see Section A in this book). The interaction with the social system, as addressed in this book Section C, adds to this complexity and makes a river basin a highly complex social–ecological system. Management of river basins, therefore, should encompass policy approaches that integrate different fields of knowledge, multiple stakeholders’ interests and various policy sectors. Problems faced in river basin management are often of an unstructured type. This implies that scientists, stakeholders and policymakers should work together in a collaborative knowledge production process to connect science and policy for river basin management.

Collaborative knowledge production processes require crossing the multiple boundaries that are present between the communities involved in river basin management, which is the focus of boundary spanning theory. For a productive connection between science and policy, four conditions should be fulfilled:

- Joint production of documents, models, etc. (‘boundary objects’)
- People who can combine different fields of knowledge and can attach to different communities (‘boundary spanners’)
- Legitimate and transparent processes to guide boundary spanning activities (boundary spanning processes)
- Joint ownership of the knowledge production process

In river basin management roughly four groups with different roles and dynamics can be discerned: (1) scientists from different scientific backgrounds, (2) stakeholders with different interests, (3) policymakers from different policy sectors and (4) politicians from different political parties. This implicates the presence of many boundaries *between* these groups but also *within* these groups. In the two case studies that were presented in this chapter—i.e. two projects that had

the intention to match science and policy—these multiple boundaries were present. In these cases especially the science–science boundary (the boundary between the disciplines) and the science–policy boundary (boundary between scientists and policymakers) were investigated. From these case studies it was concluded that crossing knowledge boundaries between scientists and policymakers seems to be facilitated by a recognised necessity—by both communities—to work together on the problem. It is further facilitated when the communities use similar language and mental concepts and share joint objectives. Boundary spanning does not emerge spontaneously, but must be actively organised in a boundary spanning process. Sharing the responsibility for and ownership of a collaborative production of policy-oriented knowledge and its application is not self-evident. Availability of resources (time and money), cultural and ‘language’ differences, specialisation and differences in rewarding or performance measurement systems are obstacles for the connection between science and policy [21, 40]. Boundary spanning benefits from clear and tangible objectives, such as the creation of boundary objects. Boundary objects can be a model, a map, a joint document, etc. The collaborative construction of a multidisciplinary model or a joint document by scientists and policymakers, and the assurance of their use, provides the pressure needed for active interaction and integration of knowledge.

In this chapter we made the distinction between the ‘process facilitators’ and the ‘boundary spanners’. Complex research projects that are aimed at crossing (scientific) disciplinary boundaries or the science–policy boundaries will benefit from the involvement of ‘process facilitators’ whose main, and perhaps only, role is to design, organise and implement the boundary spanning process. These process facilitators have preferably an education or training in mediation or process management. The ‘boundary spanners’ are the people who can act at both sides of the boundary, either as a scientist or as a policymaker. They should know both ‘worlds’ very well, for instance, because they had jobs in both communities. ‘Boundary spanners’ should have a cooperative, creative and open mind, as the ability to ‘think with the other side’ is of utmost importance to cross boundaries. Such skills can be trained. Furthermore, science and policy organisations can facilitate the razing of boundaries by offering internships and exchange of people. In this way people can gain knowledge from ‘both worlds’.

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Section D
Synthesis and Recommendations

Synthesis and Recommendations Towards Risk-Informed River Basin Management

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Abstract The health of river basins throughout the world is under pressure from economic activities and a changing climate. Water is necessary for life, agriculture and many industrial production processes. But water is also a receptor for our waste products. In Europe, diffuse pollution from agriculture and our industrial legacy, together with hydraulic engineering for navigation, water supply, hydroelectricity or flood control, is seen as the main factor adversely influencing the quality and ecology of European freshwaters and estuaries. Economic activities affect the

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chemical and ecological status of our rivers, lakes and groundwater and deplete available soil–sediment–water resources. The wide range of economic activities and the ecohydrological complexity of many river basins, in terms of the functioning of the soil–sediment–water system and the links between water quantity, quality and economic activities, make a more integrated management approach to river basins complex and challenging.

As the pressures from both anthropogenic and natural causes on environmental systems increase, it is no longer effective or efficient to deal with one issue at a time, since solving a singular problem often causes damaging impacts on other environmental compartments or in other places. We must consider the consequences of our actions on all parts of the environment in an integrated way and configure these actions to cope with an uncertain future. These challenges demand a different approach in order to achieve actual improvement of the ecological quality of our river basins and thus sustain the goods and services they provide for the well-being of society. Risk-informed management is this new approach. It involves the integrated application of three key principles: be well informed, manage adaptively and take a participatory approach.

Be Well Informed: This implies that a sound understanding of the functioning of the soil–sediment–water system (ecosystem) and its interaction with the social system is the basis to river basin management. A range of European Commission (EC) Framework Programme projects, like AQUATERRA and MODELKEY, have helped deliver, through a range of applied tools, new ecosystem understanding at the site-specific, catchment and river basin scales. For instance, they produced evidence that ecosystem functioning is threatened by contaminants, such as pesticides, nutrients and metals, that are propagated via groundwater pathways from the land surface to rivers, lakes and the sea. Furthermore, there is also evidence that this functioning is threatened by historic contamination mobilised by extreme floods from sediments within rivers, on riverbanks or in floodplain soils. The first generation of river basin management plans (published end of 2009) has only rarely included targeted measures to mitigate these risks. However, the Water

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Framework Directive (WFD, Annex IV) demands that such system understanding should be integrated in the first (to be published in 2015) or subsequent updates of these plans.

Manage Adaptively: Using our best available understanding on how river ecosystems function will certainly improve river basin management. However, when using scenarios—like the Intergovernmental Panel on Climate Change (IPCC) Special Report on Emission Scenarios (SRES)—or other tools to frame plausible trajectories of change, uncertainties will always remain. This is intrinsic to social as well as ecological systems. Systems, especially at larger scales, are extremely complex and dynamic and can respond in non-linear and unexpected ways. We may be able to cope with these uncertainties by applying the concept of adaptive management, characterised as ‘learning-by-doing’ or ‘learning to manage by managing to learn’. In addressing changes in climate and hydrology, the EC Framework Programme project NEWATER delivered guidance to apply the concept in practice.

Take a Participatory Approach: Participatory processes involve stakeholders in management and aim to enable them to exchange their views and opinions on problems and bring their knowledge to the table. By learning together to understand the land–water system in a better way, better solutions can be found. This process of social learning requires a common language. The rapidly developing ecosystem services approach may provide that language. A common understanding of the value of the goods and services that a healthy ecosystem can provide, and how their present poor status due to our actions can be improved, is the key to a new approach to river basin management.

The WFD recognises several of these aspects. It is both risk-informed and ecologically centred. It also recognises the need to balance improvements to water and ecosystem quality with economic benefits including the need to supply water for human requirements. Increasingly, governments also see the need to grow and supply food as part of the balancing act we have to make.

Some examples from practice are already available where integration of these three key principles is attempted. They show very encouraging results and may inspire others. However, it is our conviction that well-designed, coordinated and monitored ‘learning catchments’ (i.e. aimed at stepwise improvement of the effectiveness of measures) are needed to transform our general framing and develop best practice. The International Risk Governance Council’s (IRGC) risk governance framework is recommended as a source of inspiration for the design and execution of such learning catchments.

Keywords Adaptive management • Ecosystem services • Groundwater • Informed decision-making • Participatory approaches • Risk-informed management • River basins • Sediment • Social/ecological systems • Soil • Water

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1 Be Well Informed

A sound, evidence-based understanding of the functioning of the soil–sediment–water ecosystem and of its interaction with the social system is an essential basis for river basin management.

1.1 Conceptual Model as Basis to Understand

A conceptual model—and potentially a subsequent quantitative or semi-quantitative model—is an ideal tool to assist with the development of an improved understanding of the interrelationships between the ecosystem and social system. The most straightforward definition of a model is that it is a simplification of reality, created in order to assist in the clarification and understanding of some aspect of the real world. The key to its success is in achieving an appropriate balance between simplifying a complex reality, making it both easier to understand and applicable to a wider range of circumstances, and preserving the most important relationships in order to obtain results that are a reliable, representative indication of the functioning of the original system [1].

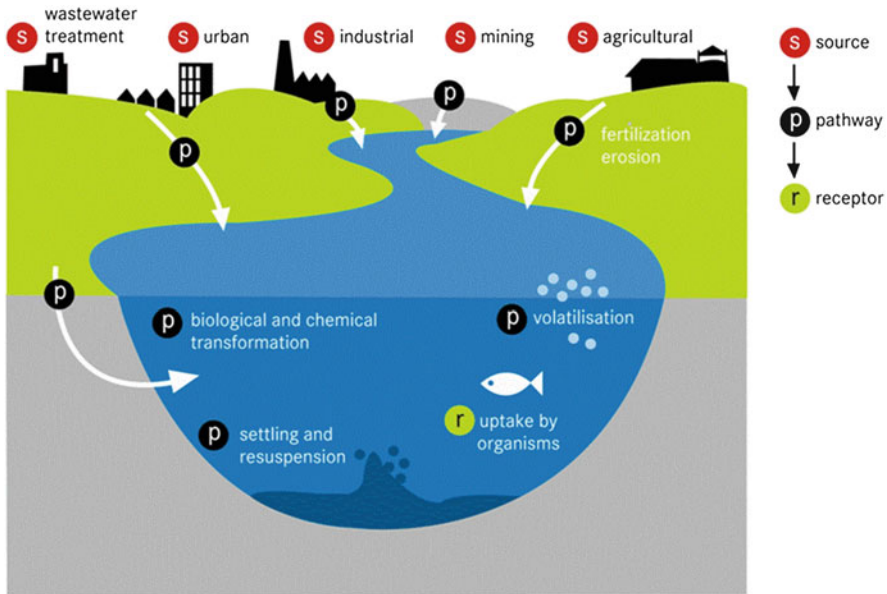


Fig. 1 Risk ‘sources (S)–pathways (P)–receptors (R)’ model

A useful—and often used in the Water Framework Directive (WFD; see [2]) context—conceptual framework for environmental processes and the links between human activities and their impact on the ecosystem functioning is provided by the DPSIR model (see Fig. 1 in [3], this volume). It treats the environmental management process as a feedback loop controlling a cycle consisting of five stages: driving forces (D), pressures (P), state (S), impacts (I) and responses (R).

Another concept for understanding the risks to river basin ecosystems is the risk paradigm which uses the ‘sources–pathways–receptors’ (SPR) linkage (Fig. 1 and see also Fig. 2 in [3], this volume). This SPR model is also at the core of the risk-based land management approach [4]. In any river basin there are numerous sources of risk and pathways where risks can be propagated through the basin towards (potentially) affected receptors, i.e. the goods and services provided by the river ecosystem.

1.2 Understanding the Risks to River Basin Ecosystems

Book Section A described the state of the art in this understanding. The social system interacts with the river ecosystem through the use of land for agriculture, industry and housing and through river engineering, e.g. for navigation, water supply, hydroelectricity and flood control. These interventions combined with

changes in climate and geology result in impacts and risks to the functioning of river ecosystems and specifically to the goods and services they provide (i.e. the 'risk receptors'). These risks can occur on site, or after propagation through the soil–sediment–water system, down- or even up-stream of the origin of the risk source. The main risk sources in river basins include the following: the decline of organic matter in soils and their erosion, compaction and sealing; the contamination, acidification and salinisation of soils and water; hydro-morphological changes in rivers and lakes; and the invasion of alien species. These risk sources give rise to multiple stresses on the biological system, including harmful concentrations of toxicants, eutrophication and oxygen depletion, increasing occurrence of pathogens, and other unfavourable changes in habitat such as changes in sediment quantity and quality and hydrological and morphological alterations. River ecosystems respond sensitively to the action and interaction of these risk sources. The health, reproduction or competitiveness of species is impacted, and the composition of the various species changes. Consequently, the functioning of the ecosystem and its resilience to changing pressures may be adversely affected. Under the WFD, such effects are indicated as an impairment of the ecological status and quantified by determining Ecological Quality Ratios (EQRs) for the Biological Quality Elements (BQEs) fish, invertebrate fauna, phytobenthos, phytoplankton and macrophytes. This WFD focus on biodiversity can be seen as metaphor for the health of ecosystems in general and therefore the functioning of natural processes.

Book Section A also duly describes to date understanding of the risk source–pathway–receptor linkages for the above-mentioned risk sources. By using that knowledge of the system and the currently available tools, we can evaluate if any given society-desired ecosystem service is already impacted (retrospective) or can become at risk when circumstances change (prospective), e.g. due to a changing climate. This system understanding can be used to set priorities and define or refine WFD programmes of measures. In fact, the WFD also requires this as in annex IV where it is stated: 'The first update of the river basin management plan and all subsequent updates shall also include: . . . an assessment of the progress made towards the achievement of the environmental objectives, . . . and an explanation for any environmental objectives which have not been reached' [2]. This first update should be published in the year 2015.

However, we have to realise and accept that science will never be able to explain *all* the complexity of ecosystems and reduce the uncertainties in decision-making to very low levels. Thus, we have to examine how much we really need to know by focusing on the key linkages that drive ecosystems and then manage these accordingly. An important role for the scientific community therefore is to reduce the uncertainties in decision-making for policymakers and river basin managers to tolerable levels—i.e. a risk-informed approach. We will need to manage the risks of not achieving the desired outcomes by focusing on those risk source–pathway–receptor interactions that are most important.

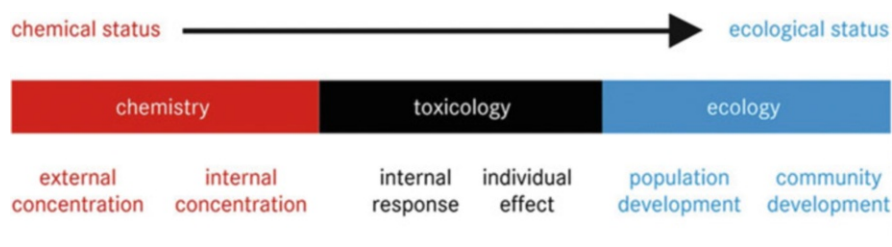


Fig. 2 Toxicant exposure—ecosystem effect chain (see Section A of this book)

1.3 Available Tools for Assessment of Status, Risk and Causes

WFD surveillance monitoring provides data on BQEs, concentrations of priority pollutants and basin-specific chemicals and physicochemical and habitat parameters. These data should provide the basis for subsequent risk assessment and management but also allow hypotheses on potential causes to be developed. Unfortunately, data on the effects of pressures per se are not required by the WFD, and thus, a direct link between biology and chemistry cannot be provided. Powerful statistical tools are available for multi-stress diagnosis (see Section A, [5]), but also causal pathway assessment can help to confirm hypotheses on causative risk source derived from an evaluation of monitoring data. Major elements are effects assessment, cause identification and source identification. Where toxicants are hypothesised as the relevant stressors, the objective of the effects assessment is to unravel the exposure-effect chain (Fig. 2). Effects assessments should include in vivo assays, together with mode-of-action-specific in vitro assays and biomarkers that help to provide more specific information on effect types and possible causes. On a community level, pollution-induced community tolerance (PICT) is a powerful tool for assessing and confirming the effects and causative toxicants [6].

If adverse effects on biological systems can be confirmed, the identification of causative toxicants by effect-directed analysis (EDA) is often the next step of causative pathway assessment. EDA combines bio-testing with a sequential reduction of the complexity of a mixture by fractionation and subsequent chemical analysis ([7]; see Section A). Especially where sediments are the focus of interest, bioavailability may play an important role and should be considered in EDA by bioaccessibility-directed extraction and partition-based dosing techniques. Recent results suggest that considering bioavailability may shift the focus from classical non-polar contaminants such as polycyclic aromatic hydrocarbons to more polar ones. Source identification may be based on the geographical evaluation of monitoring data together with characteristic tracers, pollution patterns and fingerprints. These approaches, known as environmental forensics, have been successfully applied to many compounds, metals as well as organic chemicals (see Section A, [8]).

Risk assessment on different scales should involve a proper exposure assessment with a specific focus on bioavailability and bioaccumulation. Freely dissolved concentrations measured by passive sampling techniques provide good measures for the prediction of internal concentrations and for comparison with critical body burdens. Food web models help to predict internal concentrations in organisms of higher trophic levels such as predatory fish or birds. Since chemical-based risk assessments using simple Predicted Environmental Concentration (PEC) over Predicted No Effect Concentration (PNEC) ratios provide only limited information on actual site-specific risks, higher tier risk assessments focusing on population, community or ecosystem level under in situ conditions have been developed. One of the most promising models may be AQUATOX released by US-EPA.¹ Basin-scale exposure models such as EXPOBASIN [9] considering major processes such as erosion, transport and sedimentation of contaminated particles, together with transport in solution and bioaccumulation, help to assess risks on the basin scale. It becomes obvious that risk assessment may strongly depend on the scale of interest. Water bodies with high dilution capacity often do not exhibit severe local water quality problems. However, since they are very effective in propagating pollution downstream, major problems will occur in sensitive coastal ecosystems and lakes. Water bodies with low dilution capacity often show severe local pollution problems but do not cause major problems downstream, unless a severe flood (re)mobilises the local pollution.

1.4 System Understanding of Importance to River Basin Management

In the first WFD river basin management plans (RBMPs), published in the year 2009, new and unaddressed sources of risk and the need for new knowledge to improve the effectiveness of measures have been identified. Several EC-funded projects, like the sixth Framework Programme Integrated Projects AQUATERRA² and MODELKEY,³ have delivered new ecosystem understanding relevant to support the achievement of the WFD objectives. This understanding is of importance when the WFD RBMPs are updated (first update to be ready in 2015; see Fig. 3 in [3], this volume). Two such examples are the importance of the groundwater–surface water interaction processes in influencing water quality and the mobilisation of historic contamination from soils and sediments as a secondary source of pollution.

One of the main risks of not achieving WFD objectives is the high concentration of nutrients, such as nitrogen and phosphorus, in surface water. Scientific research

¹ See <http://water.epa.gov/scitech/datait/models/aquatox/index.cfm>

² See <http://www.eu-aquaterra.de/>

³ See <http://www.MODELKEY.org>

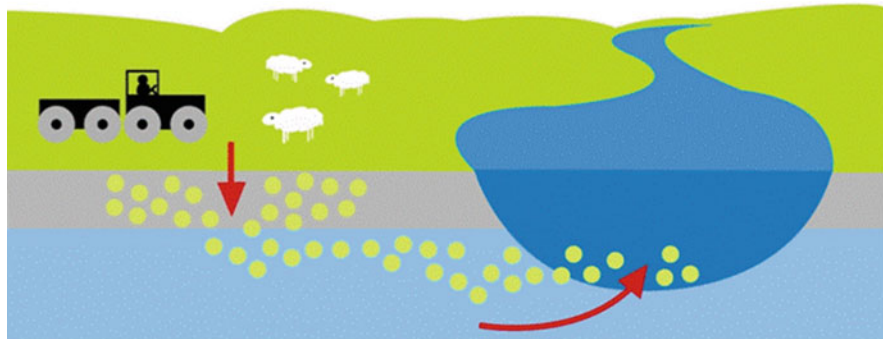


Fig. 3 Groundwater–surface water interaction



Fig. 4 Mobilisation of historic contamination

results, of inter alia AQUATERRA, clearly demonstrate that contaminants such as nutrients and metals are propagated via groundwater to surface water, thus hindering the achievement of surface water quality objectives (Fig. 3). Furthermore, it can be demonstrated that more contaminants will reach surface waters when contaminated land becomes saturated, e.g. after heavy rainfall or flooding. This source–pathway–receptor linkage has only rarely been addressed in the first RBMPs.

Another risk source–pathway–receptor linkage that has seldom been addressed in the first RBMPs is the mobilisation of historic contamination (Fig. 4). The currently available global climate projections anticipate crucial changes in extreme weather conditions, oceanographic conditions and in the water regime of rivers. These changes will, in turn, severely modify basic riverine processes like currents and erosion, thus inducing important physical, geochemical and biological responses. Increased as well as decreased sediment loads (see Section B in this book) may create both environmental and socio-economic problems.

Several recent research studies have concluded that floods and storm flows are likely to lead to the remobilisation of sediments and soils associated with historic contamination, as well as high-energy mass flows (eroded soil and remobilised sediment). Either separately or jointly, both are likely to impact river ecosystems. Changes in precipitation and temperature regimes will also result in changes in the pattern of application of pesticides and biocides and hence will lead to new sediment contamination. Land clearance, agricultural development and urbanisation may further increase soil erosion and hence sediment yield. In contrast, afforestation and soil conservation practices may reduce sediment loads (see Section B).

The system understanding on both these areas, the interaction of groundwater with surface water and the mobilisation of historic contamination, as well as the tools used to achieve that understanding, is available now to:

- Identify and map vulnerable areas where the ecosystem functioning is threatened by contaminants that under wet conditions are propagated via groundwater from land to surface water and/or areas where (extreme) floods mobilise historic contamination from sediments, river banks and floodplain soils.
- Predict how selected measures can help nature to mitigate these threats.
- Design these measures in such a way that supports the implementation of the WFD as well as the Floods Directive [10].

1.5 Framing of an Uncertain Future

It is very likely that considerable changes will occur in the major socio-economic driving forces in our river basins [11]. This will also affect risk source–pathway–receptor linkages. However, we are uncertain of the direction and magnitude of these changes. We can use scenarios (see Section B) to chart several plausible trajectories of change leading to different societies inhabiting landscapes that are not as they look now. Scenarios should be seen as sets of contrasting but internally consistent, plausible descriptions of how the world would look in the distant or near future. For river basin management purposes, the Intergovernmental Panel on Climate Change (IPCC) Special Report on Emissions Scenarios (SRES) may be useful. These IPCC-SRES scenarios are well articulated [12] and have developed into a benchmark input for quantitative modelling (e.g. [13]). However, the strength of exploratory scenarios is also their limitation: whereas every effort is made to be internally consistent and plausible, predictive power is not claimed [14]. The wide span of scenario–outcomes is considered to cover a probable band of possible futures. Envisioning the scenario–outcomes consequences for day-to-day river basin management is a key exercise. Some stress the importance of sets of multiple and contrasting scenarios as

collective images for learning about the future [15]. It is this learning aspect that is important for well-informed river basin management. New scenario-type future visioning is developed in FP7 projects such as PSI-CONNECT⁴ and RESPONSES.⁵

2 Manage Adaptively

We have to learn-by-doing as social/ecological systems are complex and dynamic and can respond in non-linear and unexpected ways.

2.1 Changing Management Paradigm in Water Management

During the nineteenth and twentieth century, our societies changed from small rural communities based on agricultural production to become urban centred. The growth of cities, based on industrialisation, and more intensified primary production (agriculture, forestry and mining) has given rise to challenges. The development of water management can be considered to have progressed through (see Section C, [16]):

- A sanitisation phase (eighteenth century to 1950)—when driven by public health problems, the emphasis was on clean water supplies and safe sewage disposal.
- A pollution control phase (1950s to 1990s)—when the focus turned to water quality and to the control of urban and industrial waste water discharges.

We are currently in a third phase, of sustainable development, where pressures like changes in water quantity, erosion and contamination are being seen to interact in a complex pattern. In this phase it is necessary to take a holistic, integrated approach to environmental issues in order to meet sustainable development aspirations. Risk management decisions are challenged by complexity, uncertainty and ambiguity. Complexity refers to the difficulty to quantify the exact relationship between causes and effects (see Section A), as there may be several paths leading to the same effect. Uncertainty exists if scientific data are absent or if results conflict. Ambiguity denotes the variability of (legitimate) interpretations of evidence by different actors. Depending on the challenge, risk management must adapt to the specific circumstances and context.

While simple risk problems (e.g. water pollution by point sources) can usually be addressed through routine-based procedures and measures (e.g. waste water treatment), complex risk situations (e.g. environmental quality of river basins) require new processes. These processes should be well informed and robust.

⁴ See <http://www.psiconnect.eu>

⁵ See <http://www.responsesproject.eu>

Strategies to address uncertainty might be either aiming at precaution (e.g. policy on hazardous chemicals) or seeking resilience, like ‘working-with-nature’ (e.g. managing floods, contamination and erosion by combining engineered and ecohydrological measures (see [17])).

Developing river basin management towards a longer term, goal driven, strategy at a large geographical scale requires a clear-cut relation with spatial planning and regional planning decisions, as these affect the way the terrestrial part of the basin (the land or the soil) is managed (see Section C, [18]). It will be crucial to anticipate trade-offs. For example, even if land use does not change in the spatial planning sense, the way the land is used as a resource may change dramatically. The discussion in RISKBASE about the change in agriculture from food production to biofuel crops showed that this change is highly relevant for the environment, but for spatial planners, it is less so as the land remains in agricultural use. This example also demonstrates that political and economical driving forces often operate at a larger scale, beyond the control of river basin managers.

2.2 Adaptive Water Management

It is clear that the integrated management of river basins is both complex and challenging (see [3], this volume). The use of the best available understanding on how river ecosystems function (see Section A) will resolve some of this complexity, reduce uncertainty and thus will certainly improve river basin management. However, even by using IPCC-SRES scenarios (see Section B) or other tools to frame several plausible trajectories of change, much uncertainty will remain. This is intrinsic to the complexity of social/ecological systems.

We need to accept and may learn to cope with this uncertainty by applying the concept of Adaptive Integrative Water Management (AWM; see Section C, [16]). This implies a paradigm shift in water management from the current prediction and control to a more flexible ‘learning-by-doing’ management approach. This concept acknowledges the different kinds of uncertainty that there is (and will always be) lack of knowledge on how the different parts of the water system work and interact and how it will change with time. AWM calls for learning cycles through small steps (accompanied by well-designed and targeted monitoring) and avoiding irreversible decisions. ‘Adaptive management is learning to manage by managing to learn’ ([19] and see Fig. 5). A central part of this management approach is to take informed decisions, but it is equally important to improve the capacity of the actors to process this information and draw meaningful conclusions from it.

The EC FP6 Integrated Project NEWATER studied and fostered AWM as a concept to guide theory and practice. Within NEWATER, research was developed to match the needs of water policymakers and practitioners. The core aim was to understand and facilitate change to adaptive strategies for integrated water resource management, and it developed a number of applied tools. Such tools include simulation model scenarios, maps, guidelines and handbooks. These (synthesis)

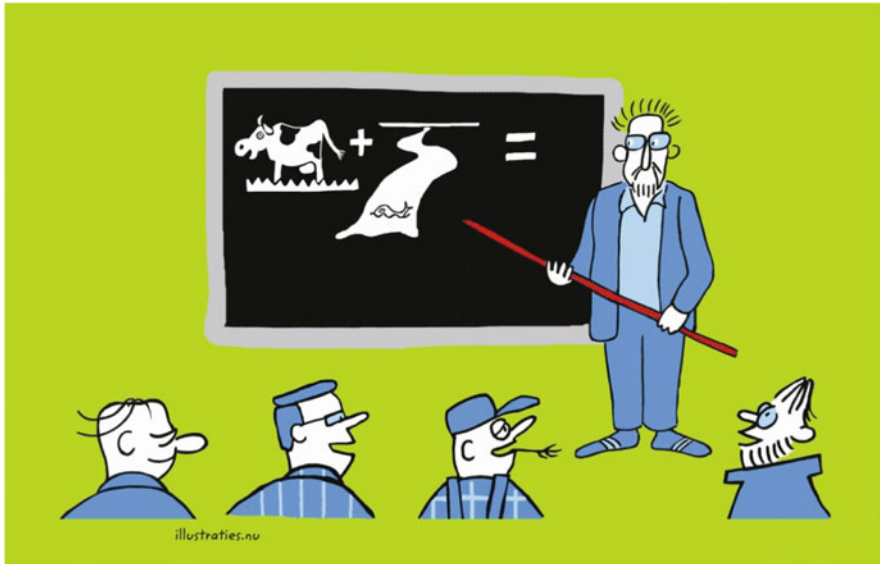


Fig. 5 Adaptive management is, according to Pahl Wostl [19], ‘learning to manage by managing to learn’

products, together with documented experiences of working with AWM (case studies), are available via the NEWATER website.⁶ To further enable dissemination, NEWATER cooperated with WISE-RTD, the European web portal for knowledge transfer in water management. This joint effort has made AWM resources available worldwide.⁷ It offers approved structures and search techniques and links to results from various sources.

3 Take a Participatory Approach

The involvement of stakeholders will improve management, e.g. because they may bring in local knowledge.

⁶ See <http://www.newater.uni-osnabrueck.de/>

⁷ See <http://www.wise-rtd.info/en/info/water-system-perspective-adaptive-water-management>

3.1 Learning Together to Manage Together

A framework for water governance was developed in the 1990s and has been defined by the Technical Committee of the Global Water Partnership [20] as: ‘A process which promotes the coordinated development and management of water, land and related resources, in order to maximise the resulting economic and social welfare, in an equitable manner without comprising the sustainability of vital ecosystems’. The framework emphasis is on the integration of management interests with the values of stakeholders involved in all sectors influenced by ecohydrological services. It requires the inclusion and integration of stakeholders in river basin planning processes. Consequently, the WFD includes public participation as an important element of river basin management planning.

Participatory approaches (see Section C, [21]) enable stakeholders to exchange views and opinions on a certain problem, bringing their knowledge to the table and ‘learning’ together to understand the land–water system in a better way in order to find the best fitted solutions. This is a process of social learning in which new capacities between social agents are developed in the form of learning how to collaborate and understand each others’ roles and capacities. The Adaptive Integrated Water Management (AWM) approach also pursues a strong participatory approach with an emphasis on social learning, which means including local knowledge as well as the use of tools and techniques like (eco-)hydrological modelling. To pave the way for getting AWM into practice requires the promotion of new approaches of social learning as well as a closer cooperation between science and land and water management to integrate knowledge, establish rules and develop solutions.

Useful practical guidance for improving the participation in water management, i.e. for improving social learning, is, for instance, provided by the EC project HARMONICOP in the form of a handbook [22], available in several European languages via the project’s website.⁸

3.2 Who to Involve in Relation to the Risks to Be Managed?

An interesting question is ‘who should be involved in determining the risks to be managed?’ Some decisions are legitimately made by regulatory bodies and governmental agencies or departments in a top-down governmental approach, such as legislation/regulations to limit polluting discharges, or monetary incentives to induce changes in industrial or agricultural practice. However, decisions fraught by uncertainties, by the complexity of a larger natural system and by interests of various groups and stakeholders are best made through a bottom-up governance approach which encourages participation. The International Risk Governance

⁸ See <http://www.harmonicop.uos.de>

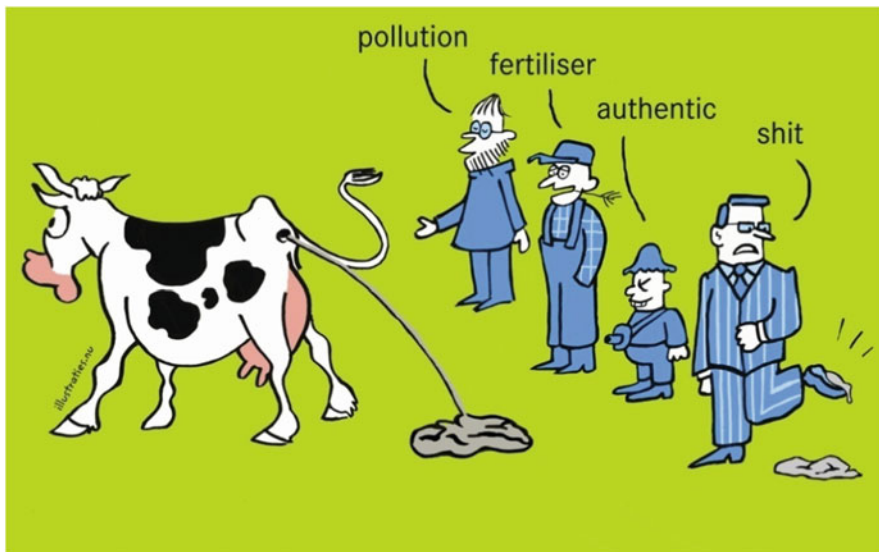


Fig. 6 Communication difficulties originate—according to Quevauviller et al. [26]—to a large extent from the ‘jargon’ used in the different communities

Council (IRGC)⁹ provides clear and transparent guidance [23] on whom to involve for the type of risk to be managed (see Fig. 6 in [16], this volume).

3.3 *Ecosystem Services as Facilitating, Common Language*

With multidisciplinary and multi-stakeholder processes, language is very important. The capability to understand each other is critical in bridging the gap between science and policy making ([24] and see Section C, [25]). Communication difficulties originate to a large extent from the ‘jargon’ used in the different communities ([26] and see Fig. 6). The WFD has developed its own jargon. It enables trans-boundary communication between river basin managers, as all speak the same WFD ‘language’ and face the same objectives and same timelines, and it stimulates communication between the policymakers, implementers and scientists who have a stake in river basin management. However, at the local/regional water management practitioner scale, jargon does not easily facilitate communication with important stakeholder groups such as citizens or farmers. In this context the few first practical experiences using the ecosystem services concept are more promising (see Section C, [27]). Social learning processes can benefit from this concept, as it seems to offer an easier ‘language’ to communicate stakeholder’s

⁹ See <http://www.irgc.org/>

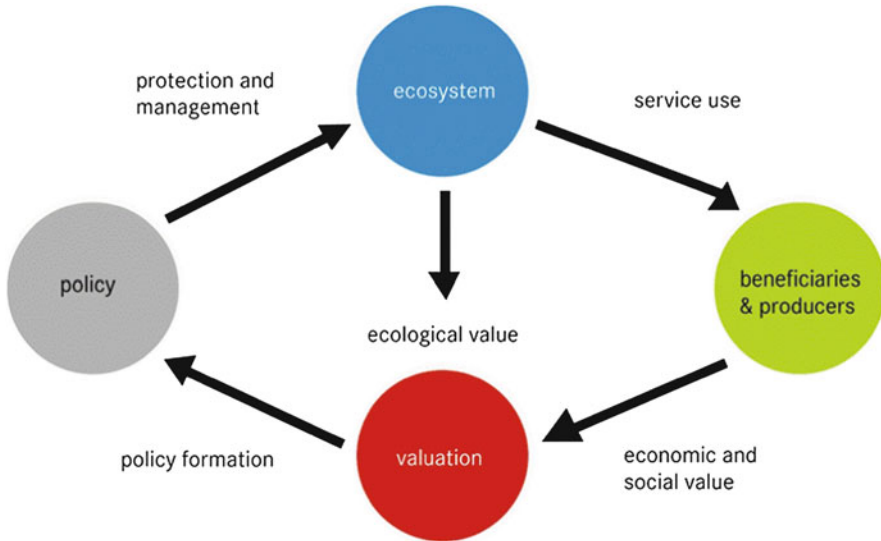


Fig. 7 Ecosystem services organisational framework (Adapted from [33])

positions and interests and to discover common interests in the land–water system. Some state that it may even facilitate the common implementation of different environmental policies. However, these very promising benefits have yet to be demonstrated in practice. Furthermore there is a huge need for guidance, tools and recommendations for the implementation of the ecosystem services approach, which are firmly rooted in practical experience.

The ecosystem services approach is rapidly attracting considerable interest at the global scale throughout environmental policy making, management and science arenas. Many see a huge potential in the application of this approach in support of the development and implementation of policies aimed at the sustainable use of our natural resources. It is an ecosystem-based and dynamic approach which fits the dynamic, non-linear nature of social/ecological systems. It calls for the joint development of system understanding (multidisciplinary scientists, stakeholders, locals, etc.). An organisational framework for the management of ecosystem services is provided in Fig. 7. The sustenance of ecosystem services for human well-being has also become a main European Union environmental policy objective [28]. A common understanding of the value of the goods and services that a healthy river basin ecosystems can provide, and the diminution of these values by our actions, is an important key to a new approach to river basin management.

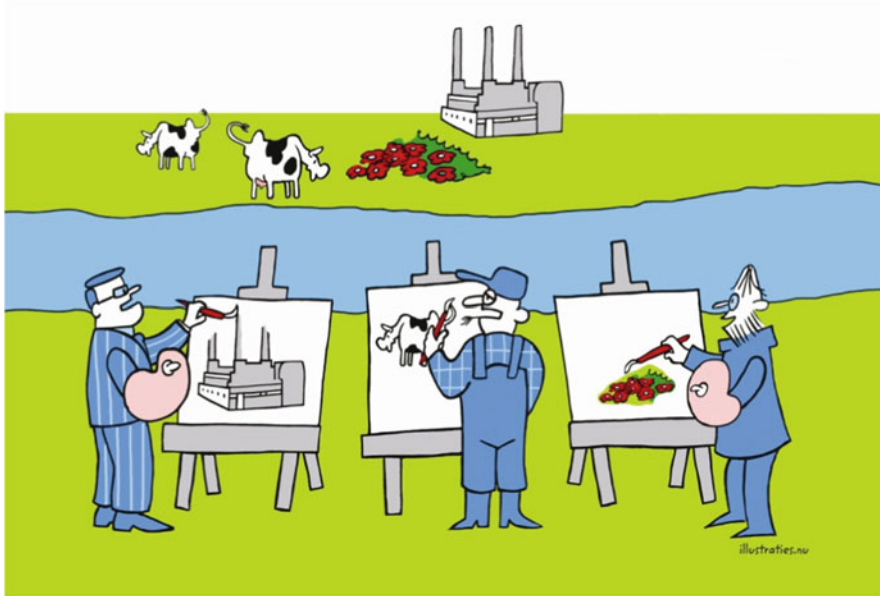


Fig. 8 It is human nature to stay within our own comfort zones

4 Towards Risk-Informed River Basin Management

Integrated management of the river basin soil–sediment–water ecosystem is both complex and challenging. Due to this complexity, command and control, single issue focused management approaches will not help us to achieve actual improvement of the ecological quality of our river basins and thus sustenance of the ecosystem services they provide. A different management approach is needed. This approach involves the integrated application of the three key principles to risk-informed river basin management: be well informed, manage adaptively and take a participatory approach.

4.1 Integrated Application of the Three Key Principles

As the pressures from both anthropogenic and natural causes on environmental systems increase, it is no longer effective or efficient to deal with one issue at a time, since solving one problem often causes another somewhere else. We must consider the consequences of our actions on all parts of the environment in an integrated way and configure these actions to cope with an uncertain future. This will be difficult for all actors in river basin management since it is human nature to stay within our own comfort zones (Fig. 8), at our own scale, focusing on the here and now.

Management of water and land cannot proceed as an isolated activity. Water is by its very nature an agent of connectivity, moving between physical locations and between human populations. Our land and water environments are also intimately connected to social, economic and political factors, at scales ranging from the choices made by individuals to the implementation of international legislation. The management of water is therefore just one (important) element of an interdependent web of environmental, social and economic components. This web is a highly complex and interconnected system, where changes in one component can cascade through the system to result in a series of synergistic and conflicting changes in other components.

These challenges demand a different approach, indeed a different mind-set for policymakers, regulators and all participants alike. The risk-informed management approach promoted by RISKBASE—i.e. integrated application of the three key principles—provides us with a way forward. Risk-informed management is a decision processes that deals with change while minimising the negative consequences of the associated risks. The WFD recognises several of these aspects. It is both risk-informed and ecologically centred. It also recognises the need to balance improvements to water and ecosystem quality with economic benefits including the need to supply water for human requirements.

4.2 Inspiring Examples from Practice

Some examples from practice are already available where the three key principles to risk-informed management have been applied in an integrated way, to a greater or lesser extent. These examples show very encouraging results and may thus inspire others to proceed in this direction. An outstanding example is the management of the Lonjsko-Polje park (see Section C, [16]). This is a floodplain area in the Sava basin in Croatia, a highly dynamic ecosystem which is flooded several times annually. By true integrated application of the three key principles, the park is managed sustainably in conditions of change and unpredictability [29]. Further successful case examples have, amongst others, been provided by the NEWATER project.

Although inspiring examples exist, there has been little connection between the different scales (site and local to international) and little learning from one another. The WFD-related river basin management learning process is still not well developed. To build up learning capacity at the European scale, it is recommended to establish a ‘reflection platform for the WFD’ in which different scientific disciplines, different involved policy areas (like water quality, quantity, spatial planning, soil and land use, etc.), stakeholders and river basin managers will—on a regular basis—reflect on the lessons learned. In this way, the capacity for a ‘science–policy interface’ (see Section C, [25]) in the context of the implementation of the WFD and its current revision process will be built up. Also on the river basin scale, such platforms should be established to encourage exchanges between scientists and water managers. Such platforms may facilitate face-to-face communication [30].

Furthermore, the European Framework Programme that continues in 2014 as ‘Horizon 2020’ and its sister programmes (LIFE, Interreg, etc.) have a crucial role to play in more rapidly synthesising learning and spreading ideas and good practice, since in a rapidly changing world we have little time to change our culture and practice.

4.3 Governance Framework

How to govern the integrated application of the three key principles; what framework to use for risk-informed management? According to the International Risk Governance Council [31], governance refers to the actions, processes, traditions and institutions by which authority is exercised and decisions are taken and implemented. Although it has, to our knowledge, not yet been applied in this context, the IRGC also produced a very promising, well thought out and thus highly recommendable framework (see [16], Fig. 4) for risk-informed river basin management. At the heart of the framework is *communication* (relates to key-principle ‘Participatory’). Then the framework distinguishes between an assessment (understanding: relates to key-principle ‘Informed’) and management (deciding: relates to key-principle ‘Adaptive’) sphere. The *assessment sphere* aims at analysing and understanding a risk—for which the risk appraisal is the essential procedure—is the key activity. The *management sphere* focuses on risk management—i.e. the actions to take to reduce the risk.

IRGC’s approach begins with risk pre-assessment: early warning and ‘framing’ the risk in order to provide a structured definition of the problem and how it may be handled. Risk appraisal develops and synthesises the knowledge base for the decision on whether or not a risk should be taken and, if so, how the risk can possibly be reduced or contained. Risk appraisal comprises both a scientific risk assessment and a concern assessment. Concern assessment comprises a systematic analysis of the associations and perceived consequences (benefits and risks) that stakeholders, individuals, groups or different cultures may associate with a hazard or cause of hazard. This ensures that decision makers account for how the risk is viewed when values and emotions come into play. Characterisation and evaluation is deliberately intended to ensure that the evidence based on scientific facts is combined with a thorough understanding of societal values when making the sometimes controversial judgment of whether or not a risk is ‘acceptable’ (risk reduction is considered unnecessary), ‘tolerable’ (to be pursued because of its benefits and if subject to appropriate risk reduction measures) or, in extreme cases, ‘intolerable’ (to be avoided). Finally, all tolerable risks will need appropriate and adequate risk management. Risk(–informed) management involves the design and implementation of the actions and remedies (i.e. measures; see Fig. 2 in [3], this volume) required to avoid, reduce, transfer or retain the risks. Based on the development of a range of options and a consideration of the most appropriate, risk management decision is taken and put into practice. Risk(–informed) management includes the generation, assessment, evaluation and selection of appropriate risk reduction options as well as implementing the selected measures, monitoring their effectiveness and reviewing the decision if necessary [23].

A key challenge hindering the development of a new approach to river basin management is the current disconnect between the top-down, largely government-led processes operating at regional, national and even international scales (the river basin scale), with the energy and activity operating at smaller, more local, scales (the river reach or water body scale) [32]. The need for a third, and intermediate, level of governance, at the catchment or landscape scale, has not been sufficiently appreciated. This level has several key roles to play: translating national or regional policy and guidance to the people who are implementing activities at the operational level; co-ordinating funding for local activities and developing so-called win-wins; synthesising learning from local practice; and feeding key messages upwards, ensuring that local-scale activities inform the development of new policy and practice. More essentially, the catchment level provides a link to spatial planning, which is well established in the urban and developmental context, but less so in relation to ‘softer’ rural land uses, such as agriculture and forestry. Finally, the catchment is a tangible and relevant scale for the coordination of knowledge related to the integrated environmental, social and economic system operating within its boundaries. We recommend that administrations consider carefully how the disconnect in scales can be bridged.

5 Recommendations

5.1 *Recommendations for River Basin Managers*

- Develop a network of more well-designed, coordinated and monitored ‘learning catchments’ that are, amongst others, aimed at a stepwise improvement of the effectiveness of measures. Apply in these catchments—following the IRGC’s risk governance framework—the three key principles to risk-informed management: be well informed, manage adaptively and take a participatory approach. This is needed to transform our general framing and develop best practice.
- Establish a ‘reflection platform for the WFD’ in which different scientific disciplines, different involved policy areas (like water quality, quantity, spatial planning, soil and land use), stakeholders and river basin managers will—on a regular basis—reflect on the lessons learned.
- Also on the river basin scale, such platforms should be established to encourage exchanges between scientists and water managers.
- Be proactive, on a personal level, in trying to bridge the gap from management to science; more actively seek out the knowledge that you need.
- Award incentives (e.g. prizes) for most effective measure in river basin management. Creating a prize for the most effective measure or tool used in the implementation of the WFD would increase the visibility of best practices, stimulate managers to innovate and provide positive attention for river basin management.

5.2 Recommendations for European Policymakers

- RISKBASE would welcome a European Union policy initiative that encourages Member States and river basin managers to consider the ecosystem goods and services provided by river basins in addition to ecological status per se. This broadening of the scope opens up possibilities for more stakeholder involvement and more scientific input in decision-making. A common understanding of the value of the goods and services that healthy river basin ecosystems can provide, and the diminution of these values by our actions, is the key to a new approach to river basin management.
- Administrations please consider carefully how the disconnect in scales can be bridged.
- Be proactive, on a personal level, in trying to bridge the gap from policy to science: seek more actively for the information that you need.

5.3 Recommendations for Research Funders

Make funds available to:

- Monitor measures implemented in the first RBMPs as well as the Floods Directive and characterise ecological, economic and socio-economic reactions of the natural and the social system. Nested demonstrations should focus on regional scales, developing understanding of how to combine and optimise engineered solutions (including the resilience of infrastructures) and ‘working-with-nature’ approaches (e.g. ecohydrology, phytotechnologies) at the catchment scale. All measures should be assessed in relation to wider environmental impacts and feasible climate adaptation strategies.
- Provide consistent concepts and applicable tools to identify relevant stressors in multiple stressed environments and to better understand the interaction of different stressors. This will help water managers who are frequently confronted with a multiple stressor situation of toxic pressure, organic pollution, eutrophication, adverse hydro-morphological conditions, pathogens and invasive species to develop effective programmes of measures for improving the status of aquatic systems.
- Provide tools for the identification of emerging pollutants that pose a risk to aquatic ecosystems. The very large and increasing amount of chemicals and environmental transformation products, the general tendency towards higher polarity and complexity of emerging compounds and the increasing relevance of compounds with highly specific modes of action (e.g. pharmaceuticals) demand more innovative approaches and techniques for isolation, structure elucidation and effects assessment.
- Compare and analyse (through joint studies, experiments and discussions) experiences and insights from ecosystem services oriented, or supporting, activities. Then target experiences and insights from pilot, place-based

ecosystem services case studies and focus on the relation between ecosystem services and sustainable use and management of our natural resources, especially in the in the context of spatial planning and river basin management. The results could be an overview of best practices and the delivery of practical guidance (valuation, trade-offs, stakeholder participation, adaptive governance), tools (mapping, indicators, index, modelling) and recommendations for the implementation of ecosystem services in environmental and spatial policy making and implementation/management.

- Better understand the effects of global change on sediment quality and quantity processes and the anticipated, resulting impacts on river ecosystems. Research projects should develop, and to the maximum possible extent apply to real-world situations, novel approaches aimed at understanding, assessing and forecasting and where feasible intervene with these effects and resulting impacts. The expected impact is a provision of an interdisciplinary scientific knowledge base, including models, tools, risk-assessment approaches, scenarios, databases, etc. This is needed to inform river basin management policy and decision-making, amongst others by indicating feasible measures that help us to adapt to the global change induced effects on sediment quality and quantity and the resulting impacts on river ecosystems.
- Develop basic courses on integrated water management/policy making for stakeholders.
- Train professionals (boundary-spanners) who can bridge the ‘science–policy’ interface and educate a new generation of managers and scientists.
- Facilitate face-to-face communication. Make budgets available for scientists to disseminate their knowledge to those who need it (e.g. establish a fund that enables scientists to participate in meetings of river basin management-related policy or practice groups, such as river basin committees).

5.4 Recommendations for Scientists

- Be proactive, on a personal level, in trying to bridge the gap from science to policy/management and communicate transparently on the uncertainties/error margins that go with your scientific findings.
- Realise that environmental management poses more challenges to scientific research than underpinning of environmental quality standards.

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