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WISER

Water bodies in Europe: Integrative Systems to assess Ecological status and Recovery

DELIVERABLE

Deliverable D7.2-6: End-user summary and booklet

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RE	*Restricted to a group specified by the consortium (including the Commission Services)*	
CO	*Confidential, only for members of the consortium (including the Commission Services)*	

Content

Non-technical summary

From March 2009 until February 2012, more than 120 scientists of the EU-funded project WISER addressed major knowledge gaps in the assessment and management of Europe's surface waters. The outcome comprises new assessment approaches for lakes, transitional and coastal waters. Assemblage metrics were developed and tested for reliability and uncertainty of their response to different environmental stressors, some of which were also useful for intercalibration. With regard to aquatic ecosystem management, the response of aquatic assemblages to mitigation and restoration were examined, while potential effects of climate change were explicitly involved in this examination. While the overall outcome is being presented publicly at www.wiser.eu, this document aims at presenting the conclusions that may be drawn form the WISER outcome, also in light of the state-of-the-art as reported in the contemporary literature. These conclusions of the WISER consortium are far from being complete, but may provide a concise collection of helpful key messages for the practitioners in river basin management to better inform future management and restoration of Europe's waters. For more detailed results and conclusions, the reader is invited to access the products and reports as indicated at the end of this document.

Summary

In brief, the major outcome of the WISER project may be summarised with the following statements:

- **There are almost 300 different aquatic bioassessment methods** that are currently being used in Europe.
- A huge amount of the aquatic ecological monitoring has been compiled into several **WISER databases,** which sustain beyond the project duration and, thus provide an invaluable data source for corresponding future research and management projects.
- **WISER has developed new pressure-sensitive bioassessment methods** for robust, comparable and concise assessment and monitoring of the ecological status of lakes, transitional and coastal waters in Europe that allow of the detection of the impacts and improvements due to **pressure-targeted management measures**.
- A number of **common metrics** have been developed for biological quality element-specific assessment of the ecological status of water bodies that are used to compare and harmonise the outcomes of the indicator-based assessments. The common metrics help ensure that there is a same level of ambition in the setting of the ecological water quality targets (on the Ecological Quality Ratio level) in Europe. This work supported the **intercalibration exercise** and shall facilitate comparative and consistent assessment of surface water bodies across the river basins and countries in the future
- WISER has developed tools and methods to assess the various levels of **uncertainty** in the assessment of the ecological status of water bodies with regard to the location and timing of sampling, lab processing and data analysis.
- WISER has developed approaches and tools to tackle the assessment of ecological status in water bodies impacted by **multiple pressures**; their influence across water categories has been investigated and compared with regard to the assessment and management of water bodies, also in light of climate change.
- WISER has developed approaches and modelling methods to address the critical questions in the contemporary **management and restoration** of aquatic resources.
- WISER has analysed and compared the response of different Biological Quality Elements stressors across water categories and suggests rules for the combination of results into a **holistic assessment of the impacts of degradation and management**.

Introduction

Aquatic ecosystems in Europe have been heavily impacted by human activities since centuries. There is a wide variety of aquatic ecosystems (e.g. lakes, estuaries) in an equally variable range of different ecological states, from nearly pristine Alpine and Boreal rivers and lakes to heavily degraded river systems alike open sewers. Recent European policies target a good ecological status for all surface waters, i.e. water bodies need to be assessed by comparison with a reference quality target and, if the quality is below the target, they need to be restored until the target status is being achieved. For many aquatic ecosystem types, ecological assessment systems have been developed; river basin management plans outline the required restoration measures.

The EU WISER project (Water bodies in Europe: Integrative Systems to assess Ecological status and Recovery; March 2009–February 2012) has been supporting the implementation of the Water Framework Directive by developing tools for the integrated assessment of the ecological status of European surface waters. The project analysed existing data from more than 90 databases compiled in previous and ongoing projects, covering all water categories, organism groups and environmental stressor types. Field-sampling campaigns were carried to supplement the data on lakes and coastal systems. The obtained data has been used to test and complement existing assessment schemes with a focus on uncertainty affects on classification strength.

Besides integrated assessment, WISER has specifically addressed biological recovery processes using large-scale data to identify linkages between pressure variables and ecosystem responses. A variety of modelling techniques have been applied to more than 20 selected case study river basins all over Europe to evaluate the efficacy of restoration. The aim was to provide guidance for the next steps of the implementation of the WFD, while working in close cooperation between the project partners and end users (coordinators of Geographical Intercalibration Groups, River Basin Managers, and Environmental Ministries and Agencies).

The activities of the WISER project covered data base and guidance development, development and intercalibration of biological indicators and assessment tools for lakes, transitional and coastal waters, development of modelling tools for analysis of climate change and land use on restoration of water bodies, and integration of the assessment based on different biological quality elements into an integrated assessment of the ecological status of water bodies.

The End-user summary report presents the key results that are particularly relevant for the current work of the river basin managers in the revision of the characterization and classification of EU water bodies as part of the on-going river basin management cycle (2012–2018). The aims of the end user summary report are to i) inform the end users about the major outcome of WISER, ii) to provide the main lessons learnt as key messages and iii) to provide all links and references to further information on the website.

Each section begins with the short key message and the evidence supporting the conclusion based on the project results with some figures or table to illustrate the outcome. The implications for management are shortly summarized after each message with references for further and more thorough information available in the deliverables and publications of the WISER project.

We hope that the report will function as a useful source of information for river basin managers and other interest end users as well as an appetizer and a catalogue of the scientific work supporting the assessment of ecological status of surface waters as well as the evaluation and selection of the measures targeted to improve ecological status and for restoration of the water bodies.

Key messages from the WISER project

The presentation of key messages follows a logical structure of tasks, as they are organised in the practical implementation of the WFD. The start is set by messages on the generation and compilation of data, followed by the messages on the assessment of ecological status and its inherent uncertainty, handing over then to the messages of management and restoration measures in aquatic ecosystems, and ending with the potential adverse effects of global and climate change.

The messages are structured into nine chapters:

- Overview of available methods and data *(WISER Module 2)*
- Assessment and monitoring of lakes in Europe *(WISER Module 3)*
- Assessment and monitoring of transitional and coastal waters in Europe *(WISER Module 4)*
- Management of rivers in Europe *(WISER Module 5)*
- Management of lakes in Europe *(WISER Module 5)*
- Management of transitional and coastal waters in Europe *(WISER Module 5)*
- Uncertainty in water body assessment *(WISER Module 6)*
- Integration of different Biological Quality Elements *(WISER Module 6)*
- Management across water categories *(WISER Module 6)*

1 Overview of available methods and data

Intercalibration is a fundamental prerequisite to compare the results of hundreds of bio-indicator systems in Europe

European countries currently use nearly 300 different methods to classify the ecological status of their surface waters. The methods mainly consider species abundance and sensitivity and focus on the impacts of organic pollution and eutrophication. The intercalibration exercise aimed at harmonising the national classifications in order to provide common denominators for the comparison of individual national results within a European context of ecological status classification

Evidence

The WISER project reviewed 297 assessment methods, based on a questionnaire survey sent to water authorities in all Member States and additional countries that are being implementing the Water Framework Directive. Twenty-eight countries reported on methods applied to rivers (30% of all assessment methods), coastal waters (26%), lakes (25%) and transitional waters (19%). More than half of the methods are based on either macroscopic plants (28%) or benthic invertebrates (26%); in addition, phytoplankton (21%), fish (15%) and phytobenthos (10%) were assessed (Figure 1).

About three-quarters of the methods identified organisms to species-level while in particular phytoplankton-based methods used class- or phylum-level, or included no taxonomic information. Out of nine metric types distinguished, river methods used more sensitivity and trait metrics while for other water categories abundance metrics were the most common. Fish-based methods had the highest number of different metrics. Fifty-six percent of the methods focussed on the detection of impacts of eutrophication and organic pollution pressures. The most commonly used organism groups in decreasing order were phytoplankton > phytobenthos > macroscopic plants > benthic invertebrates > fish. The order was almost reverse for the detection of the impact of hydrological or morphological deterioration mainly targeted in rivers and transitional waters: fish and macroscopic plants > benthic invertebrates > phytoplankton > phytobenthos.

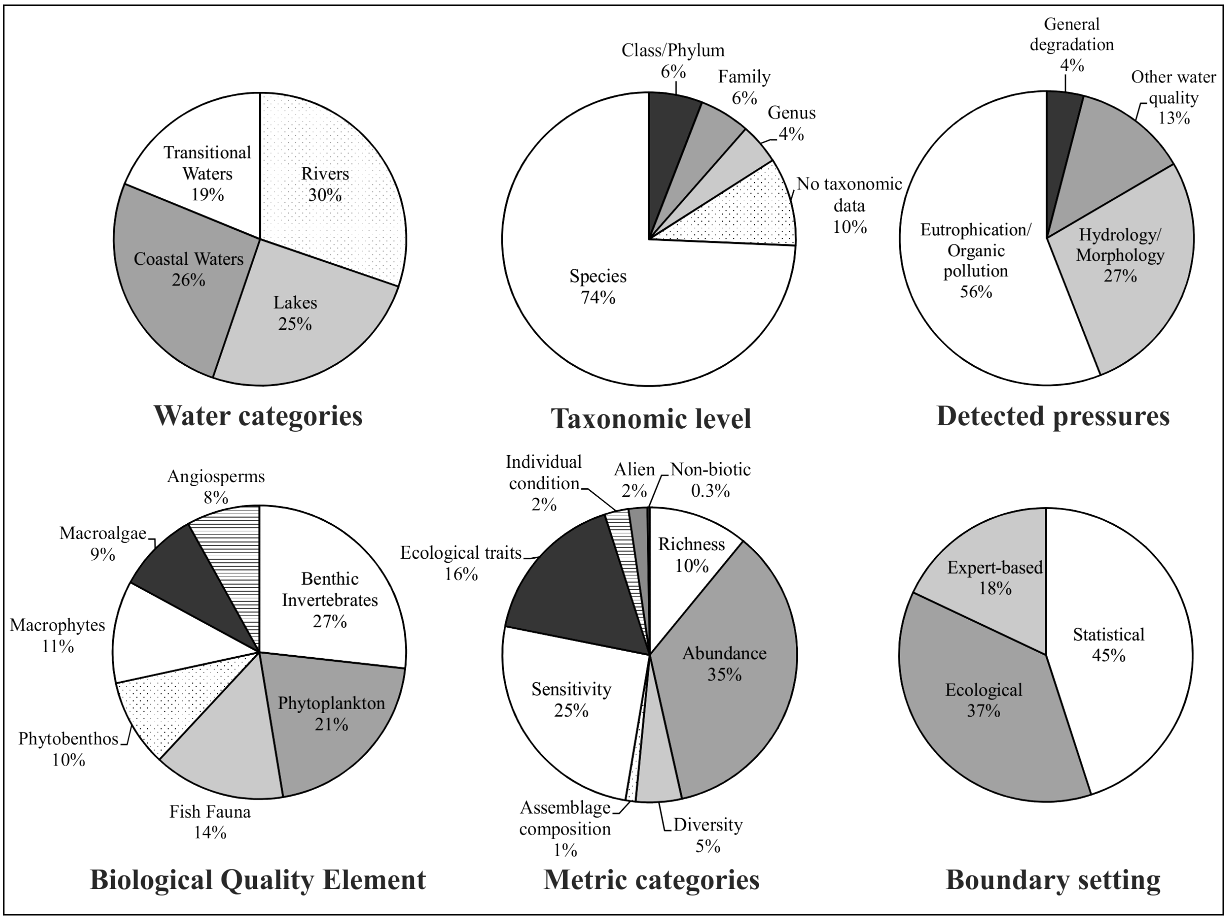

Figure 1. *Results and distribution of the characteristics of the 297 national assessment methods reported by 28 countries and reviewed by the WISER project (based on a questionnaire survey sent to water authorities in all countries implementing the Water Framework Directive).*

The pressure-impact relationships were tested empirically for two-third of the methods, mostly for rivers, lakes and coastal waters, while the methods for transitional waters were least validated. The strength of the relationships differed significantly between organism groups and water categories. The correlation coefficients generally covered a broad range (<0.4 to >0.8). The strength of the relationships decreased in order: Phytoplankton > macroscopic plants > benthic invertebrates > phytobenthos and fish fauna, and for the water categories in order: Coastal waters > lakes > transitional waters > rivers. Status boundaries were mostly defined using statistical approaches.

The overview of the WFD intercalibration exercise revealed that the assessment methods for the following biological elements are almost fully intercalibrated: Phytoplankton and macrophytes in lakes, and benthic invertebrates, phytobenthos and fish fauna in rivers. Intercalibration has not been fully completed for the remaining biological elements / surface water types.

Implications

The multitude of aquatic bioassessment methods used for the assessment of the European surface waters is perplexing. It is questionable if the methodological patchwork allows for comparable ecological status classification across Europe. Nevertheless, the WFD intercalibration exercise has provided methodology to check the comparability of results and consistency in classifications. However, despite of more than 10 years of development, there are not fully set of methods for all quality elements in all categories of surface waters. Also the intercalibration still need to be continued in the future to ensure comparability of new methods and improvements of the existing methods.

The outcomes of the pressure-impact analyses conducted to test the national methods are promising, but more effort is needed in order to develop a comprehensive understanding of the human pressures detected by the individual methods. In particular there is a need to better understand cause (human pressure) - effect (metrics or indicators) relationships for highly integrative biological elements such as fish or plants. Such models would help to choose the right management actions to improve the quality of the vegetation and fish fauna that are important for people using lakes, rivers and coastal waters for recreation and fishing.

The boundaries in the ecological classifications were not often based on ecological principles. The ecological targets are generally based on statistical distributions rather than on meaningful ecological changes in ecosystem functions and in the biological communities. The challenge remains to incorporate ecological components and functions into the national systems of ecological water quality classifications.

Further reading

Birk, S., Bonne, W., Borja, A., Brucet, S., Courrat, A., Poikane, S., Solimini, A. G., van de Bund, W., Zampoukas, N., Hering, D. (2012). Three hundred ways to assess Europe's surface waters: an almost complete overview of biological methods to implement the Water Framework Directive. Ecological Indicators, 18, 31-41.

Birk, S., Bonne, W., van de Bund, W., Poikane, S., Zampoukas, N. (2012). Europe's quest for common management objectives of aquatic ecosystems. In: Schmidt-Kloiber, A., Hartmann, A., Strackbein, J., Feld, C.K., Hering, D.: Current questions in water management. Book of abstracts to the WISER final conference - Tallinn, Estonia, 25-26 January 2012: 28-29. (Downloadable file available at http://www.wiser.eu/meetings-and-events/final-conference/abstracts/)

The WISER Central Database is of great value for future research

Key message

A large number of datasets from rivers, lakes and coastal waters have been compiled and stored in the WISER Central Database (CDB). Data for all biological quality elements and all water categories are available from the CDB in a harmonised format. More specifically, the CDB can be used to combine (1) biological data with environmental pressure data (chemistry etc.), (2) data for different biological quality elements, (3) data from different water categories. These data are accessible both for WISER partners and for other scientists. The conditions for use of WISER data depend on the intellectual property rights (IPRs) stated by each data owner. Detailed information on all WISER datasets, including IPR information, is available in the WISER metadatabase (http://www.wiser.eu/results/meta-database/).

Evidence

The WISER Central Database contains biological and other environmental data from 26 European countries (Figure 2). The WISER field campaign in 2009/2010 resulted in ca. 8,000 biological samples from ca. 1,000 sampling stations in lakes and coastal/transitional waters from 14 countries, containing altogether 40,000 records of species abundance. In addition, the CDB contains existing datasets from previous research projects, national monitoring etc., containing more than 1,500,000 records of species abundance and 900,000 other environmental observations from ca. 75,000 sampling stations in rivers, lakes and coastal/transitional waters. This extensive database can be very useful also for future research related to river basin management, as well as more general research in e.g. aquatic ecology, biodiversity and environmental stressors.

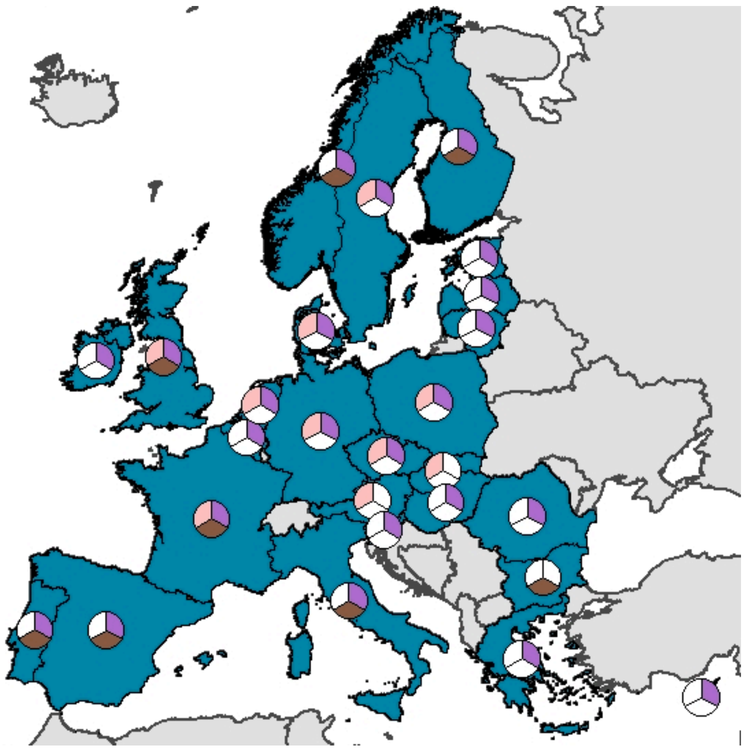

Figure 2: Geographical coverage of the WISER Central Datbase (CDB). Countries represented in the CDB are coloured blue. Coloured pie sectors indicate data from lakes (lilac), coastal/transitional waters (brown) and rivers (pink) (white n/a).

Implication

Data from the WISER CDB has been used for the many publications (http://www.wiser.eu/results/publications/) as well as for the lake load response tool (http://lakestate.vyh.fi/) for planning of river basin management, and for comparison of responses to stressor gradients across different biological quality elements in rivers and lakes. Because of close collaboration between WISER scientists and GIGs (Geographical Intercalibration Groups), WISER data have also been used extensively in the WFD intercalibration exercise. However, the WISER data may have limited usability for some purposes due to uneven representation of the different countries, water categories and biological quality elements. Due to the intellectual property rights stated by the data owners, only the project partners can download the WISER data. Other persons who are interested in using these data are encouraged to contact the WISER Data service (wp2.1@wiser.eu) or other WISER partners for scientific collaboration. The publicly available WISER metadata search tool (http://www.wiser.eu/results/meta-database/) will provide contact information to the relevant WISER partners for each dataset.

Further reading

Moe, S. J., B. Dudley, R. Ptacnik (2008). REBECCA databases: experiences from compilation and analyses of monitoring data from 5000 lakes in 20 European countries. Aquatic Ecology 42:183–201 (and references within).

2 Assessment and monitoring of lakes in Europe

The reliable assessment of the impact of different lake stressors requires the use of different Biological Quality Elements

Different Biological Quality Elements (BQEs) are being used to assess the ecological status of lakes in Europe: fish, benthic invertebrates, macrophytes/phytobenthos and phytoplankton. The different responses of these BQEs to different stressors require the use of several BQEs in order to assess the multiple impacts by multiple stressors (Table 1). In brief:

- Phytoplankton and macrophytes show strong responses to eutrophication pressure.
- Littoral benthic invertebrates clearly respond to morphological shoreline degradation, and macrophytes to water level fluctuations.
- Fish assemblages show less clear signals to individual pressures, but may be good indicators of climate warming.

Table 1: Overview of general stressor-response relationships of lake BQEs (indicated as correlations according to Pearson's R^2 or Spearman's rho).

BQE	Pressure and indicators	Best common metrics	R^2	Rho
Phytoplankton	Eutrophication (TP)	Chlorophyll-a	0.63	
		PTI (taxonomic composition)	0.67	
Macrophytes	Eutrophication (TP)	ICM (taxonomic composition)		
	HyMo (water level fluctuations)	WLi (taxonomic composition) (NO+FI)	0.77	
Benthic fauna (littoral)	Eutrophication	MMI	0.40	
	HyMo (shore modifications)	MMI (LIMCO) (DE+DK)		0.70
		MMI (LIMHA) (DE+DK)		0.72
Fish fauna	Eutrophication	MMI (CPUE< BPUE, OMNI)	0.25	

BQE= biological quality element

Evidence

Phytoplankton is highly sensitive to eutrophication pressure, based on the statistical analyses using all regional data sets (Table 2). The best common metric, with high sensitivity, is the Phytoplankton Trophic Index, which includes both taxonomic composition data as well as chlorophyll a. These two metrics have been combined into a common metric for the intercalibration of phytoplankton methods with successful results in both the Northern GIG and the Central-Baltic GIG. Cyanobacterial blooms are common in all eutrophied lakes across Europe. The risk that the WHO health alert threshold for cyanobacteria biovolume (1-2 mg/l) would be exceeded is 10% at a total-P concentration of 20 $\mu g\ L^{-1}$ and 30% at 40 $\mu g\ L^{-1}$.

The best metric for macrophytes indicating eutrophication pressure is the intercalibration common metric for taxonomic composition (ICM; Table 2), which has also been used for intercalibrating macrophyte methods in the same GIGs.

Other metrics for phytoplankton and macrophytes responding to eutrophication have also been tested within WISER, such as cyanobacteria abundance and macrophyte growing depth. These metrics also show highly significant relationships with nutrient pressures and may be easier to communicate to the public and water managers. A shift from macrophytes to cyanobacteria highlights an important functional shift that can greatly affect the use of freshwaters for recreation, swimming or as a reservoir for potable water.

Table 2: Overview of metric sensitivity to pressure for biological quality elements in lakes. GIG = Geographical Intercalibration Group. CB GIG = Central European and Baltic region, NGIG = Northern region, MGIG = Mediterranean region. GAM = generalised additive model. The other regressions are linear models. N = number of lake-years. Sensitivity has been assessed from regression analyses of dose-response curves along pressure gradients using large scale pan-European datasets from > 1000 lakes from 21 countries.

Metric	Metric description	Pressure	r^2	GIG or country	p	N
Phytoplankton						
Chla	Chlorophyll a	Eutrophication (Total-P)	0.63[1]	All, but mainly NGIG & CBGIG	<0.001	16949
PTI	Phytoplankton Trophic Index	Eutrophication (Total-P)	0.67 (GAM)	All, but mainly NGIG & CBGIG	<0.001	2287
Cyano bloom intensity	Cyanobacteria biovolume	Eutrophication (Total-P)	0.34 (GAM)	All, but mainly NGIG & CBGIG	<0.001	1710
SPI	Size Phytoplankton Index	Eutrophication (Total-P)	0.23 0.34 0.19	CB GIG N GIG M GIG	<0.0001 <0.0001 <0.05	122 77 29
MFGI	Morpho-Functional Group Index	Eutrophication (Total-P)	0.33 0.05 0.38	CB GIG N GIG M GIG	<0.0001 <0.05 <0.001	122 77 29
FTI	Functional Traits Index (mean of SPI and MFGI)	Eutrophication (Total-P)	0.39 0.22 0.50	CB GIG N GIG M GIG	<0.0001 <0.0001 <0.0001	122 77 29
J'	Evenness	Eutrophication (Total-P)	0.19 0.07	N GIG CB GIG	<0.001 <0.001	716 559
Macrophytes						
ICM	Intercalibration Common Metric	Eutrophication (Total-P)	0.52	All, but mainly NGIG & CBGIG		
EI	Ellenberg Index of taxonomic comp.	Eutrophication (Total-P)	0.47	All, but mainly NGIG & CBGIG		
Cmax	Maximum colonization depth (abundance proxy)	Eutrophication (Total-P) (Chlorophyll) (Secchi depth)	0.31 0.31 0.58	All, but mainly NGIG & CBGIG	<0.0001 <0.0001 <0.0001	478 612 475
WIc	Water level Taxonomic comp index	Hydromorphological changes (water level fluctuations in ice-covered lakes)	0.77	NGIG (NO+FI)		26
Benthic fauna						
MMI	Multimetric Index	Eutrophication (Total-P)	0.40 (whole lakes)	CB-GIG	<0.001	161
MMI	Multimetric Index	Morphological alterations and Eutrophication (shore line modifications, landuse and TP)	0.53	CB-GIG	<0. 001	161
MMI	Multimetric Index	Morphological alterations (shore line modifications)	0.49	All, mainly CBGIG	??	44
LIMCO	Littoral Invertebrate Multimetric Index based on Composite Sampling	Morphological changes of lake shore	0.70* 0.49* 0,44* 0.47*	DE+DK Italy SE+FI IE+UK		
LIMHA	LIMI based on Habitat-specific Sampling	Morphological changes of lake shore	0.72* 0.40* 0,44* 0.71*	DE+DK Italy SE+FI IE+UK		

[1] For Chlorophyll a and Total-P as a proxy for eutrophication pressure the correlations coefficient (r^2) was 0.63, for lakes with TP<100µg/l)

Fish						
MMI	Multimetric Index consisting of BPUE, CPUE and OMNI	Eutrophication (non-natural land cover)	0.25	All	<0.001	445
BPUE	Biomass per unit effort	Eutrophication (non-natural land cover)	0.19	All	<0.001	445
CPUE	Catch per unit effort (number of individuals)	Eutrophication (non-natural land cover)	0.18	All	<0.001	445
OMNI	Relative number of omnivorous individuals	Eutrophication (non-natural land cover)	0.16	All	<0.001	445
		(Total-P)	0.18	All	<0.001	445

*Value represents Spearman's rank correlation coefficient Rho; in four biogeographical regions where different metrics correlated best with the stressor index.

Macrophytes also responded clearly to hydromorphological pressure, in terms of water level fluctuations in regulated lakes in the Northern countries. The macrophytes water level fluctuation index (Wlc) has a clear threshold response concerning the indicator taxa e.g. *Isoetes* corresponding to ca. 3.5 m water level fluctuations. Thus, this metric is a very promising tool to define ecological potential in heavily modified water bodies.

Littoral macroinvertebrates respond clearly to modification and degradation of shoreline habitats in lakes. Two new multimetric indexes have been developed within WISER, including several single metrics, such as the number of taxa of mayflies, stoneflies, caddisflies, water beetles, mussels, dragon-flies, relative abundance of the functional groups like gatherers or collectors, or classes of chironomids, and Margalef diversity. Number of Macroinvertebrate species and fraction of individuals feeding on particulate organic matter were lower at both intermediately and strongly modified lake margins than at unmodified margins in 64% of 44 lakes. Another multimetric index based on littoral macroinvertebrates also responds to a combination of pressures from eutrophication and morphological modifications (Table 2).

For fish, the best metrics to assess eutrophication impacts are biomass per unit effort (BPUE) (r^2 = 0.19), catch per unit effort (CPUE) (r^2 = 0.18) and relative number of omnivorous individuals (OMNI) (r^2 = 0.18), but none of these have been used for the final stage of intercalibration of national methods. Fish has however been shown to respond to climate warming with cold-water species like arctic char being pushed further north and towards higher altitudes, while warm-water species like many cyprinids increase in dominance and widen their biogeographical range. Warm lakes were dominated by small-sized individuals, whereas in cold lakes the relative proportion of large-sized fish increased. The dominance of small fish in warm lakes was primarily the consequence of an increase in juvenile fish.

Implications

Operational monitoring and assessment of ecological status in lakes should be based on the most sensitive quality elements to different pressures. WISER evidence supports that the botanical BQEs (phytoplankton and macrophytes) are well suited to assess lake eutrophication impacts. Effects of measures to restore eutrophied lakes can only be seen when the total phosphorus concentration is reduced to less than 100 µg/l. For hydromorphological pressure, macrophytes respond well to water level fluctuations in northern regulated lakes and may thus be used as a tool to set environmental goals for heavily modified water bodies. Littoral macroinvertebrates have been shown to sensitively assess impacts of morphological alterations to lake shores. Fish should be monitored to assess impacts of climate warming.

Further Reading

More detailed analysis and results are being presented in various Deliverables and are available for download at http://www.wiser.eu/results/deliverables/.

Brauns, M., Garcia, X.-F., N. Walz & M.T. Pusch 2007. Effects of human shoreline development on littoral invertebrates in lowland lakes. Journal of Applied Ecology, 44, 1138-1144

Emmrich, M., Brucet, S., Ritterbusch, D., Mehner, T., 2011. Size spectra of lake fish assemblages: responses along gradients of general environmental factors and intensity of lake-use. Freshwater Biology 56: 2316–2333

Kolada A, Hellsten S, Søndergaard M, Mjelde M, Dudley B, van Geest G, Goldsmith B, Davidson T, Bennion, H, Nõges P & Bertrin V, WISER Deliverable D3.2-3: Report on the most suitable lake macrophyte based assessment methods for impacts of eutrophication and water level fluctuations. March 2011.

Mischke U, Carvalho L, McDonald C, Skjelbred B, Lyche Solheim A, Phillips G, de Hoyos C, Borics G, Moe J & Pahissa J. 2011. WISER Deliverable D3.1-2: Report on phytoplankton bloom metrics, March 2011

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Phillips G, Skjelbred B, Morabito G, Carvalho L, Lyche Solheim A, Andersen T, Mischke U, de Hoyos C & Borics G. 2010. WISER Deliverable D3.1-1: Report on phytoplankton composition metrics, including a common metric approach for use in intercalibration by all GIGs, Aug 2010.

Jeppesen, E., Meerhoff, M., Holmgren, K., Gonzalez-Bergonzoni, I., Teixeira-de Mello, F., Declerck, S.A.J., De Meester, L., Søndergaard, M., Lauridsen, T.L., Bjerring, R., Conde-Porcuna, J.M., Mazzeo, N., Iglesias, C., Reizenstein, M., Malmquist, H.J., Liu, Z., Balayla, D. and Lazzaro, X., 2010. Impacts of climate warming on lake fish community structure and potential effects on ecosystem function. Hydrobiologia (2010) 646: 73–90.

3 Assessment and monitoring of transitional and coastal waters in Europe

Marine phytoplankton revealed a high level of spatial and annual variability across Europe

Key message

In order to plan monitoring programs of the phytoplankton community for classification of the ecological status of water bodies, it is important to know the amount of variability subject to the variation between stations, samples, sub-samples and the processing of samples. Without this information it is not possible to develop a smart sampling design and assign the available resources appropriately.

Evidence

The pigment-based community structure of samples collected in the WISER project were mainly related to salinity and temperature and reference conditions could not be established due to the lack of reference sites within the specific salinity regimes. Although pigment-based results are not directly comparable to results obtained by the traditional microscopic method, they are cost-efficient and much less time consuming than traditional analysis in the microscope.

However, it was not possible to establish a pressure-impact relationship between the eutrophication status (total nitrogen used as a proxy) and the distribution patterns of phytoplankton pigment samples and communities. Phytoplankton biomass (estimated from chl a concentrations) was significantly correlated with total nitrogen across the different sampling locations, but the major correlation was found with salinity and temperature. Thereby, the annual variability was found to be as high as the between-station (spatial) variability.

Results from a large-scale study quantifying the sources of variation in the assessment of phytoplankton communities across European water bodies showed that the 10–68% of variation was attributable to the variation between stations. For measurements of population density recorded as number of cells l^{-1} the main proportion of the variation (35%) was explained by the variation between the taxonomists counting the samples.

Implication

The large natural variability and the major influence from salinity and temperature on the distribution pattern of phytoplankton render pigment-based phytoplankton assessment highly uncertain. Different and commonly unknown accumulation and preservation rates of the different pigments in sediments reduce the possibility of describing quantitative reference phytoplankton communities from the fossil record. Thus, reference conditions for pigment composition could not be established.

The study however, also revealed that the precision of estimates of pigment concentrations for a specific water body can be enhanced by increasing the sample efforts (number of stations). Furthermore, continuous training and intercalibration of the staff involved in counting is the single most important measure to enhance the precision of estimates of phytoplankton density.

Further reading

For more detailed results, please consult Deliverable D4.1-1 (Identification of type-specific phytoplankton assemblages for three ecoregions), D4.1-2 (Report on assessment of pigment data potential for multi-species and assemblage indices) and D4.1-3 (Sources of Uncertainty in Assessment of Phytoplankton Communities) at http://www.wiser.eu/results/deliverables/.

A new phytoplankton size spectra index (SSI) has been developed for the assessment of transitional waters in Europe

Key message

A multi-metric index of the size spectra sensitivity of phytoplankton (ISS-phyto), which integrates the size structure metrics with metrics describing the sensitivity to anthropogenic disturbance, chlorophyll *a* and species richness was developed. The index was found to produce significantly higher values at undisturbed than disturbed sites and thereby being a promising indicator to assess the status of phytoplankton communities.

Evidence

Relatively few indices have been proposed for the assessment of the community structure changes of coastal and transitional water phytoplankton. Morphological-functional traits of phytoplankton with different cell size and size spectra show a specific response to different types of anthropogenic pressures. Nevertheless, very few attempts have been made so far to utilise functional traits such as body size, at the individual level, or size spectra, at the guild or community level, to develop multi-metric assessment tools compliant with the WFD.

We have developed, tested and validated a multi-metric index of size spectra sensitivity of phytoplankton (ISS-phyto), which integrates size structure metrics with metrics describing the sensitivity of size classes to anthropogenic disturbance, chlorophyll *a* and species richness measures. The ISS-phyto was developed using phytoplankton data of 14 Mediterranean and Black Sea transitional water bodies (i.e. coastal lagoons), which were classified as either “disturbed” or “undisturbed” ecosystems based on expert quantitative analysis, evaluation of anthropogenic pressures in the catchment area and their current protection and conservation status. The index was found to discriminate between natural and anthropogenic pressures presenting significantly higher values at undisturbed than disturbed sites; it was also tested successfully to a different set of lagoon and coastal areas in the WISER field studies.

Implication

The new metric ISS-phyto is a promising tool for assessment of the response of the phytoplankton community on eutrophication pressure in transitional and coastal waters and is recommended for further testing as a WFD monitoring tool in coastal lagoons.

Further reading

More detailed analysis and results are being presented in Deliverables D4.1-x and are available for download at http://www.wiser.eu/results/deliverables/.

Lugoli F., Garmendia M. , Lehtinen S., Kauppila, P., Moncheva S., Revilla M., Roselli L., Slabakova N.,Valencia V. , Basset A., 2012. Application of a new multi-metric phytoplankton index to the assessment of ecological status in marine and transitional waters. Ecological Indicators (submitted)

Vadrucci, M.R., Stanca, E., Mazziotti, C., Fonda Umani, S., Reizopoulou, S., Moncheva, S., Basset A., 2012. Ability of phytoplankton trait sensitivity to highlight anthropogenic pressures in Mediterranean lagoons: a size spectra sensitivity index (ISS-phyto) (under preparation)

Marine macroalgae are useful ecological quality indicators under the WFD monitoring programs

Key message

Macroalgae constitute a key biological quality element both for transitional waters (TW) and coastal waters (CW). Four macroalgal assessment methods have been developed in the WISER project: BMI (Blooming Macroalgae Index) MarMAT (Marine Macroalgae Assessment Tool), RICQI (Rocky Intertidal Community Quality Index) and the RSLA (Reduces Species List with Abundance). The first one is specific for transitional waters, whereas the three others are for marine water bodies. All these methods are easy to apply in intertidal areas (soft-bottoms in transitional waters and rocky shore in coastal ones).

Evidence

These methods were developed based on macroalgae features that are sensitive on degradation of marine environments resulting in decline of some sensitive species or in abnormal development of opportunistic taxa which are more tolerant to lower environmental quality. These assessment tools combine simple metrics from macroalgae, such as the species richness, the proportion of chlorophyta, the proportion of opportunists or the cover of some taxa.

The structural and functional characteristics of macroalgal taxa/communities are sensitive to changes in environmental conditions, justifying the inclusion of metrics based on measurable attributes from marine macroalgae. The marine macroalgae thriving on hard substrata, as sessile organisms, are good indicators for environmental degradation along coastal and transitional waters in Europe.

Implication

There are already traditionally macroalgae monitoring programmes to assess the quality of coastal and transitional waters in Europe. The applications of BMI, MarMAT, RSLA and RICQI macroalgae assessment tools are easy to use and already intercalibrated, thus those are recommended to be tested as potential indicators to be used on different monitoring programs along European coasts.

Further reading

These studies are included in WISER deliverables D4.2-1 at http://www.wiser.eu/results/deliverables/ and are published in Ecological Indicators.

João M. Neto, Rui Gaspar, Leonel Pereira, João C. Marques. Marine Macroalgae Assessment Tool (MarMAT) for intertidal rocky shores. Quality assessment under the scope of European Water Framework Directive. Ecological Indicators, in press. DOI: 10.1016/j.ecolind.2011.09.006

Isabel Díez; María Bustamante, Alberto Santolaria, Javier Tajadura, Nahiara Muguerza, Ángel Borja, José Ignacio Saiz-Salinas, José María Gorostiaga, Iñigo Muxika. 2012. Development of a tool for assessing the ecological quality status of intertidal coastal rocky assemblages, within Atlantic Iberian coasts. Ecological Indicators, 12: 58-71. doi:10.1016/j.ecolind.2011.05.014.

Benthic invertebrates respond consistently to human pressure gradients in coastal waters, but not in transitional waters

Key message

Several different indices have been proposed and may be used to classify the status of benthic invertebrates in transitional and coastal waters, and in lagoons. However, the response of such methods to human pressure gradients is critical in accepting them as suitable tools in assessing the ecological status within the WFD. Until now, very few studies investigated such response of methods already accepted within the WFD.

Evidence

We investigated 13 single metrics (abundance, species richness, Shannon's diversity, AMBI, five ecological groups, Margalef index, SN, ES100, and ES50) and eight multimetric methods (ISS, BAT, NQI, M-AMBI, BQI, BEQI, BITS, and IQI) to assess coastal and transitional benthic status along human pressure gradients in 5 distinct environments across Europe: Varna bay (Bulgaria), Lesina lagoon (Italy), Mondego estuary (Portugal), Basque coast

(Spain) and Oslofjord (Norway). Within each system, sampling sites were ordered in an increasing pressure gradient according to a preliminary classification based on professional judgement, and the response of single metrics and assessment methods to different human pressure levels was evaluated. The different indices are largely consistent in their response to pressure gradient, except in some particular cases (i.e. BITS, or ISS, in some cases). Inconsistencies between indicator responses were mostly in transitional waters (i.e. IQI, BEQI), highlighting the difficulties of the generic application of indicators to all marine, estuarine and lagoon environments. However, some of the single (i.e. ecological groups approach, diversity, richness, SN) and multimetric methods (i.e. BAT, M-AMBI, NQI) were able to detect such gradients both in transitional and coastal environments.

Implication

The agreement observed between different methodologies and their ability to detect quality trends across distinct environments constitutes a promising result for the implementation of the WFD's monitoring plans.

Further reading

This study has been published in Deliverable 4.3.1 and in Marine Pollution Bulletin.

Borja, A., E. Barbone, A. Basset, G. Borgersen, M. Brkljacic, M. Elliott, J. M. Garmendia, J. C. Marques, K. Mazik, I. Muxika, J. M. Neto, K. Norling, J. G. Rodríguez, I. Rosati, B. Rygg, H. Teixeira, A. Trayanova, 2011. Response of single benthic metrics and multi-metric methods to anthropogenic pressure gradients, in five distinct European coastal and transitional ecosystems. *Marine Pollution Bulletin*, 62: 499-513.

Zoobenthos species traits are useful and reliable for the assessment of transitional water ecosystems

Key message

Structural taxonomically-based components of the benthic macroinvertebrates communities have been used to assess ecological status (*sensu* WFD) of lagoon ecosystems. However, as lagoons are naturally euthrophic and selective ecosystems, individual species traits can have a major influence on the species' distribution and their response to disturbance; functional traits, as body size and size spectra actually respond to different types of anthropogenic pressures. Nevertheless, few studies have utilised functional traits such as body size, at the individual level, or size spectra, at the guild or community level, to develop multimetric assessment tools compliant with the WFD.

Evidence

We have developed, tested and validated a multi-metric Index of Size Spectra sensitivity (ISS), which integrates size structure metrics with metrics describing the sensitivity of size classes to anthropogenic disturbance and species richness measures. The ISS was developed using benthic macroinvertebrates data of 12 Mediterranean and Black Sea transitional water bodies (i.e. coastal lagoons), which were classified as either "disturbed" or "undisturbed" ecosystems based on expert quantitative analysis, evaluation of anthropogenic pressures in the catchment area and their current protection and conservation status. Data from a thirteenth Mediterranean lagoon, characterised by a very strong abiotic stress gradient, were used for validation purposes. The index is effective to discriminate between natural and anthropogenic pressures presenting significantly higher values at undisturbed than disturbed sites (Figure 3); it showed well defined and highly significant dose-response relationships along different stress gradients, such as the salinity gradient in Margherita di Savoia salt pan (Figure 4).

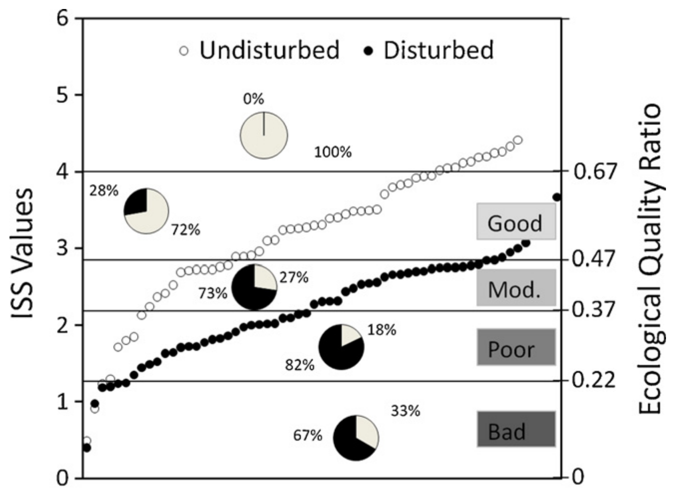


Figure 3: Distribution of studied lagoon sites across ecological quality levels (sensu Water Framework Directive). For each level the relative percentages of undisturbed and disturbed lagoon sites are reported.

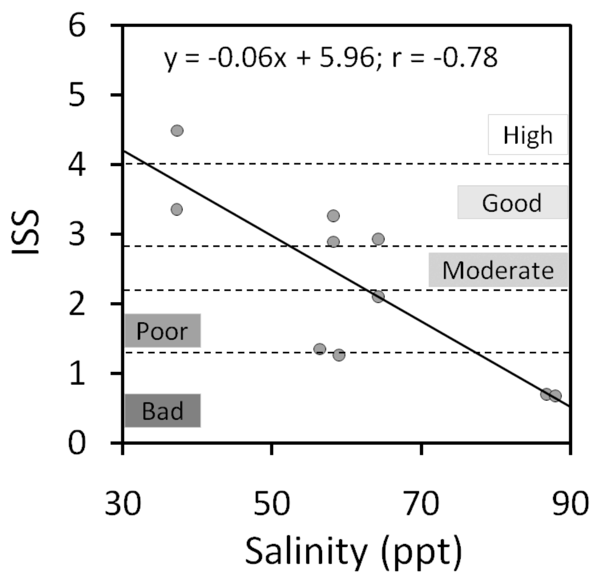


Figure 4: Validation of Iindex of Size Spectra sensitivity (ISS) in an independent ecosystem, Margherita di Savoia saltpan. Relationship between ISS values and salinity is reported ($p < 0.001$).

Implication

The new metric proposed for transitional waters is a precise and sensitive tool for discriminating various levels of ecosystem disturbance and easy to apply. The ISS has more practical advantages than disadvantages (Table 3), which favour its widespread use as a monitoring tool in coastal lagoons.

Further reading

This study has been published in Ecological Indicators.

Basset, A., Barbone, E., Borja, A., Brucet, S., Pinna, M., Quintana, X.D., Reizopoulou, S., Rosati, I. Simboura, N., 2012. A benthic macroinvertebrate size spectra index for implementing the Water Framework Directive in coastal lagoons in Mediterranean and Black Sea. *Ecological Indicators,* 12: 72-83.

Table 3: List of advantages and disadvantages of the Index of Size Spectra Sensitivity (ISS).

Advantages	**Disadvantages**
A1. Strong theoretical background on body size responses to environmental stress	**D1.** Damages to individual body size during sampling and/or handling
A2. Strong theoretical background on size spectra	**D2.** Some taxa are particularly sensitive to sampling, fixation and handling
A3. Body size is easy to measure	**D3.** Sampling probability of large sizes is affected by the sampling effort
A4. Body size measurements do not require high level of expertise	**D4**. Size spectra are sensitive to size-selective predation pressures
A5. Inter-calibration of body size measures among laboratories is simple	**D5.** Taxonomic expertise is anyway required (but see also point A7.)
A6. Consistent pressure-impact relationships are available for the most common pressures	**D6.** Assessment of individual body size is time and cost-expensive
A7. Size spectra detect early signals of anthropogenic disturbances before responses are detectable at the taxonomic level -	
A8. High discrimination power of anthropogenic pressures, even without accounting for taxonomic richness	
A9. High robustness to natural variability, embodied in the size spectra	

Fish indicators respond consistently to human pressure gradients across transitional waters

Key message

Using a matching combination of fish index, reference values and local dataset, the transitional fish index (and metrics) can be sensitive to pressure gradients. There is a proven negative response of fish quality features to pressure gradients, which make them suitable for biological quality assessments of transitional waters.

Evidence

A conceptual analysis, carried out on the strength of expected metrics responses to a set of human pressures, suggested chemical pollution and loss of habitat as the type of pressures more frequently and more strongly related to fish metrics. These pressures are often regarded as important indirect causes of alterations in transitional water fish assemblages. This preliminary analysis provided the conceptual basis for the ranking of human pressures in order of expected relevance to fish in transitional waters. In order to confirm further the relationship between fish-quality attributes and pressures, two WFD-compliant indices (the AFI and the EFAI in use for assessment in the Basque country (Spain) and Portuguese estuaries, respectively) were related to a set of pressures acting in these water bodies, while also considering their hydro-morphological descriptors. Stepwise linear multiple regression analysis indentified the following best model relating AFI index scores (as the dependent variable) to explanatory (independent) variables:

AFI = 0.013 +0.017(average estuary depth) –0.003(global pressure index) –0.001(residence time) +0.028(dredged volume) –0.007(% of channelling in ports) + 0.009(% of channelling out of ports)

Adjusted $R^2 = 0.859$, $p < 0.05$

The model identified a mixture of relevant pressure and hydromorphological covariates and indicates that, in this case, the deeper the estuary, and the shorter the residence time, the pressure index and the channelled ports within the estuary, then the higher the AFI values would be, indicating higher ecological quality. AFI clearly decreases with the increase of pressure proxies and morphological pressures. Similar analysis for the EFAI found comparable negative response of the index scores with increasing values of pressure proxies (see Figure 4). In this case, the EFAI responded to the overall anthropogenic pressure level.

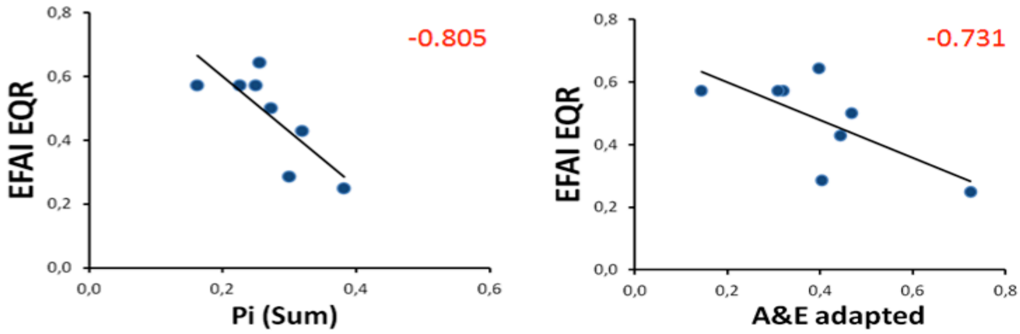


Figure 4: Response of EFAI against the overall pressure (measured as Pi (Sum) – sum of pressures, and as A&E adapted – pressure index adapted from Aubry & Elliott (2006) (see Pérez-Domínguez et al. 2012 -Appendix 4C for further details). The regression coefficient is given in the figure.

Furthermore, the good results of the intercalibration exercise suggests that each fish tool included in that analysis is in fact reacting in a common manner to a same level of human pressures, and providing a good agreement between methods in the diagnosis of ecological status. This is the ultimate goal of using fish in ecological assessments and suggests that all inter-calibrated indices are relevant and valuable indicators of human pressures in their own right. That is, there are providing an indication of ecological status independently of the pressure proxies used in the development and calibration steps.

In addition to the regression approach, an alternative method to establish metric-pressure relationship using a Bayesian approach was test-trialled in Drouineau et al. (2012). The Bayesian method allows the ability both to select relevant fish metrics and to combine them taking into account their sensitivity to pressure, their variability or any other relevant feature. For example, the method can also be a way to integrate data from expert opinion and it finally gives an assessment of the uncertainty of the diagnostic tool. It was tested on a dataset composed of a sample of 14 French lagoons. The analysis suggests that the quality diagnostics are less variable at the level of the multi-metric indicator than at the level of the fish metrics considered individually.

Implications

The BQE fish response to pressure fields in transitional waters provides a high level of ecological integration to the quality evaluation of transitional water systems. The Fish BQE is a sensitive indicator of ecological status and will be valuable to identify those specific pressures affecting fish assemblages providing targets for minimising the effects of stress in mitigation and restoration plans. Whole indices provide more consistent overall ES assessments but fish metrics considered individually may be more useful as a means to focus restoration measures.

Further reading

More detailed analysis and results are being presented in Deliverables D4.4-x and are available for download at http://www.wiser.eu/results/deliverables/.

Aubry A, Elliott M (2006) The use of environmental integrative indicators to assess seabed disturbance in estuaries and coasts: Application to the Humber Estuary, UK. Mar Pollut Bull 53:175-185

Drouineau H, Lobry J, Delpech C, Bouchoucha M, Mahevas S, Courrat A, Pasquaud S, Lepage M (2012) A Bayesian framework to objectively combine metrics when developing stressor specific multimetric indicator. Ecological Indicators 13:314-321

Borja A, Uriarte A, Muxika I, M. GJ, Uyarra MC, Courrat A, Lepage M, Elliott M, Pérez-Domínguez R, Alvarez MC, Franco A, Cabral H, Pasquaud S, Fonseca V, Neto JM (2012) Report detailing Multimetric fish-based indices sensitivity to anthropogenic and natural pressures, and to metrics' variation range. In: WISER Deliverable D4.4-3

Pérez-Domínguez R, Alvarez MC, Borja A, Cabral H, Courrat A, Elliott M, Fonseca V, Franco A, Gamito R, Garmendia JM, Lepage M, Muxika I, Neto JM, Pasquaud S, Raykov V, Uriarte A (2012) Precision and behaviour of fish-based ecological quality metrics in relation to natural and anthropogenic pressure gradients in European estuaries and lagoons. In: WISER Deliverable D4.4-5

4 Management of rivers in Europe

Riverine assemblages respond differently to individual stressors and stress levels

Key message

Fish, benthic invertebrates, macrophytes and benthic diatoms are differently affected by environmental degradation. While hydraulic alterations, for instance, may impose a strong negative impact on fish assemblages, this may be less strong for macrophytes: the **intensity** of responses varies across riverine assemblages and environmental stressors. In selected cases response to stress may even be positive; hence the **sign** of response varies too. Eventually, the stress levels of organism groups at which a response can be detected vary notably and reveal a dissimilar **sensitivity** of biological assemblages to stress.

Evidence

There is empirical evidence that river biota are almost always sensitive to general degradation (mixture of non-distinguishable stressors), land cover and water quality degradation, as opposed to hydrological and morphological degradation which affects could be less reproduced (Table 4). The response of fish to agricultural land use in the catchment, for instance, depends on the spatial scales considered for the calculation of percent land cover.

Diatoms and macroinvertebrates respond most strongly to general degradation already at low stress levels. These organism groups are weak indicators of local habitat improvement in degraded catchments, i.e. both groups are unlikely react to restoration unless broad-scale impacts are being remedied. Besides general and water quality degradation, fish and macroinvertebrates respond most intensively to morphological degradation, structural modification and catchment land use. Fish respond strongly to hydrological degradation, too. Hence, river fauna reveals a more intense, but not necessarily more sensitive, responses to stress, compared to the flora. Overall, aquatic macrophytes were found to be comparatively weak indicators of the stressors considered.

Table 4: Intensity and sensitivity of BQE's (riverine assemblages) response to different stressor groups.

BQE		general degradation	physico-chemical	hydrological	morphological	land use
Diatoms	Intensity	high	medium	low	low	medium
	Sensitivity	high	high	low	medium	high
Macrophytes	Intensity	low	medium	medium	low	low
	Sensitivity	medium	high	low	low	low
Benthic Invertebrates	Intensity	high	medium	low	medium	medium
	Sensitivity	high	medium	low	low	medium
Fish	Intensity	high	high	medium	high	high
	Sensitivity	medium	medium	medium	medium	low

Implication

Assessment and monitoring systems must account for the different capabilities of river biota in the detection and indication of single and multiple stressors. If multiple stressors act in a catchment, the use of a single assemblage only is likely to be insufficient and may lead to the wrong conclusions regarding the appropriateness of management or restoration measures.

River Basin Management must address and reduce *all* stressors relevant for ecological status. In agricultural or otherwise widely degraded watersheds, the impact of fertilizer and pesticide application, soil degradation and runoff modification is often omnipresent and can easily superimpose other, rather local impacts of structural and habitat degradation. Consequently, *any* local restoration in agricultural catchments must account for such large-scale impacts upstream of a restored site to initiate biotic recovery.

Further reading

In depth analysis of empirical data is available through WISER Deliverable 5.1-2. The conceptual linkages of environmental variables and riverine biota is available through WISER Deliverable 5.1-1 and Feld et al. (2011). The conceptual models of linkages can be accessed and used interactively at http://www.wiser.eu/programme-and-results/management-and-restoration/conceptual-models/.

Allan, J.D. (2004). Landscapes and riverscapes: The Influence of Land Use on Stream Ecosystems. Annu. Rev. Ecol. Evol. Syst. 35, 257–284.

Feld, C.K., Birk, S., Bradley, D.C., Hering, D., Kail, J., Marzin, A., Melcher, A., Nemitz, D., Petersen, M.L., Pletterbauer, F., Pont, D., Verdonschot, P.F.M. & Friberg, N. (2011) From natural to degraded rivers and back again: a test of restoration ecology theory and practice. Adv. Ecol. Res. 44, 119–209. (http://www.sciencedirect.com/science/article/pii/B9780123747945000031)

Hering, D., Johnson, R.K., Kramm, S., Schmutz, S., Szoszkiewicz, K. & Verdonschot, P.F.M. (2006). Assessment of European rivers with diatoms, macrophytes, invertebrates and fish: A comparative metric-based analysis of organism response to stress. Freshwat. Biol., 51, 1757–1785.

Paul, M.J., and Meyer, J.L. (2001). Streams in the urban landscape. Annu. Rev. Ecol. Syst. 32, 333–365.

Catchment and riparian land use control local habitat conditions

Key message

The hierarchical order of landscapes and riverscape implies a hierarchical order of stressors. Stressors, such as land use or river regulation, are ubiquitous in large parts of the world because of the multifaceted land and water uses. Flood protection is usually linked to severe modifications of hydrological and morphological characteristics. Agriculture increasingly dominates entire regions due to society's growing demand for food, resources and energy.

Broad-scale stressors impose serious problems for restoration and recovery. Not only do, agriculture and urban settlement control habitat conditions at finer scales, but land use impacts have often been present for decades or even centuries in many regions, e.g. in Central and Western Europe. Thus, the legacy of land use past may continue to impact entire river basins or sub-basins as long as such impacts are not being mitigated by appropriate (broad-scale) management schemes.

Evidence

Urban settlement and agriculture in the catchment upstream of a site largely influence and control the physical habitat conditions at the respective site. Urban settlements can influence water retention and storage through the percent of impervious area in the catchment, which in turn affects the hydrograph and can lead to severe flash floods following stormwater release. Less than 10% urban settlements in the catchment are frequently reported to significantly reduce biological and ecological quality (Paul and Meyer 2011).

The major impact pathways of intensive agriculture are nutrient enrichment (eutrophication) and excessive fine sediment entries (habitat loss). While nutrient enrichment can directly affect algal and plant communities, the loss of coarse substrates affects fishes and invertebrates.

Naturally vegetated riparian buffer strips not only can buffer impacts from agriculture, but also provide habitat (woody debris, leaves), shelter (root wads, shade), food (wood, leaves, terrestrial insects) and energy (carbon and nitrogen) to the riverine assemblages (Allan 2004, Feld et al. 2011).

Aquatic assemblages (e.g. fish and macroinvertebrates) significantly change their structural and functional composition, when the percent area as agriculture upstream exceeds 20% in mountain ecoregions (Figure 5). Lowland assemblages seem to respond less sharp to agriculture and significantly change values at 30–50%. These findings are in line with the thresholds reported by previous studies (e.g. Allan 2004).

Near-stream buffer areas along several kilometres upstream can help maintain biological diversity and functionality at a site, if a minimum of 40–50% within the buffer area is covered by forest. Ecological recovery may be promoted already by a minimum of 25% forested buffers upstream. Yet it is important to note that the increase of forest cover alone is unlikely to mitigate the impacts of land use.

Implication

Intensive agriculture and other land uses characterise large parts of Europe and constitute potential broad-scale stressors for riverscapes and its ecology. This in particular applies to the agricultural lowlands of Eastern, Central and Western Europe. Without appropriate mitigation and management, the negative impacts of land uses are likely to continue to impact rivers and hence hinder recovery, irrespective of hydrological and morphological improvements.

Consequently, restoration and river basin management *must* adequately address land use impacts. That is, restoration measures are required that i) are capable of mitigating land use impacts and that ii) address the appropriate scale of impact. Riparian buffers can be considered best practice. For instance, mixed riparian buffer strips (trees, shrubs, grass) have been proven to effectively retain nutrients and fine sediments from adjacent crop fields (see Feld et al. 2011 for a review). Buffer strips require several kilometres of length rather than tens or hundreds of metres.

Eventually, given the omnipresent character of agriculture, a re-organisation of land uses is needed and as a part of future river basin management. Conversion to less intensive land use forms in riparian areas will be most effective. This would require the reorganisation of agricultural policies in parallel.

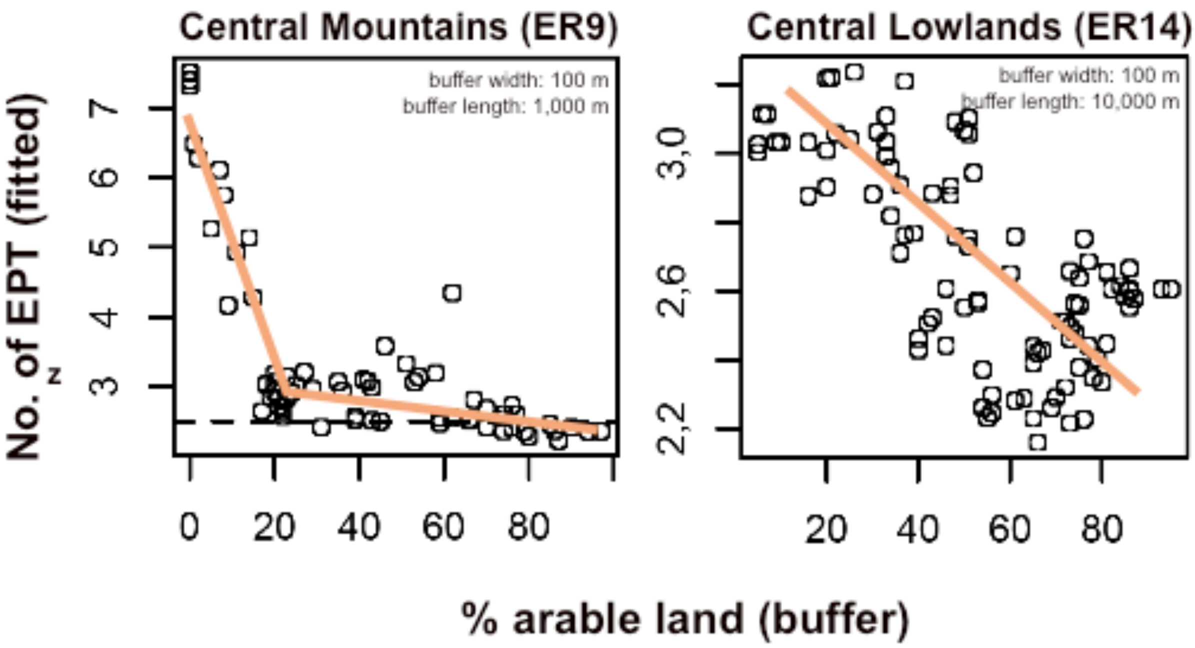


Figure 5: Boosted regression models identified the number of Ephemeroptera-Plecoptera-Trichoptera taxa (No. of EPT) to significantly decrease with increasing arable land in the riparian buffer of mountain rivers. A sharp decrease was obvious between 0 and 20% arable land. This decreasing trend is obvious too, although with less sharp the change, for lowland rivers. Note that the fitted values for EPT richness in lowland rivers mark a short gradient of one taxon difference only. The analysis was based on ca. 200 German macroinvertebrate samples in ecoregion (ER) 9 and 14. More in-depth results including fish and macrophytes are provided with WISER Deliverable D5.1-2.

Further reading

Detailed results can be derived from the sections by Feld and Lorenz in WISER's Deliverable 5.1-2.

Allan, J.D. (2004). Landscapes and riverscapes: The Influence of Land Use on Stream Ecosystems. Annu. Rev. Ecol. Evol. Syst. 35, 257–284.

Feld, C.K., Birk, S., Bradley, D.C., Hering, D., Kail, J., Marzin, A., Melcher, A., Nemitz, D., Petersen, M.L., Pletterbauer, F., Pont, D., Verdonschot, P.F.M. & Friberg, N. (2011) From natural to degraded rivers and back again: a test of restoration ecology theory and practice. Adv. Ecol. Res. 44, 119–209.

Paul, M.J., and Meyer, J.L. (2001). Streams in the urban landscape. Annu. Rev. Ecol. Syst. 32, 333–365.

Restoration is more likely to be successful, if upstream physical habitat degradation and land use impacts are low

Key message

Two previous statements address the predominant role of broad-scale stressors that may act at the scale of entire (sub-) catchments and consequently may impact any site within the catchment.

Consequently, river restoration is more likely to initiate and maintain biological recovery, if such broad-scale impacts are either completely missing or being mitigated in parallel to restoration at the fine (local) scale.

Evidence

There is empirical evidence from restoration monitoring that restoration measures can initiate biological recovery, if the physical habitat conditions several kilometres upstream of the restoration are only moderately modified or in better condition. In particular the fish and macrophyte assemblages were found to be strongly influenced by habitat quality up to 10 km upstream (Table 5). Macroinvertebrate ecological quality was related to shorter stretches upstream (up to 2.5 km). Empirical analyses imply that about 1 km length upstream in a moderate or better physical habitat quality might suffice to promote biological recovery (see Lorenz in WISER Deliverable D5.1-2).

Implication

Where broad-scale stressors impact ecological quality after restoration and may hinder recovery, such stressors require mitigation. Practitioners need to know the multiple stressors that may impact restoration candidate sites. They should prioritise those stretches that are least impacted by broad-scale stressors and thus may constitute stepping-stones within a broader restoration scheme. Local restoration measures need to be integrated into restoration schemes at the broad scale.

This broad-scale and integrated restoration is well referred to by the WFD and termed 'River Basin Management'. Yet, it seems as if this broad-scale approach deserves more attention by scientists and practitioners in order to use the limited resources available most effectively for river restoration and management.

Further reading

For a detailed analysis of the effects of upstream physical habitat quality and land use conditions on ecological quality assessment at restored and unrestored sites see Lorenz in WISER's Deliverable D5.1-2.

Table 5: Spearman rank correlation and significance levels of the relationship between ecological quality ratios (EQRs) of three BQEs at unrestored and restored sites and the physical habitat quality in several distances upstream of the sites (N = number of valid cases; significant correlations in bold). The correlations reveal a notable relationship of fish EQRs with physical habitat conditions up to 10 km upstream (maximum values at 2.5–5 km upstream) of the sampled river sites. Macrophytes showed a similar relationship up to 7.5 km upstream, while the relationship with invertebrate EQRs was significant up to 2.5 km upstream only.

	Fish		Invertebrates		Macrophytes	
Distance upstream	Unrestored	Restored	Unrestored	Restored	Unrestored	Restored
500 m	**-0.37** N=32 p=0.035	**-0.44** N=34 p=0.010	**-0.36** N=33 p=0.038	**-0.50** N=35 p=0.002	-0.27 N=34 p=0.128	**-0.49** N=35 p=0.003
1,000 m	**-0.35** N=32 p=0.048	**-0.41** N=34 p=0.002	**-0.38** N=33 p=0.027	**-0.42** N=35 p=0.013	-0.25 N=34 p=0.150	**-0.46** N=35 p=0.005
2,500 m	**-0.51** N=32 p=0.003	**-0.52** N=34 p=0.002	**-0.45** N=33 p=0.008	**-0.40** N=35 p=0.017	-0.32 N=34 p=0.068	**-0.54** N=35 p=0.001
5,000 m	**-0.47** N=32 0.007	**-0.51** N=34 p=0.002	-0.32 N=33 p=0.071	-0.31 N=35 p=0.066	**-0.37** N=34 p=0.034	**-0.45** N=35 p=0.006
7,500 m	**-0.47** N=32 p=0.007	**-0.42** N=34 p=0.014	-0.23 N=33 p=0.208	-0.24 N=35 p=0.165	**-0.36** N=34 p=0.036	**-0.38** N=35 0.023
10,000 m	**-0.50** N=32 0.004	**-0.35** N=34 p=0.043	-0.22 N=33 p=0.229	-0.25 N=35 p=0.147	-0.33 N=34 p=0.060	-0.29 N=35 p=0.089

River Basin Management Plans insufficiently account for research and monitoring demands

Key message

The assessment and monitoring of the ecological status of rivers and other surface waters is explicitly referred to in the WFD and hence constitutes a basis for all River Basin Management Plans (RBMPs). River Basin Managers and practitioners are informed about the stressors to be assessed, the BQEs to be used for monitoring and the frequency of monitoring events with regard to each individual BQE.

In contrast, the monitoring of restoration and management measures is neither specifically referred to in the WFD, nor is it sufficiently defined elsewhere. The general approach to date is to apply operational monitoring to assess restoration effects. Changes due to restoration often remain dubious as practitioners miss to sample and record the ecological status of a restoration candidate prior to the implementation of measures. Consequently, the knowledge about the specific requirements of restoration measures that determine restoration success or failure is humble due to the lack of appropriate restoration monitoring schemes

Evidence

The lack of appropriate monitoring schemes is obvious. A review of 160 restoration studies revealed two major shortcomings (Feld et al. 2011): First, restoration monitoring is often poorly designed and hence inappropriate to reliably assign any detected change (or non-change) to restoration. And second, the status before restoration is rarely being monitored, while the monitoring duration is limited to 3–4 years: Thus, long-term effects (>7–10 years) of restoration remain unknown for the majority of studies (Feld et al. 2011).

The lack of restoration monitoring is likely to continue within the first management period of the WFD (until 2015). This, in part, becomes evident from the selection of RMBS's analysed for WISER'S Deliverable D5.1-2 (see Verdonschot et al. therein). Although the selection represents only a small part of Europe, the considered RBMPs concordantly prove that little attention has been assigned to additional research and monitoring until 2015. Moreover, the RBMPs imply that practical restoration is primarily planned for the second and third monitoring period of WFD, which means that the existing knowledge gaps with regard to the reasons for success and failure of restoration remain presumably persist.

Implication

River Basin Management involves huge efforts for and investments in restoration and mitigation measures in the future, presumably for the next couple of decades. As these investments in the environment compete with other society's demands, it is necessary that any bit of these investments is being spent efficiently.

However, the ongoing lack of appropriate restoration monitoring schemes hinders the detection of effects. Consequently, practitioners do not know whether a specific measure is going to support ecological recovery

Sufficiently simple, but 'smart' monitoring designs might help scientists and practitioners fill the knowledge gaps (compare Feld et al. 2011 and WISER's Deliverable D5.1-2). First at least one sampling event prior to the implementation of measures is required to define the ecological status *before* restoration. Furthermore, an unrestored river stretch upstream to the restored section is required as *control* in order to be able to detect the degree of temporal variability within the river system. The full design is called *BACI* (before-after-control-impact) and can be considered the method of choice in restoration monitoring (Feld et al. 2011). Second, in addition to the WFD assessment and monitoring tools, more thorough records of hydrological, morphological and biological changes after restoration are required to better detect the multiple effects of individual restoration measures as well as their interactions. And third, restoration monitoring must help inform practitioners about both the short- and long-term changes after restoration.

Further reading

The comparison of selected RBMPs in Austria, France, Germany and the Netherlands is presented by Verdonschot et al. in WISER's Deliverable 5.1-2 at http://www.wiser.eu/results/deliverables/.

Feld, C.K., Birk, S., Bradley, D.C., Hering, D., Kail, J., Marzin, A., Melcher, A., Nemitz, D., Petersen, M.L., Pletterbauer, F., Pont, D., Verdonschot, P.F.M. & Friberg, N. (2011) From natural to degraded rivers and back again: a test of restoration ecology theory and practice. Adv. Ecol. Res. 44, 119–209.

Climate change alters fish assemblage structure and function distribution in Europe

Key message

Species distributions are driven by environmental conditions, be it natural landscape settings or environmental stress induced by human activities including Climate Change. The Intergovernmental Panel on Climate Change predicted changes in temperature and precipitation in Europe for the periods 2020–2030 and 2050–2060. These changes are expected to greatly alter the distribution of fish, by providing more suitable habitats for species tolerating or preferring warm water, and by restricting species adapted to cold water habitats; the latter are expected to decline or even go extinct some regions of Europe. As these changes may also affect fish assemblage metrics in use at present for assessment and monitoring purposes, this implies that the reference condition baselines use to assess the ecological status of rivers based on fish would not be adequate in the future.

Evidence

Empirical evidence of these changes was shown by the study conducted downstream of lake outlet flow in the Traun river. During the last three decades the water temperature increase by on average 2.2 °C in August. This increase led to unsuitable thermal conditions for the grayling (*Thymallus thymallus*), which was historically present in this area. Consequently the grayling population greatly decline in favour of more adapted species such as barbel (*Barbus barbus*, Figure 6).

Depending of the individual species considered, the accuracy of species distribution models (SDMs) may be very variable (Figure 7). In general, the models on species with a narrow and distinct temperature niche, i.e. both cold water and warm water-adapted species (e.g. bleak, *Alburnus alburnus*) are more accurate.

In lowland catchments (e.g. the Seine basin in France), the absence of possible thermal refugia in the upstream part of the catchment may amplify the risk of regional species extinctions (Figure 8).

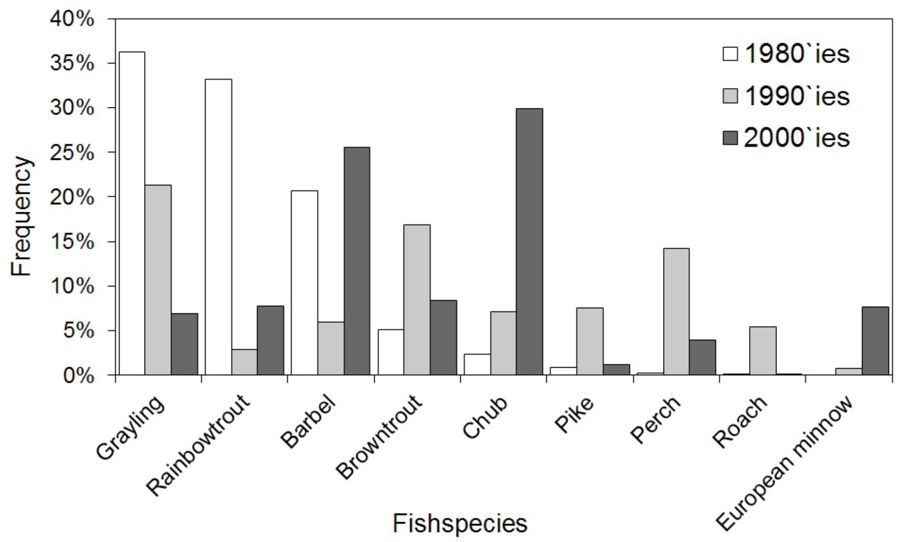


Figure 6: Shift of species composition from the 1980'ies until the 2000'ies in the River Traun in relation with an increase of water temperatures (on average +2.2°C).

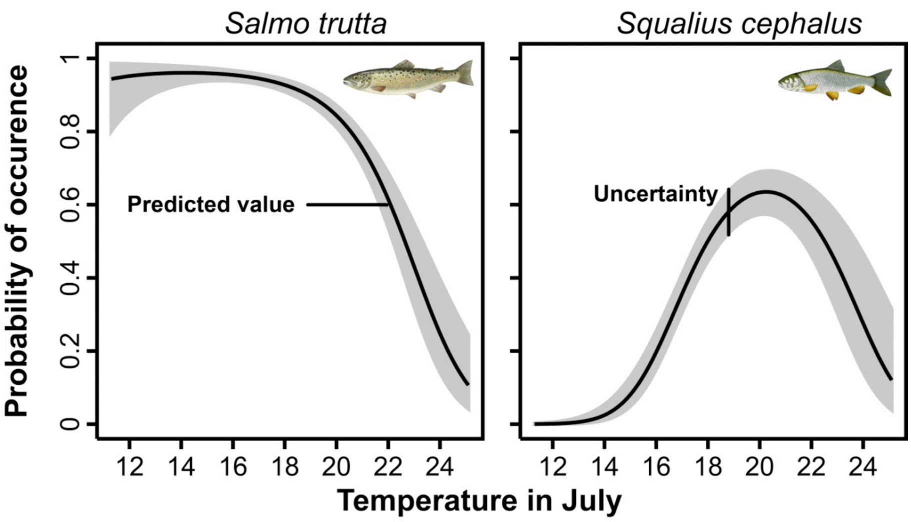


Figure 7: Marginal effect of mean air temperature in July on species probability of occurrence of two fish species, brown trout and chub, predicted with species distribution models (Logez et al. 2011). The black curve represents the predicted values and the prediction confidence bands are in grey. These representations could have been obtained by fixing the other environmental values (stream power, thermal amplitude between July and January, upstream drainage area) to their median.

Implication

Climate Change effects have to be taken into account in River Basin Management, for instance when using reference conditions as baselines for assessment or when designing restoration measures. If salmonid species, for example, go extinct in particular catchments, this requires consideration when setting the biological assessment reference in that catchment, or when defining the biological goals for restoration. Without consideration of Climate Change impacts, assessment runs the risk of misclassification. To evaluate such potential shifts, a monitoring network of reference sites in Europe may help inform the practitioners about potential consequences of global warming and its effects on both the biota and its abiotic environment.

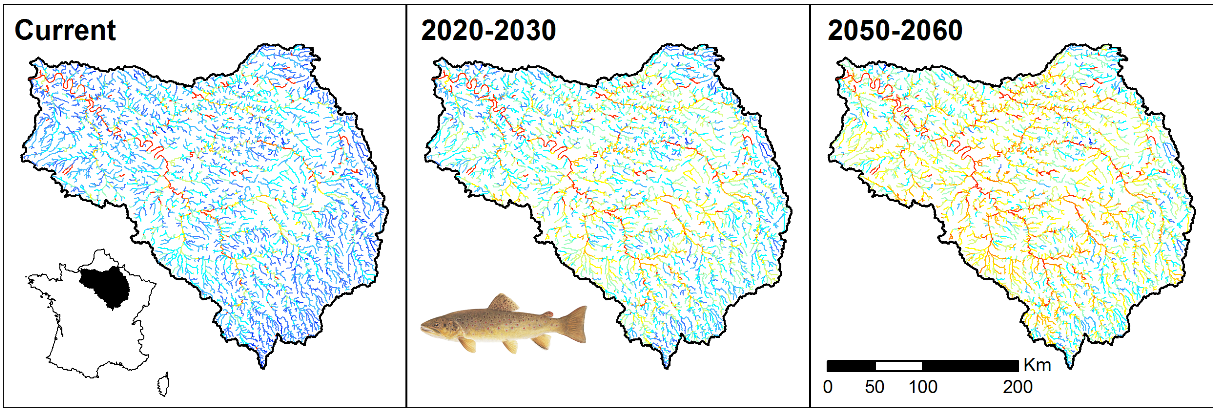


Figure 8. Probability of presence of the brown trout (Salmo trutta, L.) in the Seine river basin (France) derived from the species distribution models (Logez et al. 2011) for the (a) the current environment conditions, (b) projected climatic conditions for 2020-2030 and (c) for the projected climatic conditions for 2050-2060. Probabilities are computed for each stream reach of the CCM2 network (probabilities: — 0–0.1, — 0.1–0.2, — 0.2–0.3, — 0.3–0.4, — 0.4–0.5, — 0.5–0.6, — 0.6–0.7, — 0.7–0.8, — 0.8–0.9, — 0.9–1).

Further reading

The climate change effects on fish BQE (species and metrics) is presented by Logez et al. in WISER's deliverable 5.1-3 at http://www.wiser.eu/results/deliverables/.

Logez, M., Bady, P. and Pont, D. (2011), Modelling the habitat requirement of riverine fish species at the European scale: sensitivity to temperature and precipitation and associated uncertainty. Ecology of Freshwater Fish 21: 266–282.

5 Management of lakes in Europe

Climate warming causes profound changes in lake fish assemblages

Key message

Fish play a key role in the trophic dynamics of lakes. With climate warming, complex changes in fish assemblage structure may be expected owing to the direct effects of temperature and indirect effects of eutrophication, water level changes, stratification and salinisation. This means that warming will result in fish-mediated increase in eutrophication partly counteracting the effect of nutrient loading reduction. The response of fish to the warming in recent decades has been surprisingly strong, making fish ideal sentinels for detecting and documenting climate-induced modifications of freshwater ecosystems.

Evidence

An analysis of the effect on fish assemblages to climate change and climate variability has been conducted based on long-term (10 to 100 years) data series from 24 European lakes. These lakes constitute an appropriate and tractable sample of the world's lakes since many of them have been monitored more intensively and for a longer period of time than have most lakes elsewhere. Profound changes in fish assemblage composition, size and age structure were found during the last decades and a shift towards higher dominance of eurythermal species. The shift has occurred despite an overall reduction in nutrient loading that should have benefited the fish species typically inhabiting cold-water low-nutrient lakes and larger-sized individuals.

The cold-stenothermic Arctic charr has been particularly affected and its abundance has decreased in the majority of the lakes where its presence was recorded. The harvest of cool-stenothermal trout has decreased substantially in two southern lakes. Vendace, whitefish and smelt has shown a different response depending on lake depth and latitude, with a drastic reduction in the Estonian Lake Peipsi. Perch was apparently stimulated in the north, with stronger year classes in warm years, but its abundance has declined in southern Lake Maggiore. Where introduced, roach now seems to take advantage of the higher temperature after years of low populations. Eurythermal species such as bream, pike-perch and shad are on the increase. The climate effects have overall been larger in shallow lakes.

The fish assemblage is not only affected directly by warming and changes in the thermal stability of the lakes. Numerous recent studies and reviews indicate that warming will exacerbate existing eutrophication problems and this will in a self-amplifying manner further stimulate a shift to dominance of eurythermal species. They typically tolerate low oxygen levels and high ammonia concentrations and prevalence of small fish. A reduced ice cover period will enhance fish survival, with potential cascading effects within the food web, also reinforcing eutrophication. Therefore, we can expect an allied attack by eutrophication and warming in lakes in the future and the shifts in abundance, size and composition will be reinforced and stimulated by this process.

Diatoms and macro invertebrates respond most strongly to general degradation already at low stress levels. This renders both organism groups weak indicators of local habitat improvement in degraded catchments, i.e. both groups are unlikely react to restoration unless broad-scale impacts are being remedied. Besides general and water quality degradation, fish and macro invertebrates respond most intensively to morphological degradation, structural modification and catchment land use. Fish respond strongly to hydrological degradation, too. Hence, river fauna reveals a more intense, but not necessarily more sensitive, responses to stress, compared to the flora. Overall, aquatic macrophytes were found to be comparatively weak indicators of the stressors considered.

Implication

The most obvious alterations encompass a decline in cold-stenothermal species, in particular in shallow lakes, an increase in eurythermal species even in deep, stratified lakes. Several case studies show a decrease in the average size of the dominant species roach and perch.

This also means that warming will result in a fish-mediated increase in eutrophication partly counteracting the effect of nutrient loading reduction. It also implies that it will be more difficult to obtain the good ecological status required by the WFD in lakes facing temperature changes due to global warming. The way to (partly) counteract the effect of warming is to reduce the nutrient input to lakes even further than planned under the present-day climate. The response of fish to warming during recent decades has therefore been surprisingly strong, making fish ideal sentinels for detecting and documenting climate-induced modifications of freshwater ecosystems.

Futher reading

Detailed results can be derived from WISER Deliverable 5.2-2 "Report on using BQEs as indicators for reducing pressures" as well as the corresponding manuscript:

Jeppesen E., T. Mehner, I. J. Winfield, K. Kangur, J. Sarvala, D. Gerdeaux, M. Rask, H. J. Malmquist, K. Holmgren, P. Volta, S. Romo, R. Eckmann, A. Sandström, S. Blanco, A. Kangur, H. R. Stabo, M. Meerhoff, A.-M. Ventelä, M. Søndergaard, T. L. Lauridsen (submitted). Impacts of climate warming on lake fish assemblages: evidence from 24 European long-term data series.

Include zooplankton as a BQE in assessment of lake ecological status, please

Key message

Surprisingly to many lake ecologists, zooplankton was not included as a biological quality element (BQE) in lake assessment according to the WFD — despite they are being considered to be an important and integrated component of the pelagic food web. Using contemporary and sediment samples from Danish, Estonian and UK lakes, and time series following changes in pressures (eutrophication and top-down control) it was shown that contemporary zooplankton (and cladoceran remnants in the upper sediment layer) has a strong indicator value.

Moreover, zooplankton constitutes a cost-efficient indicator group capable of indicating the trophic state and ecological quality of lakes. In addition, zooplankton is important to measure the success or failure of management measures aiming at restoring lakes to good ecological status. Therefore, it is strongly recommended to include zooplankton, including cladoceran remnants in the surface sediment, as a central BQE in future WFD assessment and monitoring schemes.

Evidence

Using contemporary samples from numerous lakes in DK from mainly Denmark, Estonia and UK it is shown that zooplankton size structure, proportion of large zooplankton, cladoceran size and the zooolankton : phytoplankton biomass ratio are suitable indicators of ”top-down” processes in lakes. Important indicators of ”bottom-up” processes could be zooplankton biomass, the proportion of rotifers by numbers and the proportion of calanoid copepods of ”bottom-up” processes. Combination of “top-down” and “bottom-up” indicator metrics might yield a solid assessment of trophic conditions in the pelagic of lakes.

Time series for lakes in recovery from eutrophication as well as lakes restored by biomanipulation provide further evidence of the strength of zooplankton as strong indicators of changes in pressures. The paleoecological data presented suggest that sedimentary cladoceran assemblages are also sensitive to ecological change and are a relatively simple metric summarizing a combination of the benthic/pelagic balance of taxa, and size of remains as a measure of fish predation pressure could be a useful predictor of ecological quality. Further exercises are needed to develop metrics at the regional level throughout Europe as for other BQEs.

So far, the most promising metrics based on contemporary samples are zooplankton biomass, the proportion of rotifers by numbers, the proportion of large zooplankton, zooplankton size, cladoceran size, and the proportion of calanoid copepods; and for surface sediment: size and the proportion of large forms of resting eggs and the proportion of pelagic cladoceran remains.

Implication

The examples illustrate that zooplankton are important indicators of the structure and function of freshwater lake ecosystems and their ecological status. It is acknowledged, that zooplankton monitoring already today can be included in schemes of operational monitoring (for water bodies identified as being at risk of failing to meet their environmental objectives, and for those into which priority list substances are discharged), and investigative monitoring (if the reason for deviations is unknown, to ascertain the causes of a water body or water bodies failing to achieve the environmental objectives, or to ascertain the magnitude and impacts of accidental pollution).

However, based on the experience from Denmark it is clear that the risk is very high (likely close to 100%) that policy makers and managers tend to follow a "minimum requirement" policy. We, therefore, strongly appeal to the relevant EU authorities to consider (and include) zooplankton as a BQE during the first revision of the monitoring programmes. We also see the omission of zooplankton as a loss of opportunity for transitional waters and large rivers. The focus mainly on ecosystem structure and less on function in the WFD must be reconsidered, and we have shown that zooplankton are a key element here for understanding lake ecosystem function – and perhaps also for large rivers and transitional waters.

Further reading

Detailed results can be derived from WISER Deliverable 5.2-2 "Report on using BQEs as indicators for reducing pressures" as well as the corresponding manuscript:

Davidson T.A., G. H. Henderson, H. Bennion, E. Jeppesen, D. Morley, B. Odgaard, R. Rawcliffe, J. Salgado & C. Sayer, 2011. The role of Cladocerans in tracking long-term change ecosystem structure and function in shallow lakes – Hydrobiologia 676:299-315

Jeppesen E., P. Nõges, T. A. Davidson, J. Haberman, T.Nõges, K. Blank, T.L. Lauridsen, M. Søndergaard, C. Sayer , R. Laugaste, L.S. Johansson ,R. Bjerring & S.L. Amsinck, 2011. Zooplankton as indicators in lakes - a plea for including zooplankton in the ecological quality assessment of lakes according to the European Water Framework Directive (WFD)- Hydrobiologia 676:270-297.

A tool may help estimate the effects of nutrient load reduction under a variety of climate scenarios

Key message

Nutrient assimilation capacities of European lakes were estimated using a large data set. The effect of climate warming on eutrophication proved to be positive. Thus, in warmer climatic conditions, an effective reduction of nutrients is needed to achieve a good ecological condition. A model was developed and included in the LakeLoadResponse (LLR) internet tool which can be used by water managers to estimate the reduction of nutrient load required at present and under changing climate conditions.

Evidence

The linear mixed effects model is based on chlorophyll a data from 351 European lakes. The effect of total phosphorus, total nitrogen and water temperature on chlorophyll a concentrations varied among lake types, individual lakes within a type and individual samples within a lake. The amount of variation was significantly reduced using a linear mixed effects model for nested data. The statistical inference was based on a Bayesian approach thus giving a more realistic assessment of the effect of model uncertainty. The model is implemented in an internet tool and has been successfully used for the planning of restoration measures in Finland.

Implication

Using the LLR tool, it is possible to test how the changes in water temperature affect the nutrient reduction required to achieve good ecological status. The LLR delivers predictions on water quality status with statistical confidence intervals to give more insight for the management actions.

If combined with a map-based web service, the model can help water managers illustrate the forecasted effects in maps. For instance, the effect of fisheries management will be analyzed using extensive data from Finnish lakes in the GisBloom project (Life+ 2010–2013).

Further reading

A description of the mixed chlorophyll a model can be derived from WISER Deliverable 5.2-4: "Internet tool (model to assess target loads) for lake managers". Further instructions of the LLR internet tool and descriptions of the underlying models are available at http://lakestate.vyh.fi.

Lake sediments provide insight into the history of the conditions of individual lakes and, hence may assist the definition of reference conditions

Key message

Throughout Europe the majority of lakes have been modified to some extent by human activity with agriculture and sewerage being the major contributors to eutrophication, most notably since the mid-twentieth century. As a consequence, higher algal productivity has lead to filtration problems for the water industry, oxygen depletion, recreational impairment, loss of biodiversity and an overall decline in habitat quality.

Lake sediment analysis provides unique insights into the history of lake ecosystems, including evidence for the nature and timing of ecosystem change resulting from human impact. Palaeoecological methods can reveal pre-impact conditions as well as identifying any signs of recovery and have played a key role in the WFD in determining pre-enrichment reference conditions. Diatom records have proved especially valuable in this respect, largely due to their sensitivity to shifts in trophic status. In the absence of long-term chemical monitoring analysis of lake sediments can provide evidence not only of the pre-eutrophication baseline conditions, but also help track degradation and recovery pathways and thus provide a valuable tool for informing restoration programmes.

Furthermore, where restoration programmes are underway, there is evidence that the recovery pathways are not simply a reverse of the degradation process and therefore indicate that other factors such as climate change may be influencing the rate and direction of recovery.

Evidence

Diatoms are a group of single-celled, microscopic algae, which preserve well in lake sediments due to their siliceous cell walls. Many diatom species are also very sensitive to changes in water quality thus, as the cells die and are laid down in lake sediments they provide a record of the environment within which they lived. By relating the fossil species to modern diatom assemblages collected across wide environmental gradients, very good estimates of past lake water chemistry can be inferred and an environmental history tracked down through the sediment record. With the application of radiometric dating, the timing and rate of changes can be determined and pre-impact (reference) conditions established. In many European lakes, diatoms have provided clear evidence that the onset of eutrophication was associated with changes in agricultural practice and urban development, particularly since the mid-twentieth century.

Furthermore, where restoration programmes are underway it might be expected that the diatom record would show a reversal in the degradation pathway, but instead, diatom-based metrics often exhibit an alternative recovery pathway. This demonstrates that a reduction in one or more environmental stressors may not ultimately return a lake to reference conditions, but instead other processes such as internal nutrient loadings and climate change may determine the rate and direction of recovery.

Implication

Lake sediments provide a valuable means by which reference conditions may be established in lakes. Furthermore, environmentally sensitive organisms such as diatoms may be used to determine both the degradation and the recovery process. In terms of lake management, while it is important to be able to identify baselines it should also be recognized that recovery may not simply be the reverse process of the degradation pathway and that the reference state may perhaps never be achievable in some lakes. The evidence suggests that recovery is more predictable in deep stratified lakes than shallow lakes, where top-down processes exert a major environmental control, but that in all cases the recovery process has a long way to go before reaching pre-impact conditions.

This work highlights the important role that paleolimnological approaches can play in establishing a benchmark against which managers can evaluate the degree to which their restoration efforts are successful. Diatoms are just one of many biological groups preserved in sediments and by extending this work to use multiple assemblages it is possible to evaluate wider ecosystem responses to environmental stressors. These multi-proxy palaeoecological techniques therefore have an important role to play in assessing degradation and recovery pathways and informing lake management in order to satisfy the aims of the Water Framework Directive.

Further reading

WISER Deliverable 5.2-5: Bennion et al. in: "Report on effects of global change on reference conditions and ecological status of lakes", downloadable at http://www.wiser.eu/results/deliverables/.

Bennion, H. and Battarbee, R.W. (2007) The European Union Water Framework Directive: opportunities for palaeolimnology. Journal of Paleolimnology, 38, 285-295.

Bennion, H., Battarbee, R.W., Sayer, C.D., Simpson, G.L. and Davidson, T.A. (2011) Defining reference conditions and restoration targets for lake ecosystems using palaeolimnology: a synthesis. Journal of Paleolimnology, 45, 533-544.

Bennion, H., Simpson, G.L., Anderson, N.J., Dong, X., Hobaeck, A., Guilizzoni, P., Marchetto, A., Sayer, C.D., Thies, H. and Tolotti. M. (2011) Defining ecological and chemical reference conditions and restoration targets for nine European lakes. Journal of Paleolimnology, 45, 415-431.

6 Management of transitional and coastal waters in Europe

Benthic communities are more vulnerable to hypoxia under global warming

Key message

Hypoxia is a mounting problem affecting the world's coastal waters, with severe consequences for marine life, including death and catastrophic changes. The deleterious effects of hypoxia are amplified by warming. Global warming will contribute to decrease the global average dissolved oxygen in the oceans worldwide, and will also affect the oxygen requirements of marine benthic macrofauna. Increasing temperature diminishes oxygen solubility and increases the respiration rates of organisms, as temperature plays a fundamental role in regulating metabolic processes. Ocean warming increases the vulnerability of benthic macrofauna to reduced oxygen, increasing the mortality of benthic fauna and greatly extending the area of coastal ecosystems affected by hypoxia-driven mortality.

Evidence

A synthesis of experimental responses of marine organisms to reduced O_2 and increasing temperature shows that ocean warming increases the vulnerability of benthic macrofauna to reduced oxygen concentrations, increasing the mortality of benthic fauna and greatly extending the area of coastal ecosystems affected by hypoxia-driven mortality.

The meta-analysis (based on 576 published experiments) confirmed that survival times under hypoxia were reduced by on average 74% and that median lethal concentration increased by on average 16% when marine benthic organisms were exposed to warmer temperatures (Figure 1.1).

Implication

Warming will negatively impact the survival of benthic organisms under low oxygen conditions. By the end of this century survival times will be a 35.6% lower under hypoxia and the threshold oxygen concentrations for high mortality to occur will increase by, on average, 25.5% if bottom water temperature increases by 4 ºC.

Hypoxia is already expanding globally across coastal waters, parallel to increased flux of nutrients to the coastal zone and warming of coastal waters. The synergy between two global changes, oxygen depletion and warming of coastal waters, threatens coastal benthic macrofauna. Aggravation of the negative effects of spreading hypoxia by the effect of warming in rising the O_2 requirements of organisms and, therefore, the O_2 thresholds for hypoxia together with the fact that warming will accelerate oxygen depletion suggest that the threats to marine biota derived from hypoxia will be amplified in a context of global warming and may, thus, be greater than hitherto anticipated.

Further reading

Detailed results can be derived from WISER's deliverable 5.3-1 "Temperature effects on hypoxia and benthic fauna" (downloadable at http://www.wiser.eu/results/deliverables/) as well as the paper:

Vaquer-Sunyer, R. and Duarte, C.M. (2011). Temperature effects on oxygen thresholds for hypoxia in marine benthic organisms. Global Change Biology 17:1788-1797.

Hypoxia makes ecosystem recovery more difficult

Key message

Coastal hypoxia is increasing in the global coastal zone, where it is recognized as a major threat to biota. Hypoxia is defined as oxygen concentrations below a certain value, typically 2 ml/l or 2 mg/l, but the deleterious effects on the ecosystem already start at higher oxygen concentrations. Knowing the thresholds that fundamentally lead to a change in ecosystem functioning is important to quantify for management. Moreover, these thresholds are not static but regulated by other processes, associated with both local and global pressures on the system, particularly warming. Exceeding the critical thresholds associated with hypoxia may require even further nutrient reductions to restore a well-functioning benthic community. However, recovery from hypoxia is possible.

Evidence

The literature is populated with studies documenting decreasing oxygen concentrations associated with eutrophication, and how this affects the structure and functioning of the benthic community. Many coastal ecosystems in Europe and North America have now experienced decreasing inputs of nutrients, although the expected improvement of oxygen conditions and re-establishment of benthic fauna is only observed for a few systems, e.g. Delaware River and Stockholm Archipelago (Figure 9). In these systems the recovery took decades following drastic reductions in nutrient inputs. Many other coastal ecosystems show no signs of improvement, despite reduced levels of nutrients and chlorophyll.

Implication

Hypoxia is known to alter the biogeochemical processing of nutrients leading to feed-back mechanisms through reduced nitrification and releases of iron-bound phosphorus. Moreover, the loss of bioturbating macrofaunal organisms following hypoxia reduces the efficiency of nutrient removal processes. Therefore, hypoxia is a self-sustaining process and ecosystems should be managed to maintain oxygen levels about critical thresholds that imply a collapse of the benthic community. Coastal ecosystems can, however, recover from hypoxia, but so far this has only been observed for systems with large reductions in nutrient inputs and even so, still taking decades to recover. However, re-establishment of sound benthic communities can significantly enhance the recovery process. These experiences suggest that hypoxia introduces a hysteresis response to the nutrient pressure.

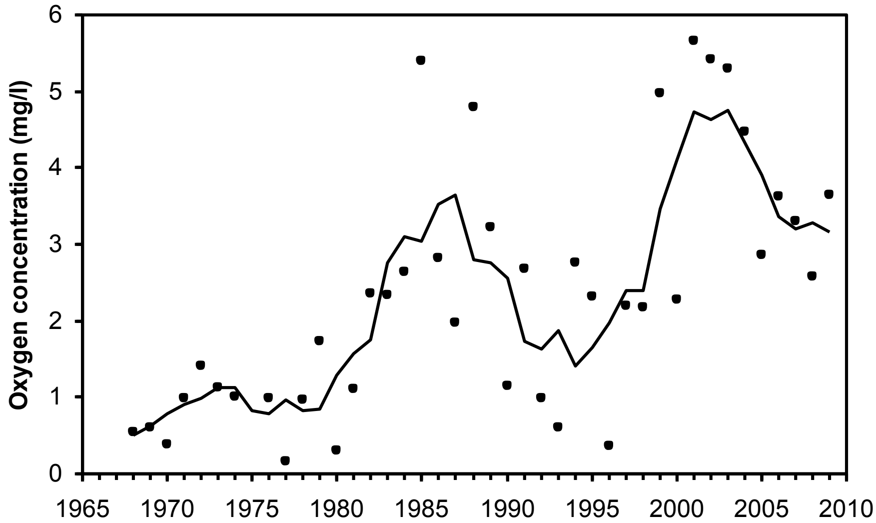


Figure 9: Annual mean bottom water oxygen concentrations from stations located in the Inner Stockholm Archipelago (compiled from data kindly provided by C. Lännergren, Stockholm Vatten).

Further reading

This study has been published in Environmental Research Letters.

Steckbauer, A., Duarte, C.M., Carstensen, J., Vaquer-Sunyer, R., Conley, D.J. (2011) Ecosystem impacts of hypoxia: thresholds of hypoxia and pathways to recovery. Environmental Research Letters 6:025103, doi:10.1088/1748-9326/6/2/025003.

The loss of benthic vegetation sustains a turbid regime

Key message

Seagrasses constitute an important biotope in coastal ecosystems, but there has been a global decline over the last century. This is a consequence of eutrophication stimulating growth of plankton and thereby reducing the light penetrating to the bottom. In response to this the depth limit of seagrasses, in temperate waters typically eelgrass, has decreased. Relationships linking nutrient levels with eelgrass depth limits, established on data during the eutrophication phase, have been proposed as nutrient management tools. However, such relationships are not valid for predicting the response to decreasing nutrient levels due to shifting baselines and feed-back mechanisms, where lack of benthic vegetation increases resuspension of sediments and thereby maintains a stable turbid regime.

Evidence

Nitrogen concentrations have decreased by 30-40% in many Danish estuaries following nutrient management plans addressing both point and diffuse sources. However, the decline has not been proportional across the different nitrogen fractions, and most of the decline is attributed to the dissolved inorganic fraction. The uneven reduction of the different nitrogen fractions has repercussion for the attenuation of light, which is related to the dissolved and particulate organic fractions. Moreover, resuspension of inorganic material from sediments appear to have increased over time, consistent with the decline of eelgrass, indicating that lack of eelgrass enhances resuspension of sediments (Figure 10). Thus, this suggests a possible feed-back mechanism that could lead to alternative stable states: 1) a clear state with eelgrass versus 2) a turbid state without eelgrass.

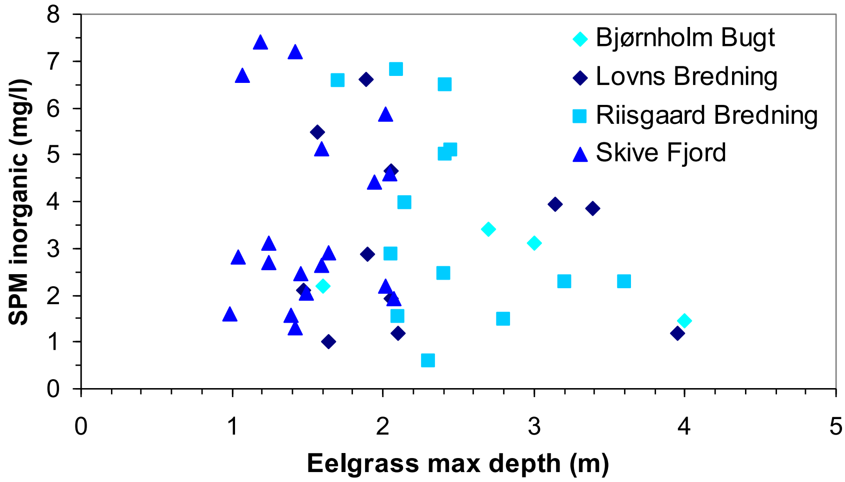


Figure 10: The concentration of suspended inorganic material versus eelgrass depth limits (annual means).

Implication

Large coastal areas are now characterised by bare sediments, where eelgrass meadows used to flourish. Re-establishment of benthic habitats with eelgrass requires improved light conditions, which has not been observed in the study area yet, despite large decreases in nutrient levels. The presence of eelgrass is important since it stimulates the sedimentation of particulate matter. Eelgrass can recolonise an area in two ways: 1) vegetative colonisation and 2) seed colonisation. A turbid regime with continuous resuspension of sediments is not favourable to seed colonisation, which means that vegetative colonisation appears to be the dominant pathway for re-establishing the eelgrass meadows. Vegetative recolonisation is, however, rather slow (<0.5 m/yr) suggesting that eelgrass recovery can take decades or even longer without any other intervention.

Further reading

This study will be contributed as WISER deliverable D5.3-4 and as a contribution to the WISER special issue in Hydrobiologia (downloadable at http://www.wiser.eu/results/deliverables/).

Ecological regime shifts affect seagrass pressure-indicator responses and delay recovery

Key message

Ecological regime shifts affect the response of seagrass indicators to pressures and may delay restoration of seagrass meadows upon release of pressure.

Evidence

We quantified and compared benthic and pelagic gross primary production (GPP) along nutrient gradients in time and space in a shallow estuary. The estuary experienced a shift from a pristine, seagrass-dominated clear water regime with high total GPP in the early 20th century to a eutrophic, plankton-dominated regime still with high total GPP in the 1980s when nutrient loadings peaked. Recent reductions in nutrient loadings reduced pelagic GPP as expected, but the water remained unclear and seagrass abundance and GPP did not increase correspondingly. The results suggest that feedback mechanisms, such as increased resuspension of the seafloor and reduced trapping of particles and nutrients, resulting from the loss seagrasses and their associated ecosystem services delay or prevent restoration to a state with seagrass dominance.

Implication

Ecosystems do not necessarily respond linearly to changes in nutrient loadings and that the response to eutrophication and oligotrophication may follow different trajectories. Reductions in nutrient loadings to levels below those causing the decline in seagrasses may be necessary, along with initiatives to e.g. reduce the disturbance of the seafloor, in order to stimulate a return to a seagrass-dominated state.

Further reading

This study was included in WISER deliverable D4.2-2 and published in Estuaries and coasts.

Krause-Jensen D, Markager S, Dalsgaard T (2011) Benthic and pelagic primary production in different nutrient regimes. *Estuaries and coasts*. DOI 10.1007/s12237-011-9443-1

7 Uncertainty in water body assessment

WISER improved the knowledge on the sources of uncertainty in ecological status classification

Key message

Knowing the main sources of uncertainty in WFD metrics informs the design of effective WFD monitoring programmes, assessment of ecological status and design of programmes of measures. The WISER project has improved our understanding of many of these sources of uncertainty.

Evidence

In the WISER project, an understanding of several potential sources of sampling uncertainty (spatial, sampling/sample processing) was built into the foreground sampling programme, aspects of temporal uncertainty could be investigated using the collated background datasets. However WISER uncertainty analysis was not able to address ALL potential sources of uncertainty in any individual BQE. Results from lake phytoplankton and macrophyte uncertainty analysis show that between-lake variation in metrics is greater than within-lake variation and between-analyst variation. Within-lake variation in phytoplankton metrics was small, and within-lake variation in macrophyte metrics was generally consistent and could be managed by sampling sufficient replicate transects. These results give confidence in the use of these BQEs for waterbody-level assessment of ecological status.

Implication

Understanding of the different sources of uncertainty and their relative magnitudes makes uncertainty manageable and is an essential part of the design of effective monitoring programmes. This includes standard protocols for sampling and laboratory processing, and it includes consistent staff training. The methodological requirements for a sampling programme for WFD status assessment may differ from those of more academic study. Current monitoring programmes may have replication where it is not needed, while they may be ignoring other more important aspects of uncertainty. Uncertainty exists in all freshwater biomonitoring, ideal BQEs which responds strongly to single stressors and can be measured with minimal uncertainty are rare or non-existent. Authorities should acknowledge uncertainty in reporting all water body assessments.

Further reading

Detailed descriptions of the data basis, analytical methods and results may be found in WISER's Deliverable D3.1-3 ('Report on uncertainty in phytoplankton metrics') and D3.2-2

('Report on uncertainty in macrophyte metrics'). Both Deliverables are downloadable from www.wiser.eu/results/deliverables/.

Uncertainty may vary between different metrics calculated for the same BQE

Key message

Many different assemblage metrics (e.g. using various combinations of taxon tolerance values, richness, abundance, traits) can be calculated for a single BQE. The selection of candidate metrics for assessment should be informed by the residual sampling variance of individual metrics, as well as their indicator value for particular stressors. This variability can itself vary considerably among different metrics describing the same BQE.

Evidence

Some comparisons could be made between alternate metrics based directly on taxonomic composition (including morpho-types) and metrics based on bio-physical (e.g. macrophyte maximum colonisation depth) or biochemical measures (e.g. chlorophyll a concentration). Results were mixed. Improved taxonomic resolution reduces uncertainty of taxonomy-based metrics: Phytoplankton PTI metric (taxonomic) showed clearly lower uncertainty than SPI metric (based on phytoplankton size groups). Replicate sampling uncertainty for chlorophyll a was low.

Implication

In general, metrics with low sampling uncertainty relative to their stressor response should be used. Metric specification is likely to need to include specification of sampling and laboratory protocols. Status assessment can be made more precise if it combines taxonomic and biophysical/biochemical measurements which show low sampling uncertainty, but metrics with high sampling uncertainty should not be used or combined.

Further reading

Detailed descriptions of the data basis, analytical methods and results may be found in WISER's Deliverable D3.1-3 ('Report on uncertainty in phytoplankton metrics') and D3.2-2 ('Report on uncertainty in macrophyte metrics'). Both Deliverables are downloadable from www.wiser.eu/results/deliverables/.

The WISER Bioassessment Uncertainty Guidance Software (WISERBUGS) helps water managers quantify the sampling uncertainty and confidence of water body ecological status classification

Key message

The new WISER Bioassessment Uncertainty Guidance Software (WISERBUGS) provides a flexible general means of using sampling uncertainty simulations to assess confidence in estimates of Ecological Quality Ratios (EQRs) and derived WFD ecological status class for water bodies. Assessments may be based on single metrics or a combination of metrics including multi-metric indices (MMIs) and multi-metric rules and involving metrics from one or several Biological Quality Elements (BQEs) for any type of water body with appropriate data.

Evidence

Users must provide prior estimates of the relevant sampling uncertainty for each metric to be involved in their water body assessments, together with metric status class limits and the rules for combining metrics and maybe BQEs into an overall water body assessment. Options include worst-case (One-Out-All-Out), mean and median class rules and the use of weighted multi-metric indices. Several WISER deliverables provide examples of how to derive the relevant sampling uncertainty measures for input to WISERBUGS (see also Figure 11).

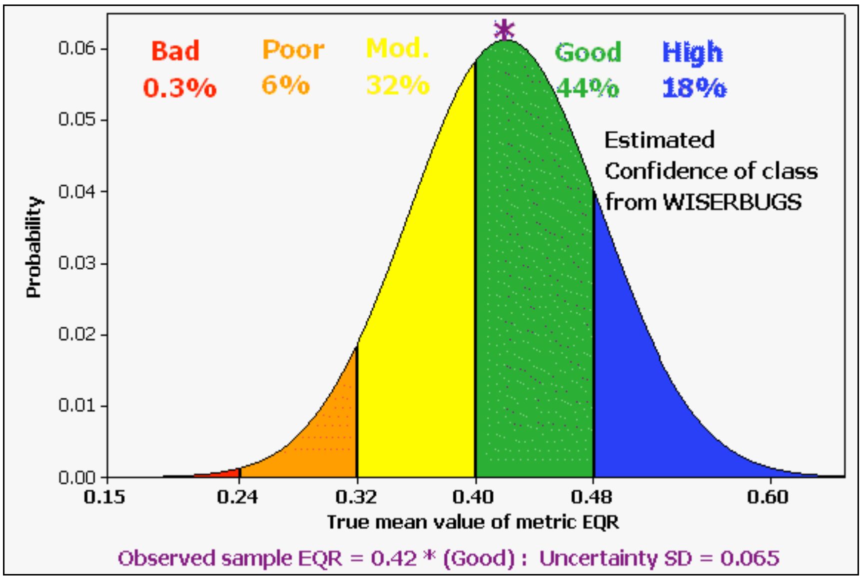


Figure 11: Illustrative example of estimated probabilities (i.e. confidence) of a water body belonging to each WFD status class based on WISERBUGS uncertainty simulations. Example is for a single metric EQR with an observed mean value of 0.42 for the water body and sampling uncertainty SD of 0.065, class limits are 0.48, 0.40, 0.32 and 0.24; based on the status classification of 10000 simulations of alternative possible sample mean EQR values for the water body, the estimated probabilities for the true water body class are 44% Good (the observed class), 19% high, 32% Moderate, 6% Poor and 0.3% Bad, leading to 38% probability that the true class is Moderate or worse.

Implication

The WISERBUGS software can help agencies with monitoring responsibilities and catchment managers quantify the confidence associated with their estimates of water body WFD status class, as required by the WFD. It is especially useful for providing water body classifications and the confidence consequences for multi-metric and multi-BQE integrated assessments in both trial and operational use. It can be used for metric EQR-based status class assessments of any type of water body (rivers, lakes, transitional or coastal waters).

WISERBUGS can also be used just to test the effect of new status class limits and multi-metric rules on site/waterbody status assessments, without any uncertainty assessment (by setting all uncertainty components to zero).

Further reading

WISERBUGS is available for download at http://www.wiser.eu/results/software/ (Windows version only). A documentation of the software and its applications is available as Deliverable D6.1-3 at http://www.wiser.eu/results/deliverables/.

Spatial heterogeneity is the main source of uncertainty when classifying ecological status using marine macrophyte indices

Key message

A wide variety of methods that use macrophyte communities for water body quality assessment fulfilling the complex requirements of the WFD have been developed by different Member States. Uncertainty analyses are a powerful tool to identify and quantify the factors contributing to the potential misclassification of the ecological status class of water bodies. When applied to different classification methods based on macrophytes, uncertainty analyses revealed that the factors related to the spatial scale of sampling (both horizontal and vertical) are the main source of uncertainty. On the contrary, the uncertainty associated to both temporal variability and surveyor is very low. In addition, the risk of misclassification also depends on the width of the status class in which the EQR score falls, with narrower range classes leading to greater probabilities of misclassification. Thus, indices which EQR range is not equally split into the 5 official quality status classes present different uncertainty levels along the EQR range.

Evidence

We conducted uncertainty analyses on EQR datasets of monitoring programmes using different macrophyte-based classification methods developed by different European Member States (Norway, Denmark, Bulgaria, Spain, Croatia, Italia and Portugal). These datasets included factors representative of the key sources of variability associated with the design and implementation the monitoring programs: the spatial and temporal scales of sampling, as well as the human-associated source of error. The spatial scale of sampling accounted for an average proportion of 39±10.2% of total variance among the different indices, whilst the temporal scale and the human-associated source of error only 4.5 ± 1.5% and 2 ± 2% respectively (in mean ± SE).

Implication

This study identifies the elements of a sampling design constraining the reliability and robustness of the ecological status classification of coastal water bodies. Once the major sources of variability are known, they can potentially be minimised through the re-design of sampling schemes, through improved training by operating procedures, etc. Horizontal spatial heterogeneity must be captured by sampling at different scales, providing robust estimates of the ecological quality status classification at the water body level that minimize the risk of misclassification. Depth should remain fixed or be controlled in monitoring programs in order to minimise vertical heterogeneity, except for indices based in the depth limit of macrophyte communities. Those indices where the distance between boundary classes is not uniform across the EQR range may need to assign a greater sampling effort to water bodies whose EQR score falls within the narrower status classes, in order to reduce their associated variability and increase the confidence of the classification. In contrast, sampling frequency has little effect on the precision of ecological status estimates.

Further reading

This study will be included in WISER deliverable D4.2-3 and published in special issue of Hydrobiologia on WISER.

Mascaró, O., Alcoverro, T., Dencheva, K., Krause-Jensen, D., Marbà, N., Neto, J., Nikolić, V., Orfanidis, S., Pedersen, A., Pérez, M. and Romero J. Exploring the robustness of different macrophyte-based classification methods to assess the ecological status of coastal and transitional ecosystems under the WFD. Hydrobiologia (submitted).

A smart sampling design may help reduce the uncertainty in lake assessment

Key Messages

The sources of uncertainty in water body assessment are manifold, but in part can be subjected to methodological issues. A smart sampling design may help reduce the level of uncertainty caused by, for instance, spatial and temporal variability or by individual researcher-dependent skills. In brief:

- Phytoplankton assessment should be based on at least 6 samples from the pelagic euphotic zone with higher frequency in eutrophic lakes, especially to catch harmful blooms. Standard methods and training should be used for sampling and analyses.

- Macrophyte field method should be based on transects covering all depth zones and different habitats.
- Macroinvertebrate assessment of shoreline modifications should be based on composite or habitat specific sampling (depending on region) at various stations representing the whole range of morphological shore modification.
- Fish assessment should be based on sampling of all depth strata with many gillnets. Hydroacoustic methods provide cost-effective assessment of fish abundance.

Evidence

In-lake variability of the various BQE metrics has been assessed from new WISER data sampled in ca. 21–51 lakes in 2009. 21 lakes were sampled for all four BQEs, while additional lakes were sampled for some BQEs.

Within-lake variability caused by natural spatial variation, as well as variability related to sampling and analyses was low for phytoplankton (Table 6 and 7), although this BQE has higher temporal variability related to sampling frequency. To minimize the risk of misclassification lake phytoplankton should be sampled on several occasions, although the minimum recommended frequency varies dependent on the metric and GIG (Table 8). Sampling should be more frequent in eutrophic lakes to increase the probability of catching harmful blooms.

For lake macrophytes, the metrics tested for variability is on the average 25–30% with station as the major variance component (Dudley et al. 2011). Thus, to reduce misclassification of macrophyte metrics several stations should be sampled to cover all major habitat types in the littoral zone, and sampling at each station should also cover the whole vertical extension of the littoral zone. The latter is important as nutrient enrichment reduces the growing depth of macrophytes. Assessment methods based on real hydrophytes are most sensitive to eutrophication, whereas helophytes are less affected by water quality. Helophytes should be sampled if water level fluctuation or hydromorphological changes are assessed.

Table 6: Major sources and levels of uncertainty detected for the lake BQEs within the WISER project. (Taken from Mischke et al. 2012)

BQE	Major variance component	Overall natural + methodological variability
Phytoplankton	Temporal (seasonal)	Small (< 25%)
Macrophytes	Spatial	Medium (30%)
Benthic fauna	Spatial (station)	Medium (30–40%)
Fish fauna	Spatial (depth stratum)	Large (> 90%)

For littoral macroinvertebrates, the major sampled variability was between sites, but this was partly (8–12%) due to consistent effects of morphological habitat modification type. Thus habitat specific sampling at various stations for each level of morphological modifications of the habitat will probably reduce the metric variability.

For fish the major variance component is depth stratum, implying that fish metrics should not be assessed without sampling all the depth strata in a lake. Biomass estimated from hydroacoustic methods versus that estimated from gill nets are well correlated in most lakes, except in very deep lakes (mean depth >30m) where hydroacoustic methods give higher estimates than gill nets for the deeper strata.

Table 7: Metric precision given as proportion of the total variance (i.e. within- and between lake variance) due to within-lake variability, and major within-lake variance components for four BQEs. Metrics with the lowest within-lake variance are the most precise whole-lake metrics. For benthic invertebrates, the in-lake variance incorporates variability associated with different levels of morphological pressure. See table 2 for explanation of metrics. (Taken from Thackeray et al. 2012)

BQE	Metric	Within lake variance (excluding temporal variability*)	Major variance component (excluding temporal variability*)
Phytoplankton*	Chl-a	0.04	Sub-sampling
	PTI	0.12	Sub-sampling
	SPI	0.35	Analyst
	MFGI	0.14	Sub-sampling
	J' (Evenness)	0.31	Analyst
	Cyano blooms intensity	0.06	Sub-sampling
Macrophytes	ICM	0.28	Station
	EI	0.26	Station
	Cmax	0.30	Station
Benthic fauna	Evenness	0.73 **	Station
	NTaxa	0.37 **	Station
	NTaxa EPTCBO	0.44 **	Station
	%POM_HabPref	0.52 **	Station
Fish	BPUE (log10)	0.999	Depth stratum
	CPUE	0.962	Single gillnets

*Temporal variability in phytoplankton is estimated to ca. 14% (coefficient of variation) for monthly sampling in some UK lakes. For more info on temporal variation and recommendations of sampling frequency, please see Mischke et al. 2012.
** includes within-lake variance of 8-12% due to margin modification type (Undisturbed ,Soft modifications, Hard modifications)

Table 8. Minimum recommended sampling frequencies for three phytoplankton metrics in three GIGs. The number of months and years mean 1 sample taken for each of the number of months in each of the number of years. For example for NGIG, chlorophyll a should be sampled at least once in 2 different months in each of 3 different years or once in 3 different months in each of 2 different years, meaning 6 samples altogether.

	Central Baltic GIG	**Mediterranean GIG**	**Northern GIG**
Chlorophyll *a*	3 months for 4 years	3 months for 3 years	2 months for 3 years or 3 months for 2 years
PTI	2 months for 4 years or 1 month for 6 years	3 months for 3 years or 1 month for 6 years	3 months for 3 years or 1 month for 6 years
Cyanobacteria	1 month for 6 years	1 month for 6 years	1 month for 6 years

Implication

Different BQEs and metrics require different monitoring and sampling designs based on the dominant sources of uncertainty.

For phytoplankton, the greatest source of variability is seasonal variability and analytical variability. The former can be reduced by utilising metrics based on repeated sampling during specific seasons (e.g. growth season or summer months) with higher frequency in eutrophic lakes, especially to catch harmful blooms. Minimum sampling frequency varies by metric and GIG, but should always cover the late summer period (Table 8). The analytical variability can be reduced by following standard counting guidance and consistent training within Member States and across Europe.

Macrophyte field method should be based on transects covering the whole depth zone and different littoral habitats. Sampling can be restricted to hydrophytes in lakes dominated by eutrophication pressure, whereas helophytes should be sampled if water level fluctuation or hydromorphological changes are assessed. More transects are needed at both ends of the trophic gradient to reduce uncertainty in status assessment.

Macroinvertebrate assessment of shoreline modifications should be based on composite or habitat specific sampling (depending on region) at various stations representing the whole range of morphological shore modification. The calculation of the whole-lake assessment score may be supported by conducting a physical habitat survey along the whole lake perimeter, relating this to the respective biological MMI, and then calculate a weighted average of site-specific MMI scores.

Fish assessment should be based on sampling of all depth strata with many gillnets. Hydroacoustic methods provide cost-effective assessment of fish abundance.

Further Reading

Dudley B, Dunbar M, Penning E, Kolada A, Hellsten S, & Kanninen A. 2011. WISER Deliverable D3.2-2: Report on uncertainty in macrophyte metrics

Mischke, U., Stephen Thackeray, Michael Dunbar, Claire McDonald, Laurence Carvalho, Caridad de Hoyos, Marko Jarvinen, Christophe Laplace-Treyture, Giuseppe Morabito, Birger Skjelbred, Anne Lyche Solheim, Bill Brierley and Bernard Dudley 2012. Deliverable D3.1-4: Guidance document on sampling, analysis and counting standards for phytoplankton in lakes.

Thackeray S, Nõges P, Dunbar M, Dudley B, Skjelbred B, Morabito G, Carvalho L, Phillips G, Mischke U. 2011. WISER Deliverable D3.1-3: Uncertainty in Lake Phytoplankton Metrics, June 2011.

Winfield, I. J., Emmrich, M., Guillard, J., Mehner, T., Rustadbakken, A., 2011. Guidelines for standardisation of hydroacoustic methods. WISER deliverable 3.4-3.

Uncertainty levels associated with metric variability in multi-metric fish indices can be managed to increase the confidence in ecological status class assignment

Key message

Technical and monitoring design factors (gear, sampling season, and survey protocol including sampling effort), and natural and anthropogenic pressures all affect the variability of fish metrics. The within-system variability is notably larger than the between-system variability. This effect is probably due to natural factors and sampling bias and hence the standardization of sampling methods and more robust fish metrics will increase the robustness of the use of the BQE fish in transitional waters

Evidence

Potential 'noise' factors (i.e. inherent variability) confounding biological quality metrics can be technical (i.e. those linked to the method of assessment including sampling effort) or natural (physicochemical and biological). We applied linear models using fish metrics as response variables and a suite of covariates to explain the metric scores and identify the sources of variability affecting them. The resulting best models contained from 3 to 14 covariates but explained only a relatively small amount of the total variance. With the available dataset, the best models explained less than 40 % of fish metric variability (with a maximum 22% for lagoons and 40% for estuaries). The remaining variability was mainly within-estuary or lagoon and can probably be attributed, at least in part, to both a habitat effect that was not accounted for in the models and to the influence of biological interactions in influencing community structure.

The effect of sampling effort on fish metrics could not be assessed in the previous analysis but this factor will have an important effect on the variability of fish metrics. The analysis showed that sampling effort is an important source of variability in fish metrics of the EFAI index, especially metrics dependent on number of species, which are common to several other fish-based indices (see figure below). In turn, metrics based on percentages (derived from the abundance of marine migrants, estuarine residents, piscivorous species) showed a lower sensitivity to the increase in sampling effort, with values stabilizing after a fewer hauls compared to metrics based on species richness. The stabilization of metrics based on species richness varied between salinity zones, with an increasing number of hauls generally required at higher salinities. In contrast, salinity zone did not have that effect on metrics presented as percentage abundance for different guilds.

The sensitivity of richness-based metrics is caused by including in the analysis species with an apparent abundance below a certain threshold, which prevent the complete characterisation of their presence. These rare species, in some cases a single individual collected on a single occasion, would only be incidentally recorded and therefore add random variability to diversity-based metrics. This in turn affects the relative scores and the outcomes of the assessment. A similar effort-related bias may be an issue for density-based metrics if fish distribution is very patchy (i.e. schooling fish or those aggregated in specific habitats) and insufficient replicates are taken to fully characterise the patchiness in their distribution. It is apparent that to overcome a potential large source of error, the reference conditions must be defined according to the level of effort used in the monitoring programme or, conversely, the monitoring must be carried out at the same level of effort used to derive the reference.

Improving accuracy without having to increase efforts may be possible by greater use of proportion metrics or the use of less-selective gear sets or multi-gear approaches. Alternatively, a more pragmatic decision could be made based on the probability of capture, thus considering in the analysis only those aspects for which the sampling method and level of effort allows for a reliable and quantitative estimation. Two possible options were identified: (1) weighting of metrics by precision or by species relevance, or (2) pooling samples to ensure sampling events provide greater habitat or temporal integration (i.e. larger effective samples).

Implications

A minimum effort is required to minimize misclassification (i.e. prevent wrong ES quality class assignment). A better, more robust assessment may be possible but residual variability should be accounted for and explained and cannot be decreased without increasing the number of replicates (effort). Reducing uncertainty in ES assessments will require a better knowledge of habitat partition within systems, understanding of metrics behaviour and precision, testing new combination rules allowing metric weighting by robustness and importantly research on new and more robust sampling tools and methods.

Further reading

Courrat A, Lepage M, Alvarez MC, Borja A, Cabral H, Elliott M, Franco A, Gamito R, Neto JM, Pérez-Domínguez R, Raykov V, Uriarte A (2012) The ecological status assessment of transitional waters: an uncertainty analysis for the most commonly used fish metrics in Europe. In: WISER Deliverable D4.4-2 (part 2)

Pérez-Domínguez R, Alvarez MC, Borja A, Cabral H, Courrat A, Elliott M, Fonseca V, Franco A, Gamito R, Garmendia JM, Lepage M, Muxika I, Neto JM, Pasquaud S, Raykov V, Uriarte A (2012) Precision and behaviour of fish-based ecological quality metrics in relation to natural and anthropogenic pressure gradients in European estuaries and lagoons. In: WISER Deliverable D4.4-5

8 Integration of different Biological Quality Elements

The 'one-out all-out' principle for combining multiple BQEs into an integrated classification must be applied with caution

Key message

Although the WFD requires the use of the 'one-out all-out' rule in classifying the biological status of a water body, its strict application is not always recommended because of the risk of downgrading sites too easily. The 'one-out all-out' rule works best if the redundancy between BQEs is as low as possible.

Evidence

The 'one-out all-out' (OOAO) is the required principle by the WFD, classifying the biological status of a water body on the basis of the biological quality element (BQE) with the worst class score (Classification guidance, 2003). This rule is very precautionary, based on the assumption that different BQEs respond to pressures in different ways and that there is a need to protect the most vulnerable biological group. However, its strict application is not always recommended because there is a risk of downgrading sites too easily.

In WISER's WP6.2 this was demonstrated using monitoring data sets and modelled data. Monitoring datasets from Swedish lakes assessed with up to four BQEs (phytoplankton, macroinvertebrates, macrophytes, fish) and Austrian rivers with two BQEs (macroinvertebrates and fish) were used to demonstrate the effect of different combination rules on classification outcome. In all cases, the OAOO rule gave the highest probability of classifying water bodies in moderate or worse status compared to using the average or median (Figure 12). Uncertainty in estimates of ecological status class for water bodies was calculated using the software WISERBUGS (Clarke 2010, http://www.wiser.eu/results/software/).

Simulations with artificial data demonstrated that, when combining multiple BQEs that are sensitive to the same pressures or combination of pressures, the OOAO rule produced unbiased results and good class agreement only when metrics had a low level of uncertainty (SD ≤0.01), which in practice is very difficult to achieve. The reliability of the classification was already very sensitive at a moderate level of metric uncertainty (SD >0.05) (Figure 13A). An alternative rule tested for combining the same set of BQEs was the average rule, producing better results for high uncertainty metrics (Figure 13B). However this is not in accordance with the WFD guidance, as averaging among BQEs is not recommended.

Implication

The uncritical application of the 'one-out all-out' (OOAO) principle could pose the danger of downgrading status class of water bodies too easily. In particular, water managers should be careful when multiple BQEs that are redundant for detecting the same pressure, or combination of pressures, need to be combined into a water body assessment. It has been demonstrated that the OOAO approach only gives acceptable and comparable results if the different BQEs are complementary, showing the effects of different pressures, on different temporal and/or spatial scales, on different aspects of ecosystem functioning. Also the level of uncertainty in the biological metrics and in the BQEs used in the assessment should not be too high and not too different between BQEs.

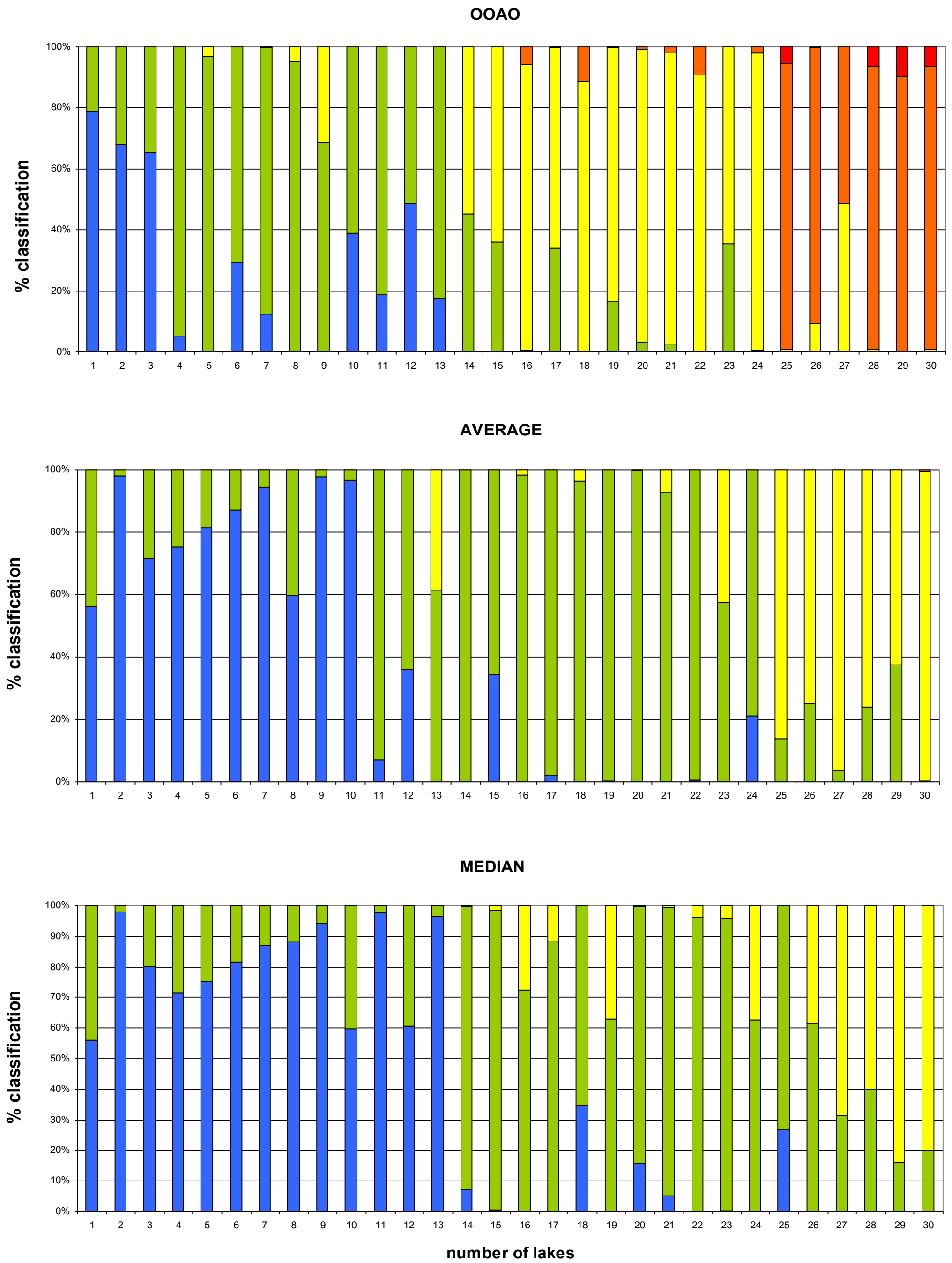
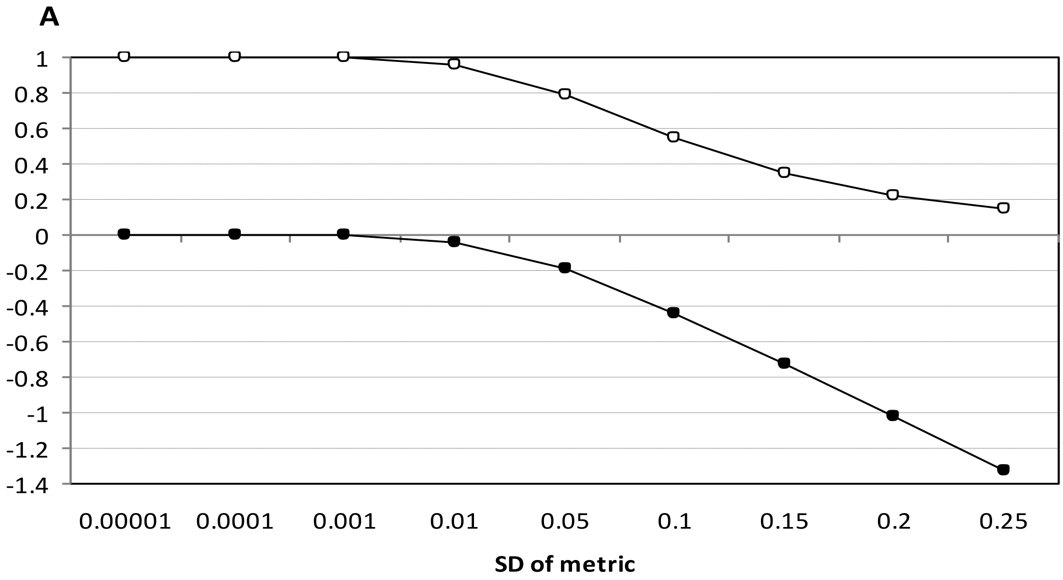


Figure 12. Example of the effect of different combination rules (OOAO, average, median) for multiple BQEs (phytoplankton, macroinvertebrates and macrophytes) on the assessment of the ecological status of 30 Swedish lakes.



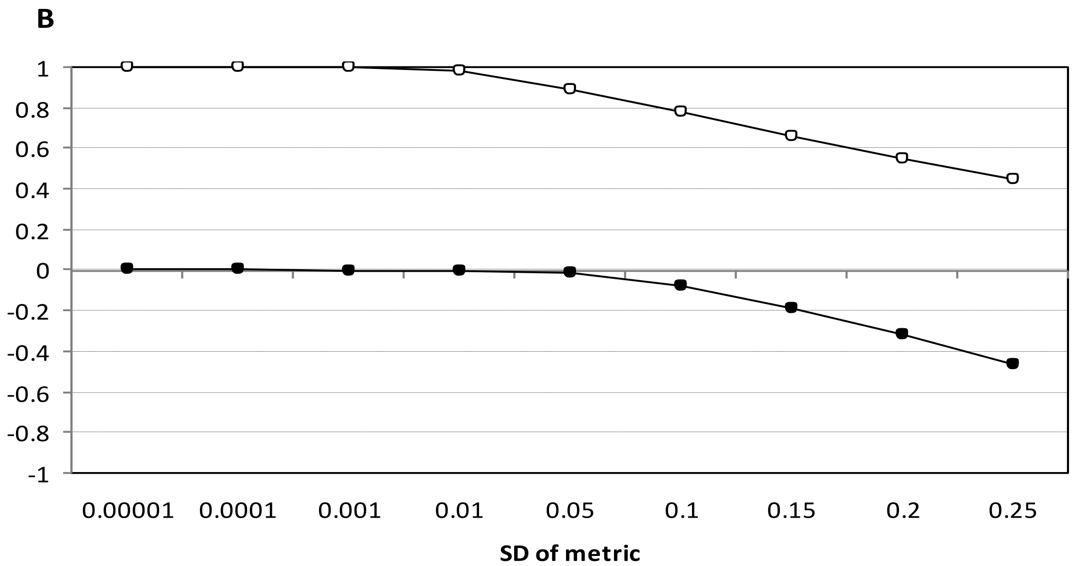


Figure 13. Multi-pressure BQEs. Multiple BQEs have been combined at water body level by A) OOAO B) averaging. Open circles indicate level of class agreement, full circles level of bias.

Further reading

Alahuhta, J., K-M., Vuori, S., Hellsten, M., Järvinen, M., Olin, M.Rask, and A., Palomäki, 2009. Defining the ecological status of small forest lakes using multiple biological quality elements and paleolimnological analysis. Fundamental and Applied Limnology, Archiv für Hydrobiology 175/3: 203-216.

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J., Ruuhijärvi, J., Jyväsjärvi, I., Kolari, M., Olin, E., Salonen, and P., Valkeajärvi, 2011. Ecological classification of large lakes in Finland: comparison of classification approaches using multiple quality elements. Hydrobiologia, 660: 37–47.

9 Management across water categories

Restoration can only become successful when all pressures are tackled simultaneously

Key message

Aquatic ecosystems are often simultaneously affected by multiple pressures, so consequently restoration must address these stressors simultaneously in order to be successful. For example, both the decrease in pH and increase in ammonium concentrations are associated with acid deposition. Phosphorus and nitrogen concentrations usually increase as a result of fertiliser run-off and the reduction of current speed coupled with an increase in siltation rate are associated with river canalisation. However, pressures are often water category specific. In general, rivers integrate the adverse effects of various human activities and associated pressures within a catchment, with hydromorphological degradation predominating, lake ecosystems are mainly affected by eutrophication and shoreline modification (at the global scale) and acidification (at the regional scale), while estuaries and coastal waters comprise the ultimate sink for nutrients, contaminants and other sources of pollution originating from entire river basins and are being physically.

Evidence

In most restoration projects measures are taken to reduce the primary stressor, but secondary stressors often confound recovery in lakes (see Figure 14). Confounding factors such as water quality, with particular emphasis on nutrient enrichment, large-scale hydrological change such as floods and droughts and catchment management/land use and multiple pressures cause delays or failures in aquatic system recovery.

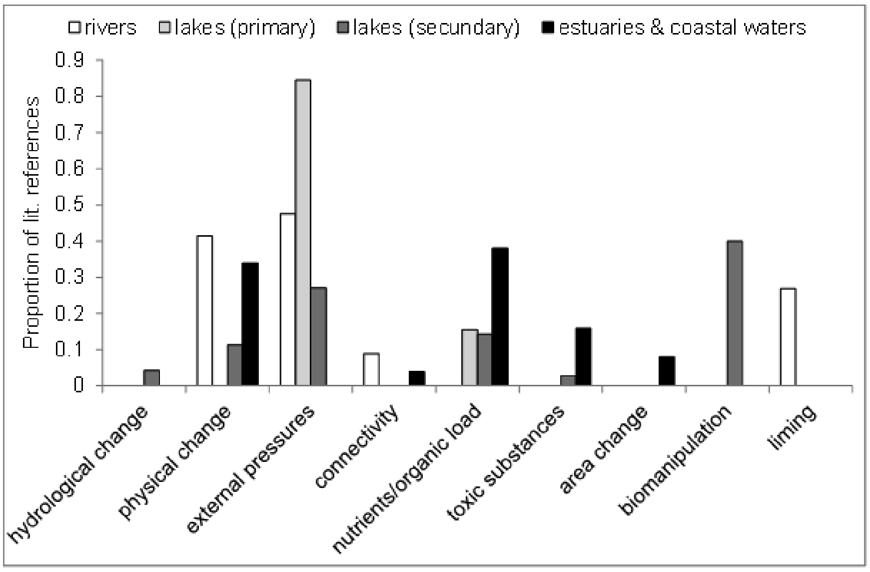


Figure 14: Proportion of literature references relating to restoration measures taken in rivers, lakes, estuarine and coastal waters, respectively. (No. of studies considered: rivers: 168, lakes: 343, estuaries and coastal waters: 51)

Recovery has not necessarily failed, but the presence of secondary pressures may have pushed response times beyond those over which monitoring is typically performed. Acidification, fisheries management, industrial pollution, non-native species and climate change were the main secondary pressures impacting de-eutrophication projects in aquatic systems. Especially, internal P loading slows down recovery in many eutrophied lakes.

Implication

Recovery depends on the type and magnitude of the pressures, especially if some are still present, and on the organism group(s) used to assess recovery. Delays in recovery can be attributed to several factors, and different water types are exposed to different combinations of pressures resulting in differences in response times.

What restoration ecology more in general, needs is:

- Definition of clear goals for restoration at catchment scale that are based on recent biological monitoring results and the actual distribution of targeted species or communities.
- Identification of best-practice restoration measures to address the specific pressures.
- Balancing all measures within a catchment in order to reach the best possible synergy effects of single component measures, and ultimately to achieve recovery of the entire catchment.

Further reading

Borja, A., Dauer, D.M., Elliott, M. and Simenstad, C.A. 2010. Medium- and long-term recovery of estuarine and coastal ecosystems: patterns, rates and restoration effectiveness. Estuaries and Coasts 33: 1249–1260.

Feld, C.K., Birk, S., Bradley, D.C., Hering, D., Marzin, A., Melcher, A., Nemitz, D., Pedersen, M.L., Pont, D., Verdonschot, P.F.M., Friberg, N., Natural, F., Feld, C.K., Birk, S., Bradley, D.C., Hering, D., Kail, J., Marzin, A., Pletterbauer, F. & Pont, D. (2011) From natural to degraded rivers and back again : a test of restoration ecology theory and practice. Advances in Ecological Research, 44, 119–209.

Søndergaard, M., Jeppesen, E., Lauridsen,T.L., Skov, C., Van Nes, E.H., Roijackers, R., Lammens, E. and Portielje, R. 2007. Lake restoration: successes, failures and long-term effects. Journal of Applied Ecology 44: 1095–1105.

Jowett, I.G., Richardson, J. and Boubée, J.A. 2009. Effects of riparian manipulation on stream communities in small streams: Two case studies. New Zealand Journal of Marine and Freshwater Research 43: 763–774.

Spears, B, Gunn, I., Meis, S. and May, L. 2011. Analysis of cause-effect-recovery chains for lakes recovering from eutrophication. CEH-report. Contribution to Deliverable D6.4-2.

Recovery needs time, long time

Evidence

Long-term studies of recovery in rivers, lakes and estuarine and coastal waters are scarce. One important question before comparing time spans of recovery between water categories is the definition of 'full recovery'. 'Full recovery' refers to an optimal functioning of the aquatic ecosystem under the given environmental circumstances that are not or only slightly changed by human activity. Literature for both riverine and marine systems addresses this issue (see Table 9 for marine examples), while for many lakes in lowland areas focus is more on a shift from turbid to clear water states. Monitoring for a large proportion of studies was < 5 to 10 years, and only a few studies (one each) in rivers and estuarine and coastal waters extended >20 years (Figure 15).

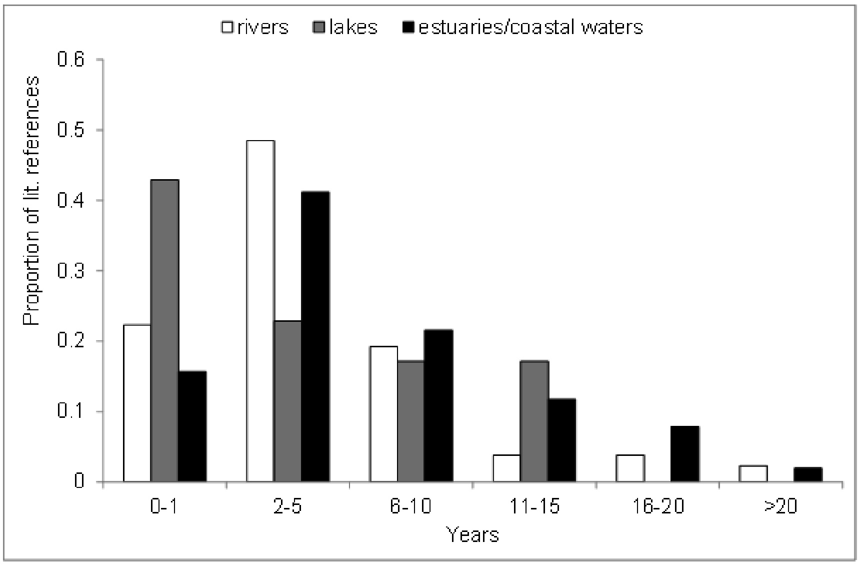


Figure 15: Proportion of literature references that reported on specific biological quality elements in river, lake and estuarine and coastal water restoration studies.

Table 9: Summary of time for recovery, for different biological elements and substrata, under different pressures in marine ecosystems. (Taken from Borja et al. 2010)

Pressure	Substrata	Intertidal/subtidal	Biological elements	Time for recovery
Sediment disposal	Soft	Intertidal	Meio and macrofauna	3-18 months
Marsh restoration	Soft	Subtidal	Fishes	1-2 yr
Oxygen depletion	Soft	Subtidal	Macroinvertebrates	2 yr
Land claim	Soft	Intertidal	Macroinvertebrates	2 yr
Oil-refinery discharge	Soft/Hard	Intertidal/Subtidal	Macroinvertebrates, fishes	2-3 yr
Dyke and marina construction	Soft	Intertidal/Subtidal	Macroinvertebrates, fishes	2-3 yr
Lagoon isolation	Soft	Subtidal	Molluscs	>3 yr
Aggregate dredging	Soft	Subtidal	Macroinvertebrates, epifauna	2-4 yr
TBT	Soft	Subtidal	Macroinvertebrates	3-5 yr
Dredging	Soft	Intertidal/subtidal	Seagrasses, macroinvertebrates, fishes	2->5 yr
Sediment disposal	Soft	Subtidal	Seagrass, Macroinvertebrates, fishes	>5 yr
Eutrophication	Soft	Subtidal	Macroinvertebrates	>3->6 yr

Pressure	Substrata	Intertidal/subtidal	Biological elements	Time for recovery
Realignment of coastal defences	Soft	Intertidal	Marshes and macroinvertebrates	>6 yr
Fish farm	Soft	Subtidal	Macroinvertebrates	2->7 yr
Physical disturbance	Soft/Hard	Intertidal/Deep-sea	Macroinvertebrates, megafauna	3->7 yr
Pulp mill	Soft	Subtidal	Macroinvertebrates	6-8 yr
Oil-spill	Soft/hard	Intertidal/subtidal	Various	2-10 yr
Fish trawling	Sand-gravel	Subtidal	Macroinvertebrates, fishes	2.5-10 yr
Wastewater discharge	Soft	Subtidal	Fishes	3-10 yr
Sewage sludge disposal	Soft	Subtidal	Macroinvertebrates	3->14 yr
Mine tailings	Soft	Subtidal	Macroinvertebrates	4->15 yr
Marsh and tidal restoration	Soft	Intertidal/subtidal	Vegetation, fishes, birds	5-20 yr
Wastewater discharge	Soft	Subtidal	Macroinvertebrates, seagrasses	7-20 yr
Land claim	Soft	Subtidal	*Zostera marina*	>20 yr
Wastewater discharge	Hard	Intertidal	Macroalgae	>6->22 yr

Large discrepancies exist between the length of monitoring programmes and the time needed for the ecosystem to reach ‘full recovery’. Although most studies do not address ‘full recovery’, some estimates are available. Recovery after weir removal may take as long as 80 years. Recovery after riparian buffer instalment may take at least 30–40 years. In lakes, time for recovery from eutrophication varies from 10–20 years for macroinvertebrates, 2 to >40 years for macrophytes, 2 to >10 years for fish. Natural recovery from acidification takes much longer compared to recovery after liming, and like eutrophication, biological recovery is taxon specific and often decades are needed to achieve pre-disturbed conditions. Estuarine and coastal waters have long periods of recovery (>10 years), although macroinvertebrates have the potential to recover within months to <5 years though mostly take >6 years. Fish recover within one to three years, depending on the type and intensity of pressure. In general, after intense and large pressures, periods of 15–25 years for attainment of the original biotic composition, diversity and complete functioning may be needed in all three water categories.

In both rivers and lakes the success rate of restoration measures appears to be much higher for the abiotic conditions than for the biotic indicators, this is particular true for hydromorphological restoration and liming. Since eutrophication is considered also to be an important pressure in rivers and lakes, this might be a major cause of hampering recovery. In lakes internal nutrient loading often delays recovery. For rivers the response of macroinvertebrates to hydromorphological restoration is questionable; some studies have shown recovery while other studies do not possibility due to the still too high nutrient levels.

Implication

Only from monitoring of biological and environmental changes after restoration can new knowledge on recovery processes be gained and implemented. Indeed, this information provides the opportunity for practitioners and scientists to evaluate the success and efficacy of the restoration measures. Restoration monitoring requires a tailor-made sampling design (preferably a BACI-design) that allows of sound statistical analysis according to state-of-the-art methods. Surprisingly, the BACI design is primarily applied to experimental studies. A Before-After-Control-Impact (BACI) monitoring design is considered the best approach for monitoring recovery, as only this approach is capable of detecting actual effects of restoration from other natural effects, such as seasonal or annual variability.

Further reading

Bernhardt, E.S., Palmer, M.A., Allan, J.D., Alexander, G., Barnas, K., Brooks, S., Carr, J., Clayton, S., Dahm, C., Follstad-Shah, J., Galat, D., Gloss, S., Goodwin, P., Hart, D., Hassett, B., Jenkinson, R., Katz, S., Kondolf, G.M., Lake, P.S., Lave, R., Meyer, J.L., O'Donnell, T.K., Pagano, L., Powell, B. and Sudduth, E. 2005. Synthesizing U.S. River Restoration Efforts. Science 308: 636–637.

Borja, A., Dauer, D.M., Elliott, M. and Simenstad, C.A. 2010. Medium- and long-term recovery of estuarine and coastal ecosystems: patterns, rates and restoration effectiveness. Estuaries and Coasts 33: 1249–1260.

Gray, J.S. and Elliott M. 2009. Ecology of Marine Sediments: science to management. OUP, Oxford, 260 pp.

Reitberger, B., Matthews, J. , Feld, C.K., Davis, W. and Palmer, M.A. 2010. River monitoring and indication of restoration success: comparison of EU and BCG frameworks. University of Duisburg-Essen, Essen, 154 pp.

Smith, E.P., Orvos, D.R., and Cairns, J. 1993. Impact assessment using the before-after-control-impact model: concerns and comments. Can. J. Fish. Aquat. Sci. 50: 627–637.

Monitoring of restoration needs a before-after-control-impact design to learn by doing

Key message

Monitoring of restoration needs a before-after-control-impact (BACI) design to learn by doing. This BACI approach allows for a sound analysis of the effects of restoration, as it i) compares the conditions before and afterwards (BA) and relates the identified changes to potential natural changes at restored and unrestored control sites (CI). Hence, the BACI approach allows of a separation of restoration effects from natural variability.

Evidence

Despite the wealth of monitoring programmes focused on rehabilitating lotic systems, most studies are designed for local situations and address single pressures. For example, Bernhardt et al. (2005) stated that of 37 000 river restoration projects in the United States, only 10% included some form of monitoring, and the authors argued that the information was often inadequate to evaluate successes and failures. Similarly, an overview of 16 European papers on river restoration studied by Reitberger et al. (2010) showed that none of the studies, for example, analysed time series of restoration monitoring. Hence, these two reviews emphasize the need for high quality data to properly evaluate the efficacy of restoration effort and to make generalisations and improvement, which might increase the frequency of successes. Poor availability of data can be due to several reasons. First, an overwhelming majority of restoration measures have not included monitoring, probably because there is no legal requirement. Second, when restoration measures are monitored, the methods and time-scales applied are often inadequate considering knowledge of recovery time lags. Third, most water authorities do not focus on long term ecological processes, but focus instead on getting the job done, with little or no interest in properly evaluating the results.

Restoration monitoring usually, at best, follows a Before-After sampling design (Figure 16). Long-term time series data, commonly available for lakes are usually lacking for rivers and even less so for marine systems. For lakes, monitoring programmes typically do not encompass the pre-impact period, both for eutrophication and acidification.

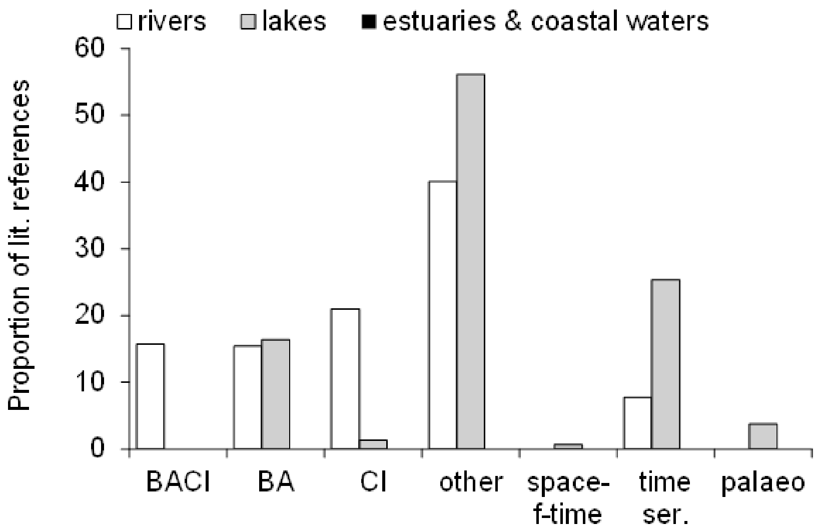


Figure 16: Proportion of literature references that refer to specific data evaluation techniques as applied in river, lake and estuary and coastal water restoration projects. (No. of references considered: rivers: 168; lakes: 343; estuaries and coastal waters: 51)

Implication

The time lags of recovery after removal of the stressor(s) are highly variable in all three water categories, from months to many decades. Recovery depends on the type and magnitude of the stressor(s), especially if some are still present, and on the organism group(s) used to assess recovery. Delays in recovery can be attributed to several factors, and different water types are exposed to different combinations of stressors resulting in differences in response times. Furthermore, there needs to be agreement upon the restoration goals for the system and also what criteria will be used to determine attainment of the desired or targeted system. For example, from the outset it should be stated whether a system is being restored merely for its abiotic features, its structural elements, i.e. the appropriate species, or full functioning.

Further reading

Bernhardt, E.S., Palmer, M.A., Allan, J.D., Alexander, G., Barnas, K., Brooks, S., Carr, J., Clayton, S., Dahm, C., Follstad-Shah, J., Galat, D., Gloss, S., Goodwin, P., Hart, D., Hassett, B., Jenkinson, R., Katz, S., Kondolf, G.M., Lake, P.S., Lave, R., Meyer, J.L., O'Donnell, T.K., Pagano, L., Powell, B. and Sudduth, E. 2005. Synthesizing U.S. River Restoration Efforts. Science 308: 636–637.

Feld, C.K., Birk, S., Bradley, D.C., Hering, D., Marzin, A., Melcher, A., Nemitz, D., Pedersen, M.L., Pont, D., Verdonschot, P.F.M., Friberg, N., Natural, F., Feld, C.K., Birk, S., Bradley, D.C., Hering, D., Kail, J., Marzin, A., Pletterbauer, F. & Pont, D. (2011) From natural to degraded rivers and back again : a test of restoration ecology theory and practice. Advances in Ecological Research, 44, 119–209.

Simenstad, C., Reed, D., and Ford, M. 2006. When is restoration not?: Incorporating landscape-scale processes to restore self-sustaining ecosystems in coastal wetland restoration. Ecological Engineering 26: 27–39.

Overview of WISER databases

The WISER Central Database stores biological and environmental data from nearly 20,000 samples in 26 countries

The Central Database (CDB) is composed of WP databases, i.e. 1-2 databases from each of the WPs 3.1-5.1. The WP databases contain both "foreground data" (i.e. data from the WISER field exercises) and "background data" (all other existing data). The WP databases were partly standardised before import to the Central DB, but the content was not quality-checked by WP2.1. Some of the WP databases contain details that are not included in the Central DB (e.g. climatic data or information on subsamples). All WP databases are available to the project partners from the WISER intranet.

A summary of the CDB content is given in Table 1 and the geographic extent is illustrated in Figure 1. Altogether the CDB contains data from 28 countries. The WISER field campaign ("foreground data") resulted in than 50 000 records of biological data, in ca. 8,300 samples from 405 stations in 69 water bodies in 14 countries. In addition, the foreground data contain almost 10,000 samples of environmental data. Moreover, the background data consist of ca. 114,000 biological samples and ca. 140,000 environmental samples from rivers, lakes and coastal/transitional waters 26 countries.

A hierarchical database structure was developed that could accommodate the various biological and physico-chemical data from all WPs, and which could enable data aggregation and extraction in any format requested by users.

Due to the intellectual property rights (IPR) associated with the datasets, the WISER data cannot be made available publicly available. Nevertheless, the publicly available metadatabase search tool can be used to find datasets of interest and contact information for relevant WISER partners. External persons who are interested in using the data are encouraged to contact the relevant WISER partners and propose collaboration using these data. An overview of IPR information for the WISER Central Database can be downloaded from http://www.wiser.eu/programme-and-results/data-and-guidelines/data-services/.

For more information on the WISER Central Database, please contact Jannicke Moe at the Norwegian Institute for Water Research (jmo@niva.no).

Table 1: Overview of content of the WISER Central Database (CDB). (A) Foreground data: data from the WISER field campaign. (B) Background data: from previous projects etc. Explanation to WP numbers: 3.1 - lake phytoplankton; 3.2 - lake macrophytes; 3.3 - lake macroinvertebrates; 3.4 - lake fish; 4.1 - coastal phytoplankton; 4.2 - coastal macroalgae and angiosperms; 4.3 - coastal macroinvertebrates; 4.4 - coastal/transitional fish; 5.1 - river phytobenthos, macrophytes, macroinvertebrates and fish. (A) Foreground data.

WP	Countries	# Water-bodies	# Stations	# Biol. samples	# Biol. values	# Env. samples	# Env. values	WP data manager
3.1	DE, DK, EE, ES, FI, FR, IT, NO, PL, SE, UK	32	104	262	11 868	986	3 158	Birger Skjelbred, Jannicke Moe
3.2	DE, DK, EE, FI, FR, IT, NO, PL, SE, UK	28	161	6 725	7 497	0	0	Bernard Dudley
3.3	DE, DK, EE, FI, IE, IT, SE, UK	53	150	96	2 159	150	150	Oliver Miler, Mario Lepage
3.4	DE, IT, UK	21	333	452	4 867	0	0	Stephanie Pedron, Simon Causse
4.1	BG, ES, FI, IT	6	43	42	2 903	0	0	Karsten Dromph
4.2	BG, ES, IT, NO, PT	8	72	331	1 881	8 357	25 847	Rosa G. Novoa
4.3	ES, IT, NO, PT	10	61	165	8 592	56	559	Karl Norling
4.4	BG, IT, PT, UK	7	72	213	489	213	803	Anne Courrat
Sum	**14**	**127**	**996**	**8 286**	**40 256**	**9 762**	**30 517**	

(B) Background data.

WP	Countries	# Water-bodies	# Stations	# Bio samples	# Bio values	# Env samples	# Env values	WP data manager
3.1	BE, CY, DE, DK, EE, ES, FI, FR, GR, HU, IE, IT, LT, LV, NL, NO, PL, PT, RO, SE, UK	6 619	10 632	16 861	463 837	123 844	768 225	Birger Skjelbred, Geoff Phillips
3.2	BE, EE, FI, IE, LT, LV, NL, NO, PL, RO, SE, UK	1 571	1 613	1 724	27 773	0	0	Bernard Dudley
3.3	BE, DE, EE, LT, LV, NL, PL, UK	180	635	889	23 016	0	0	Juergen Boehmer
3.4	DK, EE, ES, FI, FR, IE, IT, LV, LT, NO, PT, RO, SI, SE, UK	2 173	54 851	72 245	558 993	0	0	Stephanie Pedron, Simon Causse
4.2	BG, ES	32	63	1 836	6 463	3	3	Rosa G. Novoa
4.4	ES, FR, PT, UK	67	2 363	3 416	6 165	3 229	16 007	Anne Courrat
5.1	AT, CZ, DE, DK, FR, NL, PL, SE,	3 085	4 349	18 152	528 623	14 558	134 602	Andreas Melcher, Martin Seebacher

	SK, UK							
Sum	**26**	**12 882**	**74 506**	**115 123**	**1 614 870**	**141 634**	**918 837**	

The WISER Meta Database provides detailed information about 114 European project and monitoring databases

The WISER meta database provides information on the content, availability and accessibility of single databases from previous EU projects (FP5–7) and national monitoring schemes, as well as on the new data from the WISER field campaigns in 2009 and 2010. All information was provided by the data owners themselves, using an online questionnaire. The WISER meta database is accessible online at: http://www.wiser.eu/results/meta-database/.

An online query option helps the user find the available data based on several parameters, such as the region and water category of interest, the targeted stressors and biological quality elements, etc. As query results, you will find a summary of the selected parameters, the number of available sites, other available information and a list of the appropriate databases, from where you can directly link to the according metadatabase entry. An 'IPR column' indicates the Intellectual Property Rights of the database with a traffic light system, however, the user will easily recognise the limited number of freely usable databases (green traffic light).

To view all details of a single component database, just click on the database name in the result list, then a second window will open. To get an overview of the intellectual property rights (IPR) of the available databases, we have compiled a table, which can be downloaded at: http://www.wiser.eu/results/meta-database/.

For questions, the user should contact Astrid Schmidt-Kloiber (at BOKU Vienna; astrid.schmidt-kloiber@boku.ac.at). For the content of the metadatabase entries, the user should contact the contact the persons or institute as indicated in the section „Technical info".

When using the data, please acknowledge our work as follows: Schmidt-Kloiber A., Vogl R., Moe J. & Strackbein J. 2010. WISER metadatabase. Version: November 2010. Available at http://www.wiser.eu/results/meta-database/.

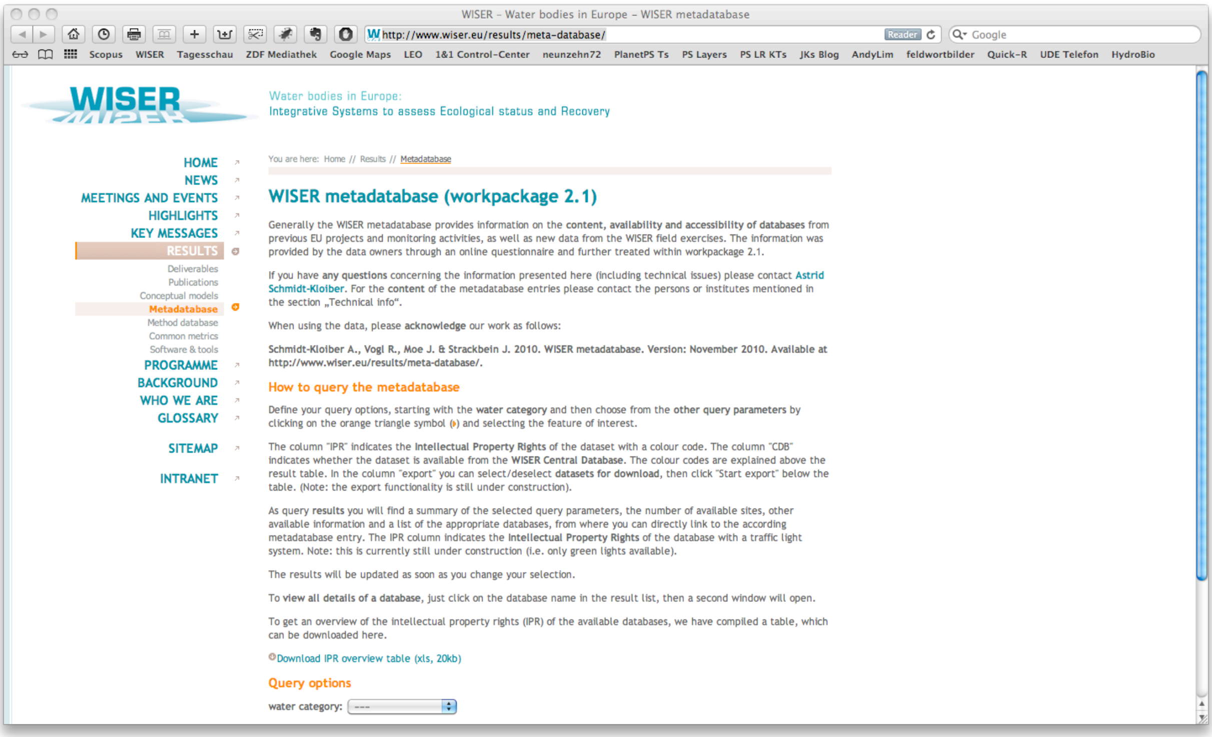
The WISER Methods Database

This WISER methods database contains information about the national assessment methods used to classify the ecological status of rivers, lakes, coastal and transitional waters in Europe. The Member States of the European Union apply these methods in their current monitoring programmes according to the EU Water Framework Directive.

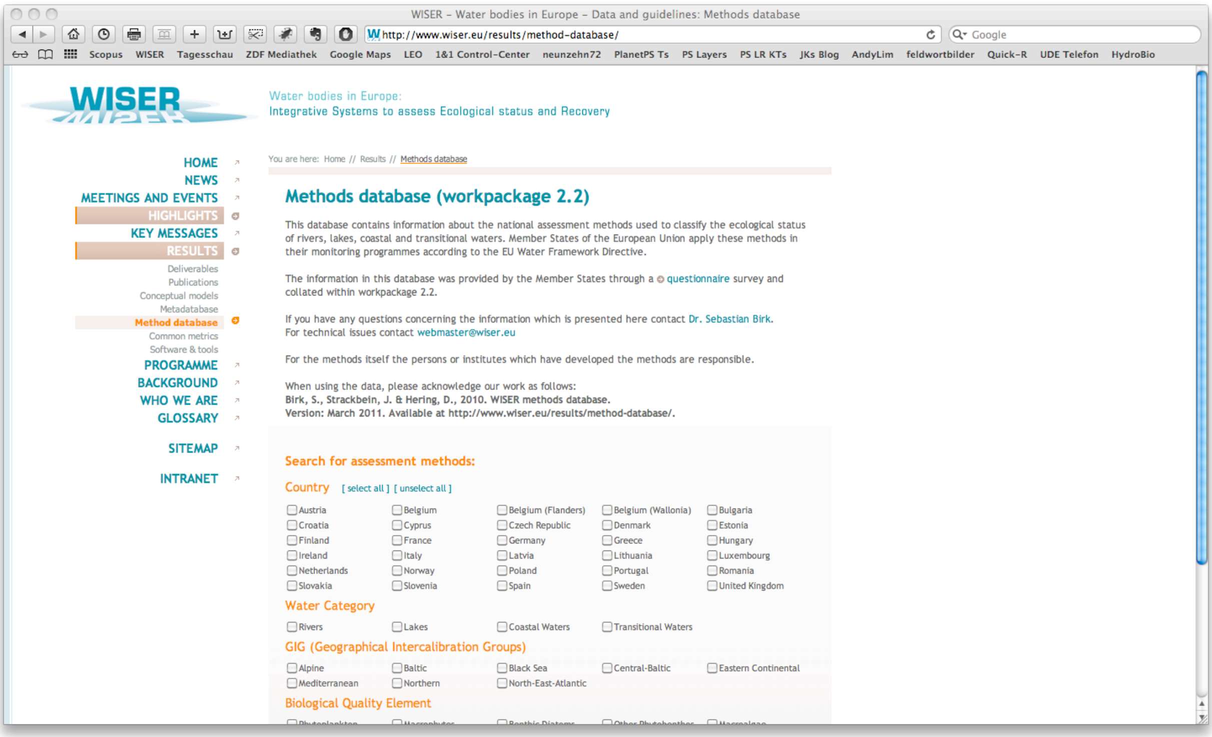
The information in this database was provided by the Member States through a questionnaire survey and collated within workpackage 2.2.

If you have any questions concerning the information presented in the WISER methods database. Please contact Dr. Sebastian Birk. (at UDE Essen; sebastian.birk@uni-due.de). For the methods itself, please contact the persons or institutes in charge of the development of the respective methods as indicated in the Methods Database, Section 1.13.

When using the data, please acknowledge our work as follows: Birk, S., Strackbein, J. & Hering, D., 2010. WISER methods database. Version: March 2011. Available at http://www.wiser.eu/results/method-database/.



Screenshot of the WISER meta database at http://www.wiser.eu/results/meta-database/.



Screenshot of the WISER methods database at http://www.wiser.eu/results/method-database/.

Overview of WISER Deliverables

WISER has produced 88 Deliverables altogether, most of which were edited either as reports (R) or as manuscripts (M) for submission to a peer-reviewed journal. These are listed below. Other Deliverables, such as data templates, or the project website www.wiser.eu and those with restricted access (e.g. due to confidential information) are not listed.

Module 2	Title	
D2.2-1	Online database on assessment methods for lakes, rivers, coastal and transitional waters in Europe	http://www.wiser.eu/download/D2.2-1.pdf http://www.wiser.eu/results/method-database/
D2.2-2	Guidance paper on indicator development to be used in Modules 3 and 4	http://www.wiser.eu/download/D2.2-2.pdf
Module 3		
D3.1-1	Report on phytoplankton composition metrics, including a common metric approach for use in intercalibration by all GIGs	http://www.wiser.eu/download/D3.1-1.pdf
D3.1-2	Report on harmonised metric for bloom frequency and intensity	http://www.wiser.eu/download/D3.1-2.pdf
D3.1-3	Report on uncertainty in phytoplankton	http://www.wiser.eu/download/D3.1-3.pdf

	metrics	
D3.1-4	Guidance document on sampling, analysis and counting standards	http://www.wiser.eu/download/D3.1-4.pdf
D3.2-1	Report on comparison and harmonisation of macrophyte survey methods including a relevant species list and a proposal of a common sampling protocol	http://www.wiser.eu/download/D3.2-1.pdf
D3.2-2	Report on uncertainty in macrophyte metrics	http://www.wiser.eu/download/D3.2-2.pdf
D3.2-3	Report on the most suitable lake macrophyte based assessment methods for impacts of eutrophication and water level fluctuations	http://www.wiser.eu/download/D3.2-3.pdf
D3.3-2	Manuscript on optimised sampling strategies and sources of uncertainty	http://www.wiser.eu/download/D3.3-2.pdf
D3.3-3	Report on assessment of European lakes using benthic invertebrates	http://www.wiser.eu/download/D3.3-3.pdf
D3.3-4	Manuscript on the assessment of ecological effects of hydromorphological lake shore alterations and water level fluctuations using benthic macroinvertebrates	http://www.wiser.eu/download/D3.3-4.pdf
D3.4-2	Manuscript on changes in size structure of fish in European lakes along eutrophication and hydromorphological pressure gradients	http://www.wiser.eu/download/D3.4-2.pdf
D3.4-3	Guidelines for standardisation of hydroacoustic methods	http://www.wiser.eu/download/D3.4-3.pdf
D3.4-4	Report on fish indicators for ecological status assessment of lakes affected by eutrophication and hydromorphological pressures	http://www.wiser.eu/download/D3.4-4.pdf
Module 4		
D4.1-1	Report on identification of type-specific phytoplankton	http://www.wiser.eu/download/D4.1-1.pdf

	assemblages for 2-3 ecoregions	
D4.1-2	Report on assessment of pigment data potential for multi-species and assemblage indices	http://www.wiser.eu/download/D4.1-2.pdf
D4.1-4	Manuscript on the review of multi-species indicators synthesised with WP results	http://www.wiser.eu/download/D4.1-4.pdf
D4.2-1	Review report/manuscript on seagrass indicator potential	http://www.wiser.eu/download/D4.2-1.pdf
D4.2-2	Report/manuscript on responses of macroalgae and seagrass indicators to drivers of deterioration	http://www.wiser.eu/download/D4.2-2.pdf
D4.2-4	Report/manuscript on benthic macroflora indicators for transitional waters, including classification boundaries, definition of reference conditions and uncertainty	http://www.wiser.eu/download/D4.2-4.pdf
D4.3-1	Manuscript on the responses of existing indicators to different pressures	http://www.wiser.eu/download/D4.3-1.pdf
D4.3-2	Manuscript on reference conditions for transitional waters	http://www.wiser.eu/download/D4.3-2.pdf
D4.3-4	Manuscript on indicators for hard bottom substrates	http://www.wiser.eu/download/D4.3-4.pdf
D4.4-1	Report reviewing existing multimetric approaches for fishes in transitional waters in Europe and elsewhere and the requirements and demands for harmonisation based on available datasets	http://www.wiser.eu/download/D4.4-1.pdf
D4.4-2	Report on testing the behaviour and sensitivity/uncertainty of the reviewed multimetrics on single and multiple datasets	http://www.wiser.eu/download/D4.4-2.pdf
D4.4-3	Report detailing multivariate analysis of fish data and metrics against pressures and impacts for different transitional waters	http://www.wiser.eu/download/D4.4-3.pdf

D4.4-4	Report summarising the definitions of reference conditions using predictive models for ecological endpoints for fish in transitional waters	http://www.wiser.eu/download/D4.4-4.pdf
D4.4-5	Final report indicating the potential for modelling approaches for fishes in transitional waters and the conclusions regarding harmonising suitable metrics and approaches for wider use	http://www.wiser.eu/download/D4.4-5.pdf
Module 5		
D5.1-1	Report on conceptual models on driver-pressure-impact-response-recovery chains for the impact of hydromorphological degradation and eutrophication on rivers (draft and final versions)	http://www.wiser.eu/download/D5.1-1.pdf
D5.1-2	Manuscript on the application of statistical models to predict the response of river BQEs to pressure reduction (hydromorphology and eutrophication) under different climate scenarios	http://www.wiser.eu/download/D5.1-2.pdf
D5.1-3	Manuscript on the application of mechanistic models to predict the response of river BQEs to pressure reduction (hydromorphology and eutrophication) under different climate scenarios	http://www.wiser.eu/download/D5.1-3.pdf
D5.1-4	Guidance on management options and measures of pressure reduction to improve the ecological status of rivers with emphasis on the implications of global/climate change	http://www.wiser.eu/download/D5.1-4.pdf
D5.2-1	Analysis of applied modelling approaches in the case studies - technical report	http://www.wiser.eu/download/D5.2-1.pdf

D5.2-2	Report on using BQEs as indicators for reducing pressures	http://www.wiser.eu/download/D5.2-2.pdf
D5.2-3	Guidelines on the use of different modelling approaches for designing Programme of Measures	http://www.wiser.eu/download/D5.2-3.pdf
D5.2-5	Report on the effects of climate change on reference conditions and ecological status in lakes	http://www.wiser.eu/download/D5.2-5.pdf
D5.2-6	Synthesis paper on options for lake management to improve ecological status	http://www.wiser.eu/download/D5.2-6.pdf
D5.3-1	Report/manuscript on temperature effects on hypoxia and benthic fauna	http://www.wiser.eu/download/D5.3-1.pdf
D5.3-2	Report/manuscript on shifting reference conditions and boundaries for BQE indicators	http://www.wiser.eu/download/D5.3-2.pdf
D5.3-3	Comparison of mechanistic and statistical modelling approaches for catchment management	http://www.wiser.eu/download/D5.3-3.pdf
D5.3-4	Synthesis report/manuscript on nutrient input reductions and uncertainties	http://www.wiser.eu/download/D5.3-4.pdf
D5.3-5	Guidelines for the use of coastal management models	http://www.wiser.eu/download/D5.3-5.pdf
Module 6		
D6.1-1	Report on a workshop to bring together experts experienced with tool development and uncertainty estimation	http://www.wiser.eu/download/D6.1-1.pdf
D6.1-2	Manuscript reviewing components of uncertainty and their assessment, including guidelines for estimation and quality assurance	http://www.wiser.eu/download/D6.1-2.pdf
D6.1-3	Generally applicable software tool for assessing confidence of status class	http://www.wiser.eu/download/D6.1-3.pdf http://www.wiser.eu/results/software/

D6.2-1	Review of approaches for combining BQEs in WFD assessment	http://www.wiser.eu/download/D6.2-1.pdf
D6.2-3	Report from the end user-workshop	http://www.wiser.eu/download/D6.2-3.pdf
D6.2-4	Report with recommendations for WFD monitoring and assessment	http://www.wiser.eu/download/D6.2-4.pdf
D6.3-1	Report from workshop on among BQEs, habitats and systems comparisons	http://www.wiser.eu/download/D6.3-1.pdf
D6.3-2	Report and manuscript on the use of BQEs, habitats and ecosystems for detecting human-induced change	http://www.wiser.eu/download/D6.3-2.pdf
D6.4-1	Literature report on biological processes on catchment scale, such as connectivity, dispersal and metapopulation dynamics	http://www.wiser.eu/download/D6.4-1.pdf
D6.4-2	Report on the differences between cause-effect-recovery chains of different drivers	http://www.wiser.eu/download/D6.4-2.pdf
D6.4-3	Final report on impact of catchment scale processes and climate change on cause-effect and recovery-chains	http://www.wiser.eu/download/D6.4-3.pdf
Module 7		
D7.2-4	Book of abstracts	http://www.wiser.eu/meetings-and-events/final-conference/abstracts/
D7.2-6	End user summary and booklet	http://www.wiser.eu/download/D7.2-6.pdf

Overview of WISER publications

More than 100 WISER publications in peer-reviewed journals have been produced during the lifetime of WISER. The list provided below, however, cannot be complete, since WISER partners shall continue to publish their results after the lifetime of WISER. Thus, we recommend visiting the WISER website for up-to-date information on publications and other releases: http://www.wiser.eu/results/publications/. Furthermore, a WISER Special Issue is currently being produced in cooperation with the Journal Hydrobiologia. A release of the special issue is planned for late 2012/early 2013. The special issue shall summarise the major WISER outcome and shall contain more than 35 articles on all aspects of assessment and management of surface waters in Europe.

List of WISER publications (status: January 2012):

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Basset, A. 2010. Aquatic science and the water framework directive: a still open challenge towards ecogovernance of aquatic ecosystems. Aquatic Conservation. Marine and Freshwater Ecosystems, 20, 245–249.(www.interscience.wiley.com). DOI: 10.1002/aqc.1117

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Buchaca, T., Skov, T., Amsinck, S., Gonçalves, V., Azevedo, J.M.N. & Jeppesen, E. (2011) Rapid ecological shift following piscivorous fish introduction to increasingly eutrophic Lake Furnas (Azores Archipelago, Portugal): a paleoecological approach.- Ecosystems (in press).

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