Current questions in water management

Book of abstracts to the WISER final conference
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Edited by Astrid Schmidt-Kloiber, Anne Hartmann, Jörg Strackbein, Christian K. Feld & Daniel Hering

WISER – Water bodies in Europe

Integrative Systems to assess Ecological status and Recovery

Funded by the European Union under the 7th Framework Programme,
Theme 6 (Environment including Climate Change), contract No. 226273

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Coordinator:
Prof. Dr. Daniel Hering
Faculty of Biology, Applied Zoology/Hydrobiology
University of Duisburg-Essen
D-45117 Essen, Germany

Layout and compilation:
Anne Hartmann & Astrid Schmidt-Kloiber, University of Natural Resources and Life Sciences Vienna, Austria
Jörg Strackbein, Faculty of Biology, Applied Zoology/Hydrobiology, University of Duisburg-Essen, Germany

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Content

Precision and behaviour of fish-based ecological quality metrics in relation to natural and anthropogenic pressure gradients in European estuaries

María C. Alvarez[1], Angel Borja[2], Anne Courrat[3], Mike Elliott[1], Anita Franco[1], Mario Lepage[3], Joao M. Neto[4], Rafael Pérez-Domínguez[1]*, Violin Raykov[5] and Ainhize Uriarte[2]

[1] Institute of Estuarine and Coastal Studies (IECS) University of Hull. Hull, HU6 7RX, United Kingdom
[2] AZTI - Tecnalia / Marine Research Division. Herrera kaia portualdea z/g. 20110 Pasaia, Spain
[3] Irstea, Groupement de Bordeaux, Unité EPBX, Ecosystèmes estuariens et Poissons migrateurs amphihalins. 50 av. de Verdun, Gazinet - 33612 CESTAS Cedex, France
[4] IMAR - Institute of Marine Research (C.M.A.). Department of Life Sciences, Faculty of Science and Technology, University of Coimbra 3004 - 517 Coimbra, Portugal
[5] Institute of Oceanology, Bulgarian Academy of Sciences (IOBAS), Parvi Mai str.40, P.O.Box 152, Varna 9000, Bulgaria Environment Agency, Kings Meadow Road, Reading, Berkshire, RG1 8DQ, UK

* Email address of corresponding author: r.perez@hull.ac.uk

Key words: fish, BQE, ecological assessment, ecological integrity, transitional waters, estuaries, multimetric index, mixed models, reference, uncertainty, human pressures

Introduction

Historical human habitation and resource use in coastal areas have resulted in substantial change of most European estuaries from their original condition. Current paradigms consider this change to be detrimental and deviations from the original condition a measure of habitat degradation. Estuarine systems are important for ecological functioning in providing ecosystem services which in turn deliver societal benefits (i.e. flood defence, fisheries, water purification, energy). Human pressure on estuarine and coastal areas is likely to continue increasing and together with global changes, such as sea-level rise, create conflicts between the public, stakeholders and managers. In response to future management needs and to prevent further degradation, the European Union has enacted specific legislation, the Water Framework Directive (WFD), which aims "to establish a framework for the protection of inland surface waters, transitional waters , coastal waters and groundwater" (Article 1, 2000/60/EC). Fish are biological descriptors of quality promoted in this European Directive to indicate estuarine quality status. Fish have a high relevance to the public and managers as there is an intuitive value associated to them. Therefore, fish are often better placed to raise public awareness than for example water chemistry or benthic invertebrates. Despite this important practical advantage there are technical difficulties for their incorporation into tools to define estuarine ecological quality. Some of these difficulties also apply to other biological elements in their link with natural fluctuation of estuarine systems but others are specific to fish such as their mobility, large recruitment variability or gear avoidance bias.

The work conducted within the WISER fish transitional waters work package (WP4.4) has analysed key issues necessary to develop practical estuarine fish tools and promote evidence-based discussion of the underlying ecosystem functioning affected by anthropogenic alterations. The specific objectives set were: 1) to review the state of the art in estuarine fish indices and test and compare available indices using common methods through a dedicated field exercise; 2) to evaluate the uncertainty associated with fish-based assessments; and 3) to propose and where possible test new approaches for the modelling of fish communities to be used in defining reference conditions.

Experimental Approach

Review of estuarine fish indices and WISER field exercise

We reviewed seventeen published fish-based indices of estuarine quality worldwide. The comparison was done on common development strategies and assessment methods using cluster analysis and Similarity percentage (SIMPER) analysis. In order to assess the intercomparability between tools, a standardized field sampling campaign was conducted at 8 different transitional sites across Europe. This dataset includes data from a dedicated WISER transitional fish field survey (4 sites) and additional Basque Water Agency (Spain)

(3 sites) and Environment Agency (UK) data (1 site) gathered with comparable sampling protocols. On this extended dataset, fish ecological and feeding guilds, as well as sensitive and reference species lists, were harmonized across sites according to lifecycle theory and bibliographical sources. Finally Ecological Quality Ratios were calculated with six different fish indices. Analyses were conducted on comparable gears (fyke nets, beam trawls and seine nets) and standardised to salinity groups. To assess score agreement between indices, Kendall rank correlation coefficients and Kappa values (Fleiss-Cohen weights) between pairs of indices were used. Finally Corine Land Cover (CLC) pressure proxies and expert opinion were used to test the sensitivity of the index values to anthropogenic pressure. The sensitivity of fish indices to pressure gradients was further assessed using the AZTI's Fish Index (AFI) on a sample of 12 estuaries. Initially 16 pressure proxies and 8 estuarine descriptors were selected. After elimination of highly correlated variables, ordination analysis (Principal Component Analysis) was used to explore the general gradients segregating the sample estuaries. Finally multiple regression analysis between the more influential pressure proxies and AFI scores was used to assess sensitivity of the index.

Uncertainty associated to assessment metrics

A sample of seven common fish metrics identified in the initial index review was selected for analysis. Metric values were calculated using the proprietary WISER transitional fish data set and additional Basque Water Agency (Spain), IMAR-CMA (Portugal), French Water Agencies (France), Environment Agency (UK) data gathered with comparable sampling protocols. The datasets were organized in a relational database containing a total of 3249 fishing events which include fish data, sampling effort and water quality parameters. Additional physicochemical data on estuaries (such as estuarine area, continental shelf width, salinity regime, etc.) and CLC-derived pressure proxies were also used. A conceptual matrix was used next to identify a priori the more relevant sources of variability and define expected effect on the fish metric outcomes. Most relevant variance sources were then quantified using either linear mixed models (LMM) or generalized linear mixed models (GLMMs). Finally, the impact of variability at the fish metrics scale on the overall uncertainty of a multimetric index was investigated using the French multimetric Estuary and Lagoon Fish Index (ELFI). This final analysis was made with WISERBUGS (WISER Bioassessment Uncertainty Guidance Software). In addition to the confidence in the final assessment, we investigated the effect of extreme metric values on the outcome of the ELFI and the UK Transitional Fish Classification Index (TFCI). This sensitivity analysis was done by setting metric values at the extreme top and bottom percentiles of the metric distribution and then calculating the percentage change in the final index value with respect to the original value.

Reference conditions

The WISER extended dataset together with pressure proxies and physicochemical data introduced above was used to model the reference conditions. Percentage cover of agricultural, industrial, urban, and natural areas calculated at three different CLC buffer distances (1.0, 1.5 and 2.0 km from the shoreline) were used as pressure proxies. Ordination analysis (PCA) was then used to identify the subset of pressure proxies that best describe the global pressure gradient and eliminate redundant variables. LMM and GLMM was then use to model the response of the fish metrics along the pressure gradient. Finally the expected metric score at the level of the sites with lowest expected pressure or at the theoretical zero pressure was used to define the expected reference for the metric. The analysis was conducted independently for estuaries and lagoons, and for a range of sampling design (salinity class, gear, and sampling season) combinations.

Results and Discussion

Review of current estuarine fish indices and WISER field exercise

There are no globally-applicable estuarine fish indices but rather locally-applicable indices. Recent indices pay increased consideration to sensitivity to pressure and reference conditions, although expert opinion is still widely used. Assessments are more commonly made on structural attributes (i.e. diversity), followed by functional attributes (habitat use, feeding and nursery guilds) and fish condition. Current developments of the indices reviewed here have focused on improving the robustness and consistency of the assessments.

The WISER transitional fish field survey was completed in 2009. Sixty four fish species were recorded across 133 sampling events. Even using this harmonized dataset we found low agreement between index scores. In general indices are greatly dependent on the sampling methodology, particularly gear and effort. Some indices could not be calculated for all sites due to incompatibility of sampling methodology; furthermore, diversity-based metrics result in random/low quality scores outside the development area due to incompatible reference. The use of harmonized guilds allowed for a wider geographical applicability, but since site-specific reference was not always available any interpretation of the scores is difficult and could not be considered reliable or necessarily indicate a quality status. However, using a matching combination of fish index, reference values, and local dataset we demonstrate that multimetric fish indices could be sensitive to pressure gradients. This was clearly shown in the AFI case study where water pollution and global pressure index were significantly correlated with AFI scores. This indicates that fish indices can be used in biomonitoring only if there is compatibility between index structure, sampling method, reference condition and geographical area.

Uncertainty associated to assessment metrics

The analysis produced a general overview of technical (i.e. those linked to the method of assessment) and natural sources of variability that affect the reliability of fish-based multimetric indices. Some sources of variability, originally identified as important (i.e. effort, year to year or operator), could not be assessed due to the lack of data or could only be calculated on small subsets of data. Moreover, the estimation of interactions between sources of variability was often not possible. When data were available and models could be derived, it was clear that the variability of metric scores highly depends on the fish metrics considered, the way they are calculated and water body typology. General requirements for minimizing uncertainty are proposed. The impact of fish metric uncertainty on the uncertainty of multimetric indices is discussed and the uncertainty assessment of the French fish index ELFI is given as example. We then tested several metric combination rules to assess how they affect accuracy of the indices and we then provide recommendations to minimize uncertainty at the scale of the multimetric index. Finally the behaviour of multimetric indices under extreme manipulations of metric scores clearly indicated that metric type and number of metrics used are important in the index response with indices including more metrics or metrics with skewed distribution being less sensitive to extreme metric manipulations. It is still not clear whether this indicates an increased robustness of the indices or a lack of sensitivity. Comparison with pressure proxies are needed to make further interpretations.

Reference conditions

The CLC pressure proxies were very similar across the three different buffers hence only the 2.0 km buffer was kept in the analysis. It was relatively easy to separate sampling related factors when modelling reference conditions but extremely difficult to separate natural and anthropogenic factors. Ideally the reference value could be set independently from pressure proxies if there is sufficiently good information of species responses across natural environmental gradients at pristine sites. However, these are not generally available for transitional waters. Instead pressure-fish response was modelled together to forecast the expected reference at zero level of pressure. However, as there is no pristine state, results obtained this way may be inaccurate as they require an extrapolation outside the limits of the models. A compromise is to set the reference to the level of the least impacted sites. This increases accuracy but produced a reference condition set at an artificially diminished quality status which may be far from the true reference condition. All factors tested, from water body typology to gear choice, influenced the modelled reference values for all fish metrics highlighting the combined importance of natural factors and sampling methodologies. Reference state can change greatly over relatively short geographical distances or between seasons and gear- and habitat-related effects are generally very influential in defining the reference value. These effects highlight the importance of standardizing any monitoring programme to ensure that proper quality evaluation can be made using fish in estuaries.

Sources of uncertainty in estimation of eelgrass depth limits

Thorsten J. S. Balsby, Dorte Krause-Jensen & Jacob Carstensen

Department of Biosciences, University of Aarhus, Denmark

Email address of corresponding author: thba@dmu.dk

Introduction

Monitoring of ecological status of coastal ecosystems is important for management and assessment of compliance with the Water Framework Directive. To design an optimal monitoring program, we need information on the different sources of variability associated with the assessment. Seagrasses are key sentinels of ecological status in the marine environment (Marba et al. in prep.). A wide range of seagrass indicators are used in Europe, the three most common being shoot density, cover and depth limit (Marba et al. in prep.). The rationale behind using seagrass observations to derive indicators is that seagrasses are sensitive to e.g. changes in eutrophication pressure. The depth limit of eelgrass and other seagrasses is mainly determined by water clarity, most commonly measured as Secchi depth (Duarte et al 2007, Ralph et al 2007, Krause- Jensen et al. 2011), which is affected by the degree of eutrophication (e.g. Cloern 2001). Contrary to measurements of Secchi depth, which provide snapshots of a highly dynamical variable, the eelgrass depth limit constitutes an integral indicator of the light conditions at a site on a much longer time span, since eelgrass is perennial. Nutrient load and Secchi depth in the marine environment are strongly and negatively correlated, as high nutrient loads result in high growth rates of pelagic algae, and hence concentrations, in the upper part of the water column, thereby reducing the amount of light to the seafloor (Nielsen et al. 2002).

Eelgrass depth limits may vary substantially within an estuary or coastal area. Some of this variation is due to natural causes such as wind exposure (fetch), ice, herbivory, or seasonal variation. Other parts of the variation are associated with the way monitoring has been conducted. The major sources of variation in relation to depth limit estimation are spatial at the larger scale (variation between transects in a particular area), spatial at the smaller scale (variation between replicates for each transect), methodological (individual differences between divers in assessment of depth limits), and temporal (variation between years). Quantification of these important sources of variability can devise means to reduce uncertainties related to the monitoring methods and program, and thus improve the precision of the indicator.

A large-scale and long-term monitoring data set on the maximum depth limit of eelgrass in Danish coastal waters allowed us to estimate how much each of the factors associated with the monitoring scheme contributed to the uncertainty in the estimation of the maximum depth limit of eelgrass.

Methods

The monitoring of eelgrass maximum depth limit analyzed in the present study has been conducted as part of the national environmental monitoring program between1993 and 2009. Over this period monitoring was conducted by 29 different divers at a total of 347 transects, distributed over 65 sites. All monitoring efforts were conducted between April and September. At each transect a diver estimated the maximum depth limit of eelgrass defined as the deepest occurring shoot. The maximum depth limit was determined by a diver swimming along the transect from the coast to deeper depths. When reaching the depth where eelgrass no longer grew the diver would swim 30 m orthogonal to each side of the transect and record replicates of the maximum depth in order to achieve a better estimate of the maximum depth limit at the transect. At the location of the estimated maximum depth the diver put the depth measurement unit to the bottom to read the maximum depth limit for the eelgrass. Usually the same transects would be monitored for multiple years, although this has not been consistent throughout the entire data set.

Statistical approach

We quantified the uncertainty associated with estimating the maximum depth limit, which the factors diver identity, transect identity and year as well as replicates per transects could account for. We used a mixed model for the maximum depth limit with each site treated as a fixed variable. By including transect, diver identity and year as random effects in the model we estimated their variances. For some sites the mixed model gave a non-positive hessian matrix indicating that the model fitted the data poorly. For other sites the mixed model could not converge as one or more of the random factors only had one level and therefore the variance could not be determined. Subsequently, we removed the factor(s) for the sites where the variance could not be assessed. This produced up to 59 estimates for the variances of the four random factors.

We investigated whether the standard error, calculated as the square root of the different variance components, depended on the maximum depth limit by fitting the estimated maximum depth limit for each site to the standard error. For all four random factors the standard error increased at low estimated maximum depth limits and seemed to become constant at larger depth limits. We therefore used either a Gaussian or spherical model to fit the data. On basis of the variance estimates we then conducted a power analysis varying number of divers, transects and years. We used PROC MIXED and PROC MODEL in SAS 9.2 (Cary, NC) for conducting all analyses.

Results

Sources of variation in general

The maximum depth limit for all transects varied between 0.2 m and 12.5 m. The maximum depth limit differed significantly between sites (mixed model $F_{65, 7228}$=34.24, p<0.001). The variance estimates analysis also showed that the standard error of the variance estimates differed substantially between the random factors (i.e. diver identity, year, and transects) (Tab. 1).

Transect, year and diver identity accounted for much more of the variation than the residual or replication variation. The estimates for the variance parameter estimates for year, transect and diver identity were in the same magnitude larger as the replicates indicating that the design of the survey. These estimates of variance suggested that the design of the survey in terms of year,

Table 1: Variance parameter estimates ± SE and p-values for the random effects in the model on the full data set.

Variance component	Estimate (± SE)	p
Transect	*0.769 (±0.068)*	*<0.0001*
Diver identity	*0.165 (±0.070)*	*0.009*
Year	*0.212 (±0.114)*	*0.031*
Replicate	*0.366(±0.006)*	*<0.0001*

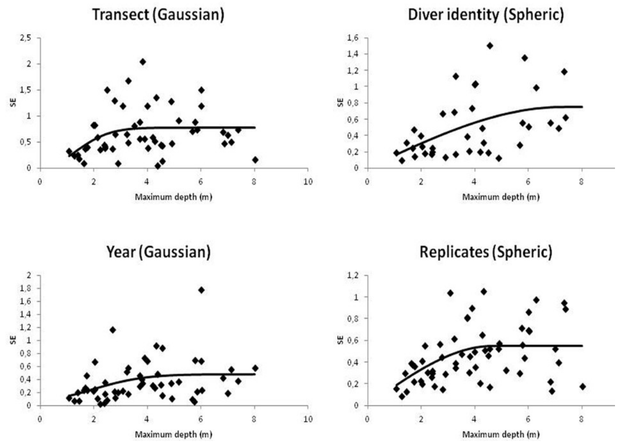

Figure 1: The standard error as a function of the estimated maximum depth limit for eelgrass for Transect; Diver identity; Year; and Replicate. Lines show the fitted spherical or Gaussian functions.

transect and diver identity would be important to consider in attempts to reduce the uncertainty of estimates of maximum depth limit.

The standard error increase with the estimated maximum depth limit

Normally, variances are assumed constant across the range, but for transect, diver identity, year and replicate the variance overall increased with the estimated maximum depth limit (Fig. 1).We attempted to describe how the standard errors increased with the maximum depth limit by fitting appropriate functions to the data. For all factors it seemed that the standard error initially increased with the estimated maximum depth limit and then levelled off. We therefore attempted to fit exponential, Gaussian and spherical functions to the curves for transect, diver identity, year and replicate standard errors, and selected the function that resulted in the best fit (Tab. 2). The replicates, which quantified the random variation unaccounted for by the model, were best approximated by a spherical function (R^2 =0.21). The range estimate for replicate variability indicated that the standard error increased with the maximum depth until 4.65 m and subsequently stabilized at SE of 0.54 (Fig. 1).

Some of the uncertainty in estimates of the maximum depth limit was caused by different divers and for diver identity the standard error also increased with the maximum depth estimates at different depths. A spherical function gave the best fit (R^2=0.23) to the observed variation in diver identity as a function of depth limit. For diver identity the range estimate suggested that the uncertainty of the maximum depth estimate (i.e.

standard error) increased until 7.18 m maximum depth and the stabilized around a standard error of 0.75 (Fig. 1).Gaussian models gave the best fits for the variability among transects (R^2=0.13) and among years (R^2=0.11) as functions of the depth limit. The variability between transects increased until 1.82 m and stabilized at a standard error of 0.77. Regarding the variability between years the range suggested that the standard error increased until reaching a maximum depth limit of 2.32 m, after which it stabilized at standard error of 0.47 (Fig. 1).

Discussion

Our study shows that the uncertainty of an estimated maximum depth limit for eelgrass generally increases with the depth limit. We have quantified how the uncertainty of the maximum depth limit estimation is affected by the variability between transects, years, and divers, which all can be taken into account when designing monitoring programs for the eelgrass maximum depth. Knowledge of how these factors contribute to the uncertainties will be used in power analyses to devise monitoring designs that minimize the overall uncertainty of maximum depth limit estimates.

We will also analyze the sources of uncertainty in estimating the depth limit for another important seagrass, Posidonia oceanica, obtained from Spanish monitoring in the Mediterranean Sea.

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Table 2: Model fits for the standard error of the four random components. Range indicated when the sill (Threshold value) is reached which is when the standard error ceased to increase with the estimated maximum depth limit. Sill estimated the threshold value for the function. Nugget indicated intercept with the y-axis. We used range=4 and sill=0.5 as start values for the estimation procedure.

Variance component	Function	R^2	Parameter estimates		
			Range (m)	Sill (m)	Nugget (m)
Transect	*Gaussian*	*0.13*	*1.82*	*0.77*	*0*
Diver identity	*Spheric*	*0.23*	*7.26*	*0.75*	*0*
Year	*Gaussian*	*0.11*	*2.73*	*0.39*	*0.09*
Replicate	*Spheric*	*0.21*	*4.65*	*0.55*	*0*

Implementing the Water Framework Directive in transitional and coastal water ecosystems using a new size spectra sensitivity index (ISS).

Alberto Basset[1], Enrico Barbone[2], Nicola Fiore[1], Maurizio Pinna[1], Federica Lugoli[1], Sofia Reizopoulou[3], Ilaria Rosati[1], Maria Rosaria Vadrucci[2]

[1] Department of Biological and Environmental Sciences and Technologies, University of the Salento 73100, Lecce (Italy)
[2] ARPA PUGLIA – Puglia Regional Environmental Protection Agency. Corso Trieste 27, 70100, Bari (Italy)
[3] HCMR, Institute of Oceanography, 1903 Anavyssos, Greece

Email address of corresponding author: alberto.basset@unisalento.it

Key words: ecological indicators, benthic macroinvertebrates, phytoplankton, size-spectra, transitional and coastal waters, Water Framework Directive

Introduction

Aquatic ecosystem health and ecological status (sensu Water Framework Directive, 2000/60/EC, 2000; hereafter WFD) are properties that can be measured indirectly from structural components of the ecological communities (Basset et al., 2012). The WFD cites benthic macroinvertebrates and phytoplankton as biological quality elements essential to be used in the ecological status classification of transitional and coastal waters. For this classification, a large number of benthic taxonomic indices have been proposed, ranging from simple community diversity indices to more complex indices (Pinto et al., 2009). Regarding phytoplankton, most indices based on community diversity have been developed for marine coastal waters (Spatharis and Tsirtsis, 2010) and only a few for transitional waters (Facca et al., 2009).

Taxonomic-free assessment tools based on functional traits (Mouillot et al., 2006), including biomass and body size, have also been developed. These offer an alternative or complementary perspective to taxonomic analysis for both benthic and phytoplankton communities. Functional traits include individual body size, which is known to affect individual metabolism, life cycle parameters, population growth and interactions (Basset et al., 2012), functionally linking size spectra to ecosystem properties such as vigour, organisation and resilience (Basset, 2010). Size spectra are thus particularly suitable structural community components for translating ecological status into measurable metrics (Basset et al., 2012).

Based on these evidences, a new multimetric size spectra sensitivity index (hereafter, ISS) has recently been developed to assess the ecological status in transitional waters by describing the sensitivity of size classes respect to anthropogenic disturbance. The ISS index was initially developed and validated using data on benthic macroinvertebrates (Basset et al., 2012) and subsequently also using data on phytoplankton (Vadrucci et al., submitted) of Mediterranean and Black Sea lagoons. Moreover, the new index was also tested in marine waters on phytoplankton data (Lugoli et al., submitted).

Here, we show the ability of the ISS based on benthic macroinvertebrates and phytoplancton, to: (i) discriminate between disturbed and undisturbed conditions in transitional and in coastal water ecosystems; and (ii) illustrate the dose–response relationships with respect to specific stress gradients.

Materials and Methods

Study sites

The ISS was developed using benthic macroinvertebrates and phytoplankton data originally collected from 12 and 14 Mediterranean and Black Sea lagoons respectively (Figure 1a), in the context of the European TWReferenceNet project. They included micro and non-tidal lagoons, saltpans and oligohaline coastal wetlands, which differed in terms of their physiographic, hydrological and chemical-physical characteristics and the degree of anthropogenic disturbance (Barbone et al. 2012; Vadrucci et al., submitted). The selected lagoons 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 (Table 1).

The new proposed index was then validated with reference to an independent set of benthic macroinvertebrate and phytoplankton data from two lagoons

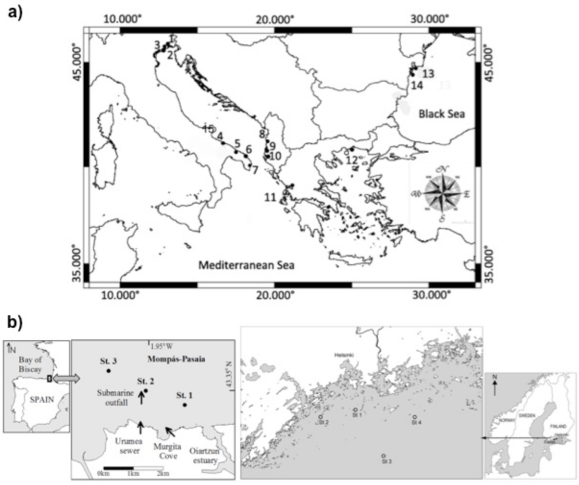

Figure 1: Study sites: a) Mediterranean and Black Sea Lagoons: 1. Grado Marano Lagoon, 2. Grado 'Valle Cavanata', 3. Grado fish farm, 4. Margherita di Savoia Salt pan, 5. Torre Guaceto brackish Wetland, 6. Le Cesine brackish Wetland;, 7. Lake Alimini, 8. Patok Lagoon, 9. Karavasta Lagoon, 10. Narta Lagoon 11. Logarou Lagoon, 12. Agiasma Lagoon, 13. Sinoe Lagoon, 14. Leahova Lagoon, 15. Lesina Lagoon; b) Costal water ecosystems: Helsinki marine area and Mompás-Pasaia coastal area.

characterised by a very strong abiotic stress gradients. These were the Margherita di Savoia saltpan for benthic macroinvertebrates and Lesina lagoon for phytoplankton, the former characterised by a salinity gradient and the latter by a eutrophication gradient (Basset et al., 2012; Vadrucci et al., submitted).

The adequacy of the new index was also tested on phytoplankton data collected during the WISER project in two coastal water ecosystems, Helsinki marine area (Finland) and Mompás-Pasaia coastal area (Spain) (Lugoli et al., submitted), selected because of their eutrophication gradients (Figure 1b; Table 2).

Description of the new size spectra sensitivity index

The ISS was computed in accordance with the formulas:

$$\mathrm{ISS_{benthos}} = \sum p(\mathrm{CL}_i) * \omega_i * \mathrm{s}$$

$$\mathrm{ISS_{phytoplankton}} = \sum p(\mathrm{CL}_i) * \omega_i * \mathrm{s} * b$$

where $p(\mathrm{CL}_i)$ is the proportion of individuals in the *i*th size class ω_i is the assigned sensitivity value for the *i*th size class; *s* is a discrete correction factor for the number of taxa and *b* is a correction factor for phytoplankton biomass (Barbone et al., 2012; Vadrucci et al., submitted).

For the ISS calculation the macroinvertebrate and phytoplankton size spectra were divided into 6 classes (CL1–CL6) by clustering the original abundance octaves into groups of three. We used this approach in order to achieve a large enough size ratio between neighbouring size classes (8:1) to be able to assign each

Table 1: List of the 15 transitional water ecosystems considered with a qualitative evaluation of protection measures and pressures. Protection measures are classified as: A = Ramsar site; B = Nature 2000 site; C = Local protection plans; X indicates the presence of protection measures. Pressures reported are: 1 = organic and nutrient loading; 2 = acidification; 3 = hazardous substances; 4 = fishing; 5 = alien species; 6 = navigation; 7 = physical and other modifications. The intensity of each pressure type was evaluated using a scale of value ranging from 0 to 4 (0 = absent; 1 = very low; 2 = low; 3 = moderate; 4 = high) in accordance with expert evaluation based on existing knowledge as reported in Barbone et al., 2012.

Country	Coordinates (Lat., Long.)	Lagoon	Protected sites	Protection			Environmental and anthropogenic pressures						
				A	B	C	1	2	3	4	5	6	7
Italy	*45.742° 13.221°*	*Grado Marano*	*Not*				*4*	*0*	*4*	*0*	*2*	*3*	*2*
Italy	*45.712° 13.470°*	*Grado Valle Cavanata*	*Yes*	*X*	*X*		*2*	*0*	*1*	*0*	*1*	*0*	*1*
Italy	*45.722° 13.362°*	*Grado fish farm*	*Not*				*2*	*0*	*1*	*4*	*4*	*0*	*2*
Italy	*41.630° 15.290°*	*Lesina*	*Not*				*2*	*0*	*1*	*3*	*0*	*0*	*2*
Italy	*41.429° 15.988°*	*Margherita di Savoia*	*Yes*	*X*			*2*	*2*	*0*	*1*	*0*	*0*	*4*
Italy	*40.711° 17.795°*	*Torre Guaceto*	*Yes*	*X*			*1*	*0*	*1*	*0*	*0*	*0*	*1*
Italy	*40.358° 18.335°*	*Cesine*	*Yes*	*X*	*X*		*1*	*0*	*0*	*0*	*0*	*0*	*2*
Italy	*40.202° 18.446°*	*Alimini*	*Yes*		*X*	*X*	*2*	*0*	*0*	*3*	*1*	*0*	*1*
Albania	*41.635° 19.590°*	*Patok*	*Yes*		*X*	*X*	*0*	*0*	*0*	*2*	*1*	*0*	*1*
Albania	*40.920° 19.472°*	*Karavasta*	*Yes*	*X*			*2*	*1*	*0*	*4*	*1*	*1*	*3*
Albania	*40.529° 18.426°*	*Narta*	*Yes*			*X*	*2*	*2*	*0*	*3*	*0*	*0*	*4*
Bulgaria	*43.199° 27.794°*	*Varna*	*Not*				*3*	*1*	*3*	*1*	*2*	*4*	*3*
Greece	*39.062° 20.900°*	*Logarou*	*Not*		*X*		*3*	*0*	*1*	*4*	*0*	*0*	*3*
Romania	*44.620° 28.888°*	*Sinoe*	*Yes*	*X*			*2*	*0*	*0*	*3*	*0*	*0*	*3*
Romania	*44.727° 29.028°*	*Lehaova*	*Yes*	*X*			*2*	*0*	*0*	*1*	*1*	*0*	*1*

Table 2: Pressures at each sampling station in two marine ecosystems considered (see Figure 1b). Intensity of each pressure type was evaluated using a discrete scale ranging from 1 to 3 (1 = low; 2 = moderate; 3 = high) in accordance with an expert evaluation based on existing knowledge as reported in Lugoli et al., submitted.

		Type of pressures	Non-point pollution sources			Point pollution sources		Habitat loss	Ports			Fisheries	
		Partial Pressures	Background from the open sea	Diffuse agricultural inputs	Freshwater inputs	Domestic discharges	Industrial discharges	Reclaimed land	Port activity	Navigation	Dredging	Fin-Fisheries	Shell-fisheries
Country	System	Stations											
Finland	*Helsinki marine area*	*1*	*3*			*2*				*1*			
		2	*3*			*2*				*1*			
		3	*3*			*1*				*1*			
		4	*3*			*1*				*1*			
Spain	*Mompás-Pasaia*	*1*			*1*	*1*	*1*			*2*		*1*	
		2			*2*	*3*	*2*			*1*		*1*	
		3								*1*		*1*	

class a different value for sensitivity to anthropogenic disturbance. Detailed information on description of the new index is reported elsewhere (Basset et al., 2012; Vadrucci et al., submitted).

Data analysis

The robustness of the ISS was tested to two criteria: (i) its ability to discriminate between groups of disturbed and disturbed sites within transitional and costal water ecosystems; and (ii) the dose-response relationships along the considered environmental stress gradients.

To this aim, the Wilcoxon rank test was used to evaluate the differences in the ISS values between "disturbed and "undisturbed" lagoons, and regression analysis was performed to test the variation of the ISS index with environmental gradients (Barbone et al., 2012; Vadrucci et al., submitted).

Results

The size spectra index showed high discrimination between disturbed and undisturbed sites using both benthic macroinvertebrate data and phytoplankton data. The ISS presented significantly higher values at undisturbed than disturbed sites in transitional and costal waters (Figure 2 a, b).

The ISS was validated using both data sets from which the index was derived and a completely independent data set based on the Margherita di Savoia saltpan for benthos and on Lesina lagoon for phytoplankton.

With $ISS_{benthos}$, 100% of sites that the index classified as having high ecological status and 72% of sites classified as having good status corresponded to sites in undisturbed lagoons, while 73% of lagoon sites classified as moderate and 82% classified as poor corresponded to sites in disturbed lagoons (Figure 3a). Similar results were also obtained with the $ISS_{phytoplankton}$ (Figure 3b).

In the Margherita di Savoia saltpan and in Lesina Lagoon the ISS showed highly significant patterns of variation along stress gradients, specifically varying as an

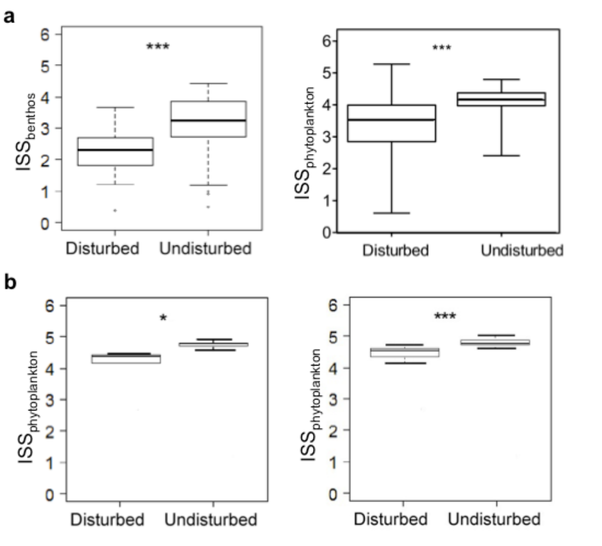

*Figure 2: a) Comparisons of results of the $ISS_{benthos}$ and the $ISS_{phytoplankton}$ tested for "undisturbed" and "disturbed" lagoon sites; b) Comparisons of results of the $ISS_{phytoplankton}$ tested for "undisturbed" and "disturbed" sites in Helsinki marine area (left) and Monpás-Pasaia (right). Horizontal bars in the box-plot graphs represent the mode of value distribution; box-plot heights represent the 25th and the 75th percentiles, and the error bars represent the maximum non-outlier range. Statistical comparison (Wilcoxon rank test) of undisturbed and disturbed sites is reported in each graph as either n.s. = not significant; * = $p < 0.05$; ** = $p < 0.01$; or *** = $p < 0.001$.*

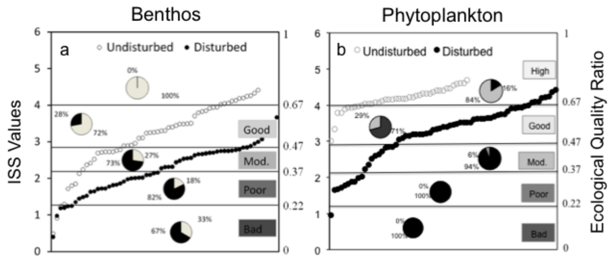

Figure 3: Distribution of studied lagoon sites across ecological quality levels (sensu Water Frame Directive). For each level the relative percentages of undisturbed and disturbed lagoon sites are reported.

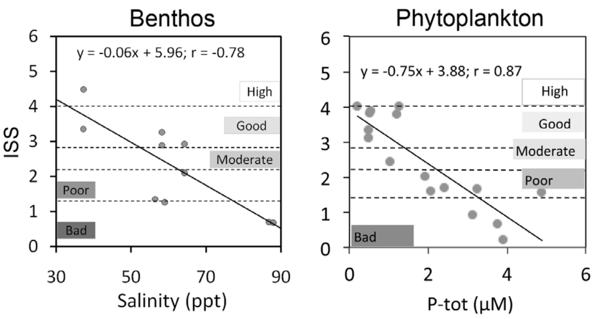

Figure 4: Validation of $ISS_{benthos}$ in Margherita di Savoia saltpan and $ISS_{phytoplankton}$ in Lesina lagoon, showing relationships between ISS values and salinity (left) and total phosphorus concentration (right) are reported. All relationships are significant ($p < 0.001$).

inverse function of salinity in the case of $ISS_{benthos}$ and as inverse function of total phosphorus concentration in the case of $ISS_{phytoplankton}$ (Figure 4 b).

Discussions and conclusions

The accuracy of an ecological indicator is related to the degree to which the selected metric mechanistically describes the relationships between various types of disturbance and biological response.

The ISS offers an approach to understanding the relationships between anthropogenic impact and ecosystem response from the point of view of individuals, whose energetics are decoded from measurements of body size. Variations in the size structure of benthic macroinvertebrates or phytoplankton along a stress gradient can be described with the use of models of size class sensitivity that show remarkable discriminatory power in terms of ecological status assessment in water bodies.

The main advantage of the ISS is its generality; it is likely to be applicable to quality elements other than phytoplankton or macrobenthos and to aquatic ecosystem categories other than transitional and coastal waters. This would favour its widespread use as a monitoring tool in aquatic ecosystems.

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Assessing degradation and recovery pathways in lakes impacted by eutrophication using the sediment record

Helen Bennion, Gavin Simpson & Ben Goldsmith

Environmental Change Research Centre, Department of Geography, University College London, Gower Street, London, WC1H 6BT, UK

Email address of corresponding author: h.bennion@ucl.ac.uk

Introduction

Most lakes have been modified to some extent by human activity. Eutrophication has affected numerous waterbodies, most notably since the mid-twentieth century with the intensification of agriculture and expansion of populations connected to sewage treatment works (Battarbee et al., 2011). The consequent high algal biomass leads to filtration problems for the water industry, oxygen depletion, recreational impairment, loss of biodiversity, fish mortality, and decline or loss of submerged plants.

Efforts to restore enriched systems have increased over the last few decades and there are now many examples of lakes in recovery as a result of active catchment and in-lake management. Recovery is however a complex process, with biotic communities tending to exhibit hysteresis and time-lags, and thus ecosystems take time to re-adjust to reduced stress (e.g. Yan et al., 2003; Johnson & Angeler, 2010). Furthermore, new pressures, especially from global warming, may counter restoration strategies. Thus the expectation that ecosystems can be returned to similar conditions that existed prior to enrichment may be a naive one and managers and policy makers may have to accept that "shifting baselines" will limit the ability to meet restoration targets (Duarte et al., 2009; Bennion et al., 2011a).

Here, palaeoecological techniques are employed to examine the degree of impact and recovery in thirteen European lakes that have been subject to eutrophication and subsequent reduction in nutrient loading. The response of several diatom-based metrics is explored including percentage plankton, diversity, community composition and diatom-inferred TP (DI-TP) concentrations. The key questions being addressed are: i) Do the observed changes reflect degradation and recovery? ii) Is the recovery pathway simply a reversal of the degradation pathway?, and iii) Can the degree of degradation be quantified?

Methods

Sediment cores from thirteen European lowland lakes were analysed for diatoms (Battarbee et al., 2001) spanning a period of approximately the last 200 years. The lakes represent a range of lake types in terms of surface area, depth and trophic status (Table 1) and for data analyses sites have been classed as either deep, stratifying or shallow, non-stratifying, in order to explore whether these lake types respond differently to nutrient reduction measures. All of the sites have experienced eutrophication within the last ~100 years and have seen a reduction in external nutrient loadings over the past 2-3 decades (Table 1).

The down-core diatom data were explored using a number of different metrics including squared chord distance (SCD) dissimilarity coefficient (Overpeck et al., 1985), percentage of planktonic taxa versus non-planktonic taxa, the Hill's N2 diversity score (Hill & Gauch, 1980) and principal components analysis (PCA). A diatom-TP transfer function was also applied to the diatom data to reconstruct the trophic status of each site (Bennion et al., 1996). The PCA scores on axis 1 and 2 of each core are also displayed passively on a covariance matrix of samples from the modern diatom-TP training sets, with logTP as a supplementary environmental variable. The logTP values are represented by generalized additive model (GAM) contours. The plots allow the direction of floristic change at each site and its relation to TP to be visualised.

Results

The dissimilarity scores between core bottom and other samples in each core indicate that all sites have experienced deviation from reference condition (core bottom sample) over the period represented by the cores (Figure 1a). The diatom assemblages of some sites, most notably the deep lakes, show signs of returning towards

Table 1: Summary characteristics of the thirteen study sites.

Site name	Country	Alt (m asl)	Lake Area (km^2)	Max Depth (m)	Current mean TP	Lake type	Management actions
Barton Broad	*England*	*2*	*0.77*	*1.5*	*74*	*Shallow, non-stratifying*	*Reduced external P loading since late 1970s; sediment removal to reduce internal P-loading from 1995-2000*
Lake Bled	*Slovenia*	*475*	*1.5*	*32.0*	*20*	*Deep, stratifying*	*Sewage effluent diversion in 1982*
Bosherston Lily Pond Central	*Wales*	*2*	*0.34*	*2.0*	*20*	*Shallow, non-stratifying*	*Sewage diversion since 1984, bypass pipeline construction in 1992*
Esthwaite Water	*England*	*65*	*1*	*15.5*	*28*	*Deep, stratifying*	*Reduced P loading since 1986 but internal loading issues and fish farm present until 2009*
Gjersjoens	*Norway*	*40*	*2.4*	*64.0*	*15*	*Deep, stratifying*	*Sewage effluent diversion in 1971*
Kiełpińskie	*Poland*	*120*	*0.61*	*11*	*105*	*Deep, stratifying*	*Decrease in fertiliser use and change in land use in early 1990s*
Loch Leven	*Scotland*	*106*	*13.7*	*25.5*	*53*	*Shallow, non-stratifying*	*Reduced P loading since 1985 but internal loading issues*
Lidzbarskie	*Poland*	*128*	*1.22*	*25.5*	*66*	*Deep, stratifying*	*Decrease in fertiliser use and change in land use in early 1990s*
Llangorse Lake	*Wales*	*156*	*1.4*	*9.0*	*118*	*Shallow, non-stratifying*	*Sewage diversion in 1981 and 1992*
Marsworth Reservoir	*England*	*115*	*0.1*	*4.0*	*476*	*Shallow, non-stratifying*	*Sewage part-diversion and improved sewage treatment works in mid 1980s*
Mill Loch	*Scotland*	*55*	*0.11*	*16.8*	*92*	*Deep, stratifying*	*Exact restoration measure and timing unknown*
Mjøsa	*Norway*	*123*	*362*	*453.0*	*4*	*Deep, stratifying*	*Improvements to sewage treatment works in late 1970s*
Rumian	*Poland*	*152*	*3.06*	*14.4*	*75*	*Deep, stratifying*	*Decrease in fertiliser use and change in land use in early 1990s*

the reference flora following reduction of nutrient load. Nonetheless most are still far from reference condition with high dissimilarity scores between the core top and bottom samples.

At four of the five shallow lakes the % plankton increases with enrichment but does not decline during the recovery phase. In the deep lakes % plankton was high throughout the cores but in Esthwaite Water, Gjersjoen, Mill Loch and Mjoesa slight increases in the planktonic component were observed with enrichment. Only in Mjoesa, and to a lesser extent in Esthwaite Water, was a slight decline in % plankton seen in the recovery period. Shifts in diatom diversity with enrichment and recovery were mainly inconclusive.

The PCA axis 1 scores show that all sites experienced shifts in diatom composition during the eutrophication phase. However, only five sites return towards an assemblage similar to that prior to enrichment following a reduction in nutrients (Figure 1b). The data suggest that for the remaining lakes the diatom flora following nutrient reduction exhibits change but is not returning towards the pre-enrichment assemblage.

The diatom transfer functions infer an increase in TP concentrations in eight of the study lakes during the enrichment period (Figure 1c) and a subsequent decline in TP concentrations following a reduction in nutrient loading at 12 of the 13 study lakes (no change at Kielpinskie). This suggests that at these 12 sites there have been compositional changes towards taxa associated with lower nutrient concentrations following the nutrient reductions.

When the deep lake cores are plotted passively on a PCA covariance matrix of the large, deep lakes training set samples with logTP as a supplementary environmental variable the core samples generally follow the direction of increasing TP concentrations in the training set. A reversal is seen in Lake Bled, Gjersjoen, Mill Loch, and Mjoesa (Figure 2) where samples move back towards the upper right following a reduction in nutrient loading. This reverse pattern is not seen in Esthwaite Water nor is a clear pattern observed for the three Polish lakes.

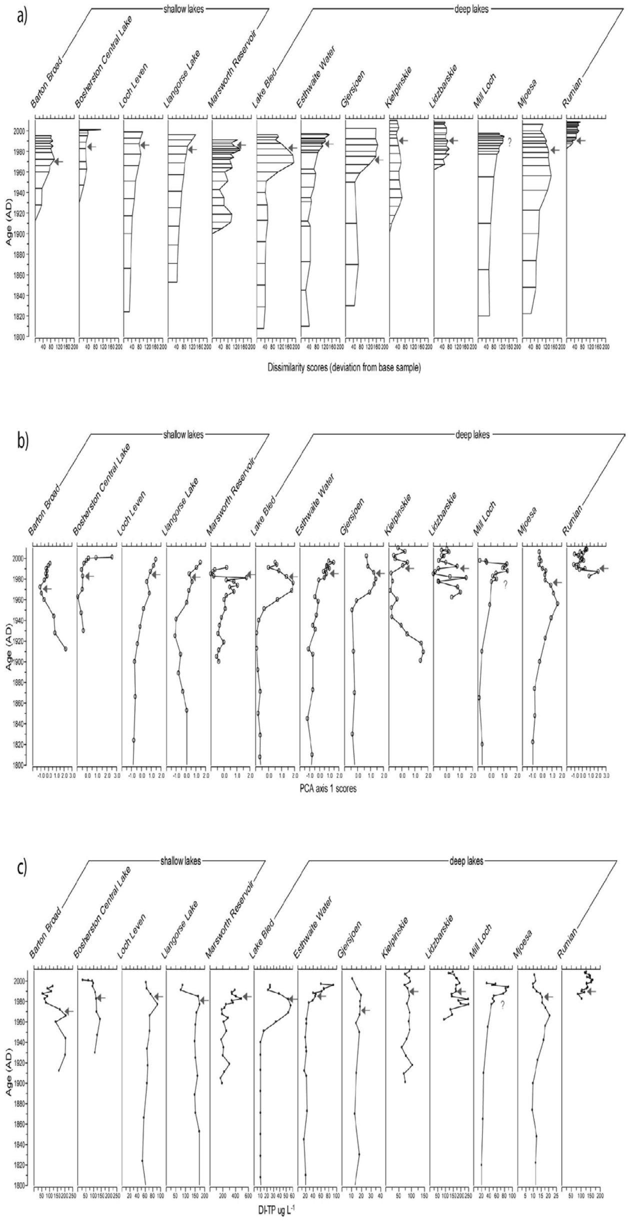

Figure 1: Diatom metrics from the thirteen study sites showing: a) Dissimilarity scores b) PCA axis 1 scores c) Diatom-inferred TP (DI-TP) reconstructions (timing of nutrient reduction is shown by the arrow)

The core samples of all five shallow lakes move from the right of the plot towards the left during the enrichment period, following the direction of increasing TP concentrations in the training set. A clear reversal is apparent only at Marsworth Reservoir following nutrient reduction, while a slight move back towards the right of the diagram is seen at Loch Leven. At Barton Broad, Bosherston Lily Pond, and Llangorse Lake, the upper core samples move to a new position within the ordination space but do not track back along the enrichment trajectory (Figure 3).

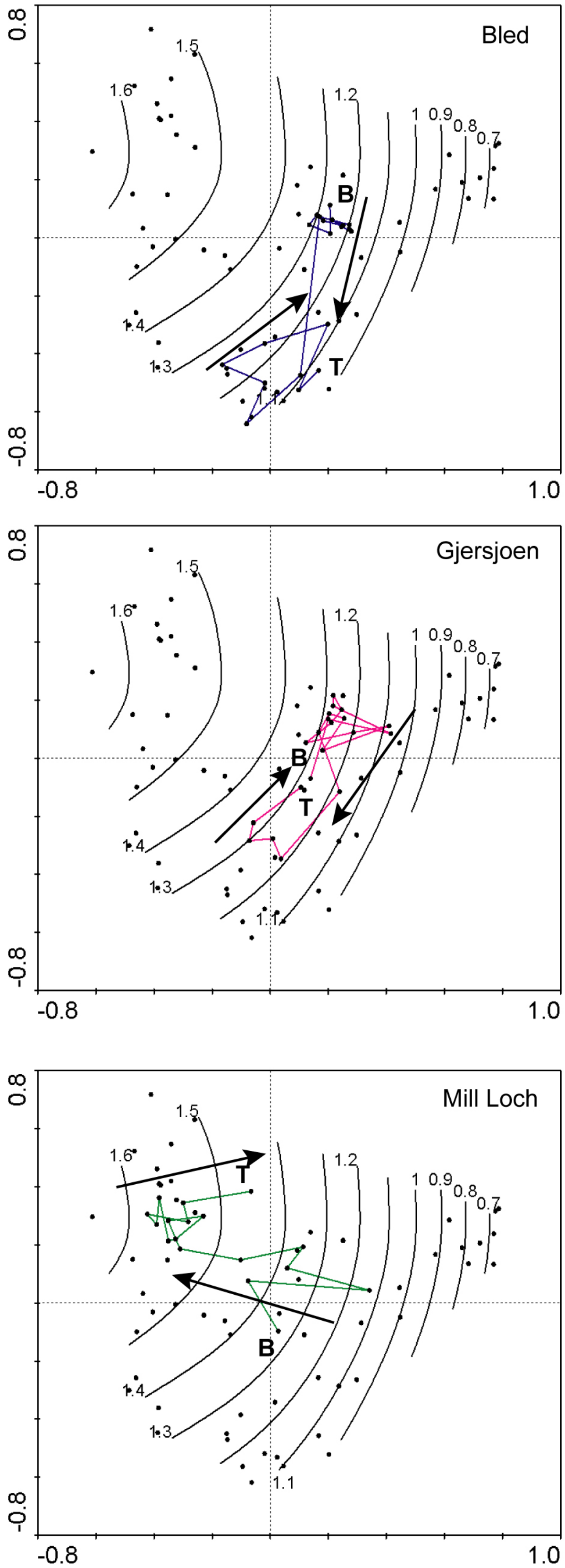

Figure 2: Three of the deep lake cores plotted passively on a PCA covariance matrix of training set samples with logTP as a supplementary environmental variable (logTP values, µg l-1, represented by GAM contours). The direction of change over time is shown by the arrows (T = core top and B = core bottom)

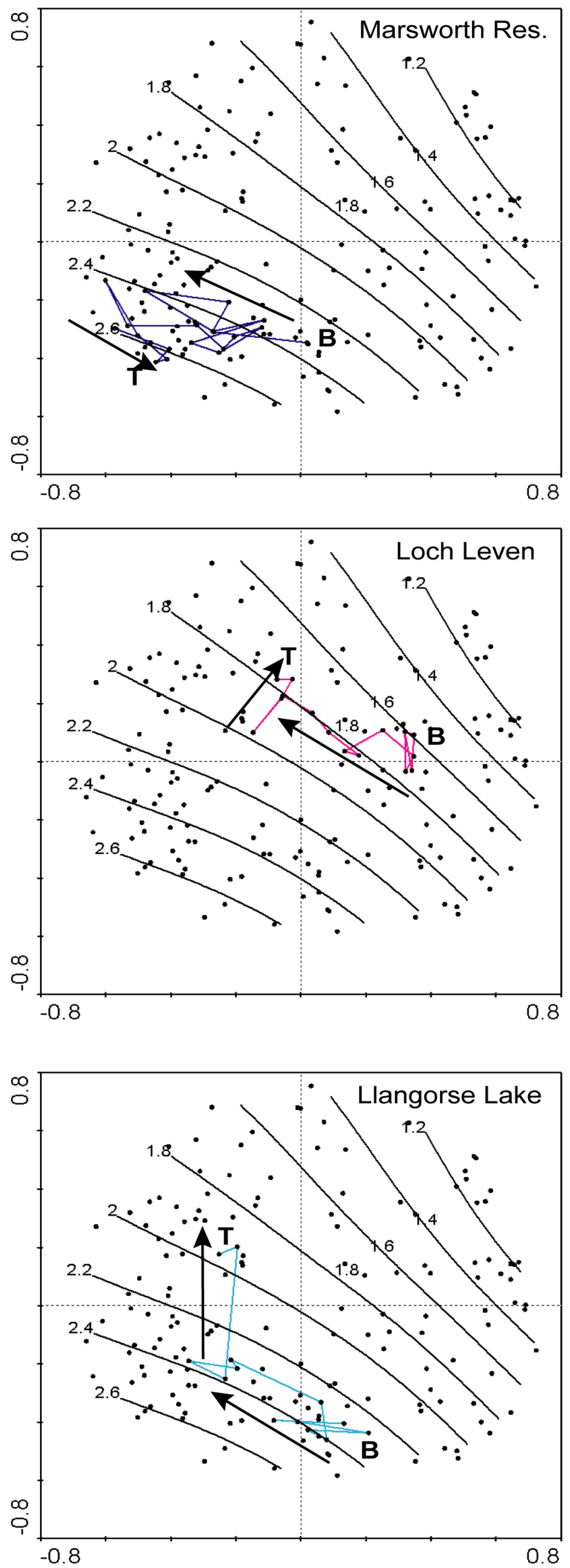

Figure 3: Three of the shallow lake cores plotted passively on a PCA covariance matrix of training set samples with logTP as a supplementary environmental variable (logTP values, µg l-1, represented by GAM contours). The direction of change over time is shown by the arrows (T = core top and B = core bottom)

Discussion

The diatom data demonstrate a progressive deviation from the reference condition occurred at all sites during the eutrophication phase, but in most cases a direct reversal towards reference conditions did not occur following nutrient reduction. Our findings accord with the findings from coastal and riverine studies (e.g. Duarte et al. 2009, Palmer et al. 1997) and highlight that whilst in some cases the diatom recovery trajectories do appear to track back along the degradation pathway, in others (particularly shallow lakes) either little sign of recovery is evident or the assemblages follow a new trajectory owing to the host of other environmental factors. While these palaeoecological data are not without their limitations, when combined with long-term datasets, a palaeolimnological approach would perhaps provide a more powerful tool for assessing timescales of ecological change (Battarbee et al., 2005; Bennion et al., 2011b; Dong et al., 2011).

The slow rate of recovery demonstrated by some of the sites in this study has major implications for the WFD which requires waterbodies to be restored to at least good status, initially by 2015. For sites where recent management has been implimentent it could be several decades before any recovery is seen. Perhaps even more importantly the data suggest that for some lake systems the assemblages following remedial action may not return back down the degradation pathway at all and, therefore, reference conditions are unlikely ever to be achieved. The key message arising from the case studies examined here is that for the most part managers are advised to isolate the main source(s) of nutrients and then wait. In most of our study lakes, the main point source of nutrients, principally P, has been the key focus of management efforts. However, in recent decades diffuse nutrient sources have become relatively more significant than urban wastewater pollution and losses from agricultural land are now the biggest challenge. There has been a growing literature on the need to reduce nitrogen (N) loads as well as P in order to reverse eutrophication, particularly in shallow lakes with moderate P levels where high summer N concentrations stimulate algal growth and cause loss of submerged plants (e.g. Jeppesen et al., 2007). Indeed, a recent assessment of nutrient sources to Llangorse Lake revealed the importance of reducing N inputs if restoration targets are to be met (May et al., 2010).

The role of climate change in exacerbating the symptoms of eutrophication and confounding recovery efforts cannot be ignored. The examination of the role of climate change in explaining the shifts in the diatom assemblages on two of the lakes, Esthwaite Water

(Dong et al., 2011) and Loch Leven (Bennion et al., 2011c) has attempted to explore the ways in which nutrients and climate interact on decadal and inter-annual timescales to affect the diatom communities. While the relationships are clearly complex, in both of these studies the presence of Aulacoseira granulata and Aulacoseira granulata var. angustissma seems to coincide with warmer temperatures. Such investigations contribute to a better understanding of the effects of multiple environmental drivers on aquatic ecosystems but equally also illustrate the complexity of ecosystem response to synchronous changes in nutrients and climate, and the difficulty of disentangling the effects of these interacting pressures.

Models that predict likely outcomes of climate change on nutrient regimes will play a vital role in improving our understanding of future lake response and in guiding management decisions (e.g. Whitehead et al., 2006). Sediment records provide an opportunity to validate hindcasts derived from dynamic models (Anderson et al., 2006) and should therefore play an important role in assessing uncertainty associated with future predictions. Conceptual models based on the DPSIRR scheme, i.e. the 'Driver-Pressure-State-Impact-Response-Recovery chain', as adopted in the EU WISER project, will also be important for providing guidance on the ecological effectiveness of restoration measures (Feld et al., 2010). In the context of the present study, the key **Drivers** are urbanisation and agricultural intensification, the **Pressure** is eutrophication, the **State** is trophic status and nutrient concentrations, the **Impact** explored is that on one of the algal groups, diatoms, in terms of community composition, functional groups and diversity, the Response is nutrient load reduction principally by management of sewage effluent but also by good agricultural practice in some instances, and the Recovery relates to shifts in the structure of the diatom assemblages. The study illustrates, therefore, that palaeoecology is a valuable tool for populating conceptual models of this kind.

Conclusions

In terms of the original questions posed we can conclude that the observed changes in the diatom records do reflect both the degradation and the recovery process. The latter has reached a different stage in each of the study lakes and is more clearly seen in the deep lakes where the diatom assemblages have started to revert back toward those seen prior to enrichment. In shallow lakes factors such as internal loading and top down control may influence the recovery process and in this study, whilst the assemblages of several shallow lakes were replaced by ones associated with lower productivity following remediation, they did not track back along the enrichment pathway. It can, therefore, be concluded that the deep stratified lakes tend to follow a more predictable recovery pathway than the shallow lakes. Nevertheless, the recovery process has a long way to go in all cases, as the present assemblages remain very different from those seen in the pre-enrichment samples. Dissimilarity and ordination scores can be used to quantify the deviation from reference condition. The study also highlights the important role that paleaolimnological approaches can play in establishing a benchmark against which managers can evaluate the degree to which their restoration efforts are successful.

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Characterization of cesium removal potentiality of some specific soils to reclaim aquatic environment

Jatindra N. Bhaktaab[1, 2], Yukihiro Munekageb[2], Kouhei Ohnishia[1], Kozo Iwasakic[3], Bana B Janad[4]

[1] Research Institute of Molecular Genetics, Faculty of Agriculture, Kochi University, B200, Monobe, Nankoku, Kochi - 783-8502, Japan
[2] Department of Environmental Engineering, Faculty of Agriculture, Kochi University, B200, Monobe, Nankoku, Kochi - 783-8502, Japan
[3] Life and Environmental Medical Science Cluster, Faculty of Agriculture, University of Kochi, B200, Monobe, Nankoku, Kochi - 783-8502, Japan
[4] International Centre of Ecological Engineering, University of Kalyani, Kalyani 741235, West Bengal, India

Email address of corresponding author: lsnjbhakta@gmail.com

Key words: Specific soil, Cesium, Adsorption, Isotherm, Water reclamation

Introduction

Cesium (^{133}Cs) is a naturally-occurring element found in rocks, soil, and dust at low concentrations. Stable Cs has shown behavioral changes, increased or decreased activities in animals at high doses. Researchers stated that fatigue, muscle weakness, palpitations and arrhythmia are also the symptoms of cesium toxicity [1]. Radioactive Cs causes cell damage, nausea, vomiting, diarrhoea and bleeding and leads to coma or even death in long term exposure [2]. The major sources for aquatic contamination of radioactive Cs are radioactive sites, nuclear power station, nuclear explosions or the breakdown of uranium in fuel elements.

From the above viewpoints, it is necessary to remove Cs from contaminated water to control cesium pollution in environment. Several improved and innovative techniques, chemical precipitation [3], coagulation [4], ultra filtration [5], biologic [6], and adsorption [7] are employed to remove toxic metals from water. In this respect, adsorption is one of the simple and cost effective techniques. Alba et al. [8] and Asfari et al. [9] removed Cs from water using the adsorption process. Recently, scientists developed low-cost sorbents, minerals and soils, zeolites and clay/soil based ceramic [10]–[13] to adsorb various toxic metals and metalloids. The present investigation also attempted to find out the simple and low-cost solution for removing Cs from water using the potential soil adsorbent. The present study employed akadama volcanic ash (AS) and shirasu soils (SS) have been characterized as a good low-cost Hg removing agent [11], whereas no such studies concerning the metal removal properties of izumo soil (IS) was performed so far though it is used as tertiary treatment of wastewater. Therefore, the objective of present study was to characterize the Cs removal properties of these specific soils in order to apply as efficient adsorbent media in practical field to reclaim the aquatic environment.

Materials and methods

Three types of soil were collected from three places [AS, Tochigi prefecture; IS, Shimane; SS, Kagoshima] of Japan, air dried, homogenized by mortar and pestle and sieved for experiment (Fig. 1a,b,c).

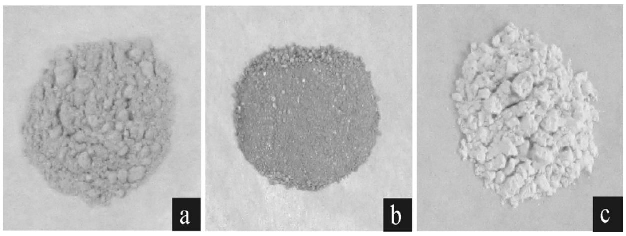

Figure 1: Photographs of AS (a), IS (b) and SS (c) employed in present Cs removal study.

Chemical characterization of soil samples was also executed by scanning electron microscopy (SEM) coupled with energy dispersive spectroscopy (EDS) facility (JSM-6500F, JEOL). The study used batch adsorption experiments to determine the Cs removal characteristics of soils under different conditions. Experiments were conducted in capped glass bottles (0.1 L) using known weight of soils, concentration of Cs (CsCl, Kanto Chemical Co., Inc., Japan) and volume of water (0.05 L). The effect of contact time on Cs removal process of soils was determined maintaining the initial concentration 4 mg/L (pH 7.3) and soil dose 1 g/L. Cs adsorption capacity was characterized using the soils dosages, 0.1, 0.3, 0.5 and 1g/L with initial concentration 4 mg/L (pH 7.3). The mean value of at least two studies maintaining identical conditions was considered for data analysis. The Cs sorption capacity (q_e, mg/g) of soils at equilibrium was calculated by following the equation of mass balance relationship [7]. Practical water treatment study was carried out using 9 (100 mL) conical flasks in laboratory condition. All conical flasks were randomly divided into three groups having three replicates (3 x 3) for three soils, AS, IS and SS. Each conical flask was provided with respective soil @ 1 g and gently filled with Cs contaminated wastewater (2.4 mg/L, pH 8.2) generated from our laboratory experiment. Experiment was continued for 48 h period. Concentrations of Cs and other metal ions (Na, K, Ca and Mg) in water were examined at 0, 24 and 48 h periods. The collected water samples were analyzed using the ion chromatography (ICS 900, Dionex Corporation, Sunnyvale, CA, USA).

Results and discussion

Chemical composition

All soils predominantly constituted of silicon dioxide (25.82 – 71.44%) and aluminum oxide (13.94 – 25.74%) with remarkable variations. Silicon dioxide and magnesium oxide content are higher in SS compare to that of the remaining soils.

Cs adsorption characterization

Removal of Cs varied from 0.15 to 0.2 mg/g in different contact times of soils. It pronounced a sharp increasing trend at 20 min period of experiment followed by gradual increasing trend thereafter and equilibrium state achieved at about 120 min in all soils (Fig. 2a). The q values of SS (0.2 mg/g) and IS (0.19 mg/g) were 9 and 11% greater, respectively compare to AS at the equilibrium state. From this experimental condition, IS and SS would be considered as efficient media to remove the maximum amount of Cs from aquatic phase.

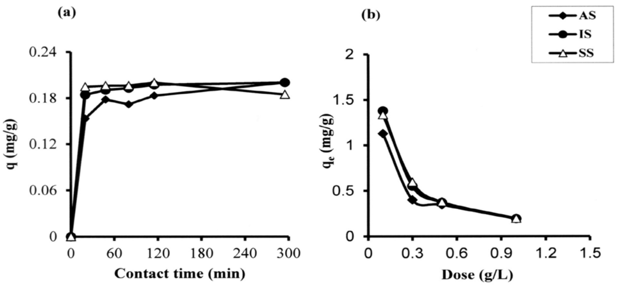

Figure 2: Influence of contact time (a) and soil dosages (b) on the Cs adsorption capacity of AS, IS and SS.

The influence of dosages on Cs adsorption capacity (q_e) of soils was depicted in Fig. 2b. Total Cs sorption increased with increasing soil dosages, whereas adsorption capacity (q_e, 0.19 – 1.37 mg/g) exhibited a declining trend with increasing the soil dosages in three types of soils. Results clearly revealed that Cs adsorption capacities (q_e) of IS and SS were higher compare to AS in lower two doses, whereas no remarkable differences were observed in higher two doses in all types of soil. The decreasing of sorption rate with the increasing soil doses may result from the electrostatic interactions, interference between binding sites, and reduced mixing for higher densities at higher doses.

Adsorption isotherms

In isotherm study, the Langmuir and the Freundlich models are commonly used to determine the effectiveness of adsorbent under different operational conditions. The equations of Langmuir (1) and Freundlich (2) models used in the study are as follows:

$$q_e = \frac{K_L q_m C_e}{1 + K_L C_e} \qquad (1)$$

Where, KL = Constant related to free energy of adsorption, C_e (mg/g) = Equilibrium concentration in solution, q_m (mg/g) = Adsorption capacity corresponding to the complete monolayer coverage

$$q_e = K_F C_e^{1/n} \qquad (2)$$

Where, K_F (L/mg) = Freundlich constant, C_e (mg/g) = Equilibrium liquid phase concentration, 1/n = Heterogeneity factor.

The equilibrium curve of Cs adsorption and fittings parameters of three soils were presented in Fig. 3a,b and Table 1, respectively. The R^2 values of Langmuir and Freundlich revealed that AS is well fitted with both isotherms, whereas IS and SS are best fitted with only Freundlich isotherm.

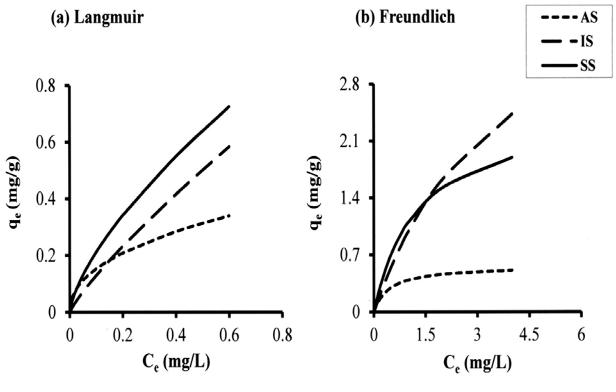

Figure 3: Equilibrium curves of soils employed for adsorption of Cs.

Table 1: Parameters of Langmuir and Freundlich isotherms.

Soils	Langmuir			Freundlich		
	K_L (L/mg)	q_m (mg/g)	R^2	1/n	K_F	R^2
AS	*2219*	*0.0005*	*0.9942*	*0.4439*	*0.4261*	*0.9927*
IS	*260*	*0.0047*	*0.2063*	*0.8391*	*0.8966*	*0.9909*
SS	*785*	*0.0024*	*0.8639*	*0.6859*	*1.0297*	*0.9905*

Practical water treatment approach

A substantial amount of Cs was removed from water column by AS (2.26 mg/L, 94%), IS (2.19 mg/L, 90%) and SS (2.35 mg/L, 97%) at 24 h period of experimentation (Fig. 4). There was no remarkable difference between the total Cs removal of 24 and 48 h periods in all soils. Registered final pH of the treated water were 7.5, 6.6 and 8.4 in AS, IS and SS, respectively. . Ion concentrations of Na, K, Ca and Mg were also decreased with few exceptions at 24 and 48 h, which indicating that tested soils possess capacity to remove other ions along with Cs. High magnitude of Cs removal capacity indicating that tested soils could be applied as low-cost adsorbent media in practical reactor systems and constructed wetlands to treat the contaminated wastewater. Application of clay based ceramic as a vesicle with plant in the constructed wetland treatment system is a new approach for wastewater reclamation [12], [14].

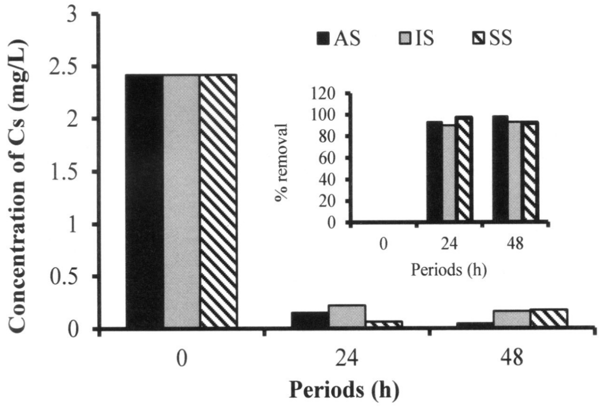

Figure 4: Practical Cs removal efficiency of soils from wastewater (inset depicting the removal percentage).

The present study revealed that examined soils have high potential in removing Cs from aquatic environment. Cs adsorption capacity is comparatively better in sirashu soil among three soils probably due to higher magnesium oxide content which was proved by our previous study. 120 min could be considered as equilibrium contact time. From the viewpoints of economic feasibility, availability and simple application method, therefore, tested soils could be employed as efficient sorbent media for treating Cs contaminated water in order to reclaim the aquatic environment.

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Europe's quest for common management objectives of aquatic ecosystems: a preliminary overview of intercalibration results

Sebastian Birk[1], Wendy Bonne[2], Wouter van de Bund[2,] Sandra Poikane[2], Nikolaus Zampoukas[2]

[1] University of Duisburg-Essen, Faculty of Biology, Applied Zoology/Hydrobiology, Germany
[2] European Commission, Joint Research Centre, Institute for Environment and Sustainability, Italy

Email address of corresponding author: sebastian.birk@uni-due.de

Introduction

Anthropogenic activities cause ecosystem deterioration worldwide, resulting in loss of biodiversity and impoverished ecosystem services. Aquatic systems are among the most degraded habitats, yet supplying the human demands for food, freshwater and power generation. Halting and reversing the process of deterioration is a global challenge and requires concerted action. The principal tool for the coordinated protection of aquatic ecosystems is the river basin management planning that acts across state boundaries and administrative barriers. To identify management priorities harmonised information about the ecosystems' quality status is indispensable. However, the definition of uniform standards and common quality targets is hampered by the multitude of different assessment methods applied.

A key element of harmonised quality classification within and between Europe's river basins is the intercalibration exercise (IC) stipulated by the Water Framework Directive. In this exercise countries compare their classification of good ecological status for similar ecosystem types across large geographical areas. The aim of intercalibration is to ensure a consistent level of ambition in the protection and restoration of surface waters among member states of the European Union. In simple terms, the intercalibration exercise assures that, for instance, a French lake in good status according to the French assessment method would also be classified in good status by the Polish or German methods if it was located in Poland or Germany, respectively.

Here, we provide a preliminary overview of results achieved in the intercalibration exercise.

IC overview indicators

Basic units to measure the intercalibration achievements were the individual exercises carried out among different national assessment methods of the same biological quality element (BQE) and water category for a common intercalibration type, e.g. river macrophytes at siliceous mountain brooks of the Central-Baltic Geographical Intercalibration Group. We compiled data from the official IC reports, considering all information submitted to the European Commission until December 15th, 2011.

The following two overview indicators were selected:

(i) Coverage of intercalibration

In theory, a country should have intercalibrated the assessment methods of each BQE defined for each water category located within its territory. We related the number of methods that were intercalibrated to this theoretical value to gain the ratio of intercalibrated methods per water category.

(ii) Feasibility of intercalibration

The overall level of comparability among national classifications depends on how similar the assessment methods are. Well-correlated national methods ensure a highly comparable classification of water bodies, while poorly correlated methods will classify individual water body differently, even after successful intercalibration. We calculated the average coefficients of determination resulting from the regression analyses of national assessment methods against the (pseudo-)common metrics, and tested for significant differences among BQEs.

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IC results

Our overview comprises 102 individual exercises in which the quality classifications of 29 countries were compared. Most exercises were completed for lakes (59 %), followed by rivers (29 %), coastal waters (9 %) and transitional waters (3 %). The biological element phytoplankton was most often intercalibrated (42 %). Benthic invertebrates and macroscopic plants (including macrophytes, angiosperms and macroalgae) held 22 % and 21 % of exercises, respectively. Least represented elements were phytobenthos (9 %) and fish fauna (7 %).

The river exercises covered 60 % of the national assessment methods required for WFD monitoring (Figure 1). The average number of countries involved per exercise was 6.0. For lakes we recorded a smaller average number of countries involved (4.8), resulting in only 43 % of accomplished intercalibration for this water category.

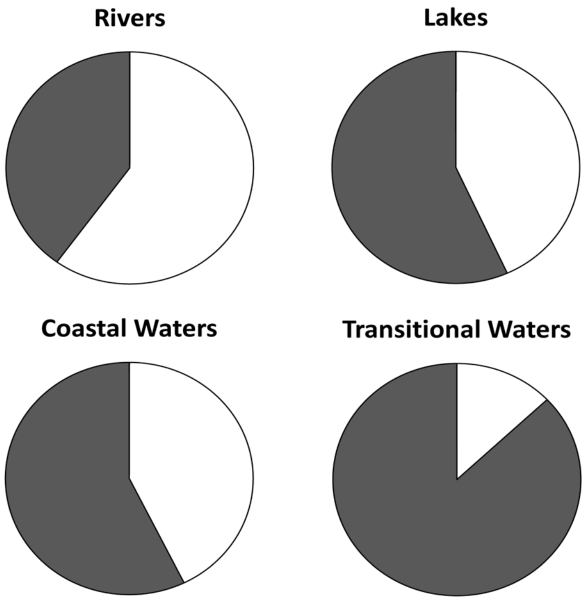

*Figure 1: Share of accomplished intercalibration per water category – intercalibrated national assessment methods (at least for one common type) related to the number of methods required for national Water Framework Directive monitoring (country * water category * BQE). white: share of intercalibrated methods, grey: share of methods not intercalibrated.*

Coastal exercises involved on average 3.4 countries and showed a similar share of intercalibrated national methods (43 %). Least intercalibrated were the methods for transitional waters (13 %). Here, on average three countries participated in an exercise.

The coefficient of determination that expresses the correlation of national methods covered a range from 0.30 to 0.92 (Figure 2). Phytobenthos-based methods were most similar (median coefficient of 0.82), fish-based methods were most dissimilar (median: 0.41). The average coefficients of methods using benthic invertebrates differed significantly between lakes (median: 0.39) on the one hand, and rivers and coastal waters on the other hand (median: 0.71) (Mann-Whitney U-test, p<0.05).

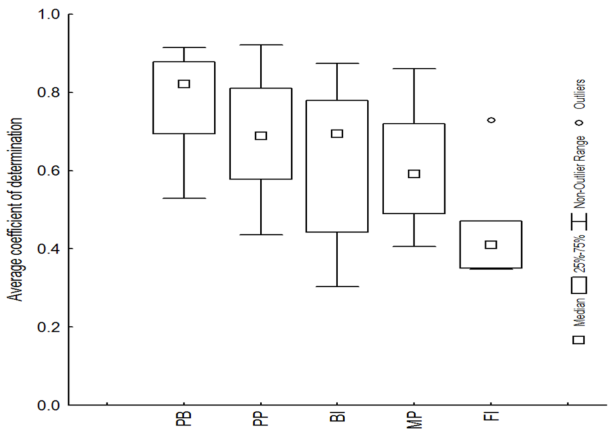

Figure 2: Range of coefficients of determination per BQE indicating the strength of relationships among national assessment methods. Differences between BQEs are significant (Kruskal-Wallis test, p=0.007). The analysis included 70 individual IC exercises. PB=Phytobenthos, PP=Phytoplankton, BI=Benthic invertebrates, MP=Macroscopic plants, FI=Fish fauna

Summary

The intercalibration exercise represents a thematic and organisational novelty in environmental policy. For the first time, an international regulation stipulated the achievement of good ecological status along with the demand of safeguarding comparable ambitions among countries. The exercise created a platform for a pan-European dialogue on environmental objectives and the assessment of ecological quality, involving more than 500 scientists and water managers.

Our contribution is based on preliminary intercalibration results; a final overview can be given after all IC results were submitted and approved in early 2012. However, we could already demonstrate that (i) for many BQEs and water categories intercalibration is not complete, and (ii) the level of comparability among different BQEs varies. While the first requires continuous efforts of the EU member states (incl. method development), the second is an intrinsic feature of the different bioassessment methods that intercalibration cannot resolve.

Transitional and coastal water assessment within the Water Framework Directive: advances produced by WISER project

Angel Borja[1], Michael Elliott[2], Peter Henriksen[3] & Nuria Marbà[4]

[1] Marine Research Division, AZTI-Tecnalia
[2] Institute of Estuarine and Coastal Studies, University of Hull
[3] Department of Biosciences, University of Aarhus
[4] Department of Global Change Research, Instituto Mediterráneo de Estudios Avanzados, CSIC-UIB

Email address of corresponding author: aborja@azti.es

Key words: methods of assessment, uncertainty, response to pressures, biological quality elements

Introduction

Transitional and coastal waters were investigated in Module 4 of the project "Water bodies in Europe: Integrative Systems to assess Ecological status and Recovery" (WISER). The main objective of Module 4 was to provide a complete set of assessment systems for the Biological Quality Elements (BQEs) relevant for coastal/transitional waters (phytoplankton, macroalgae/angiosperms, benthic invertebrates, and fish), which requires the validation of indicators, and in some cases the development of new indicators. Hence, this Module was organized into four Work Packages (WP), one for each BQE. During the investigation, particular focus was paid to the most important human stressors affecting BQEs.

Together with the existing extensive datasets, available from each partner, a field sampling survey, addressing the four above mentioned BQEs, was performed at a series of sites throughout Europe (Figure 1), using harmonised sampling and analytical methods. The main goal of this exercise was to gain comparable data for the uncertainty estimation exercise of the assessment methods.

Accounting for differences between BQE-data across Europe, each WP focused on different sites and water types. Hence, in some of the WPs the field survey focused on a Mediterranean lagoon (Lesina lagoon, in Italy), a coastal area in the Mediterranean (for angiosperms, Balearic islands, in Spain), a transitional and coastal area within the Black Sea (Varna bay and lagoon, in Bulgaria), a medium-size Atlantic estuary (Mondego, in Portugal), a coastal Atlantic area (the Basque coast, in Spain), a Norwegian fjord (Oslofjord) and Helsinki Sea (in Finland) (Figure 1). This sampling scheme covered a wide range of geographical regions and types, including both transitional and coastal waters.

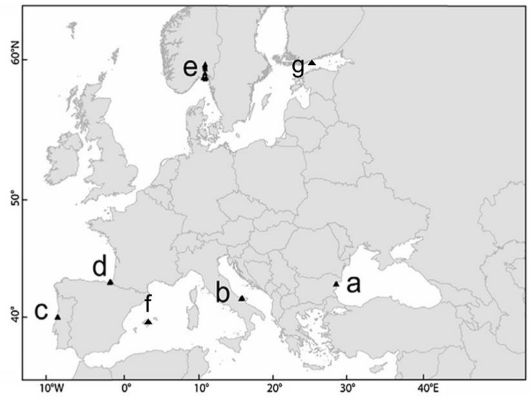

Figure 1: Coastal and transitional locations sampled during the WISER project. (a) Varna bay and lagoon; (b) Lesina lagoon; (c) Mondego estuary; (d) Basque coast; (e) Oslofjord; (f) Balearic islands; and (g) Helsinki Sea.

The common objectives for all WPs within this Module were:

– Develop and validate new indicators and multimetric indices, when necessary for some BQEs,
– Identify pressure-response relationships for different BQEs
– Define reference conditions, when necessary, for different BQEs
– Evaluate uncertainty on the use of existing and new assessment methods

These objectives related to the conceptual model linking the DPSIR (Drivers-Pressures-state Changes-Impacts-Responses) approach to the assessment indices development and validation, response to pressures and setting reference conditions (Figure 2). In this scheme, this Module investigated in: pressures, gradients, reference conditions for state change, index development for impact assessment and validation of the indices.

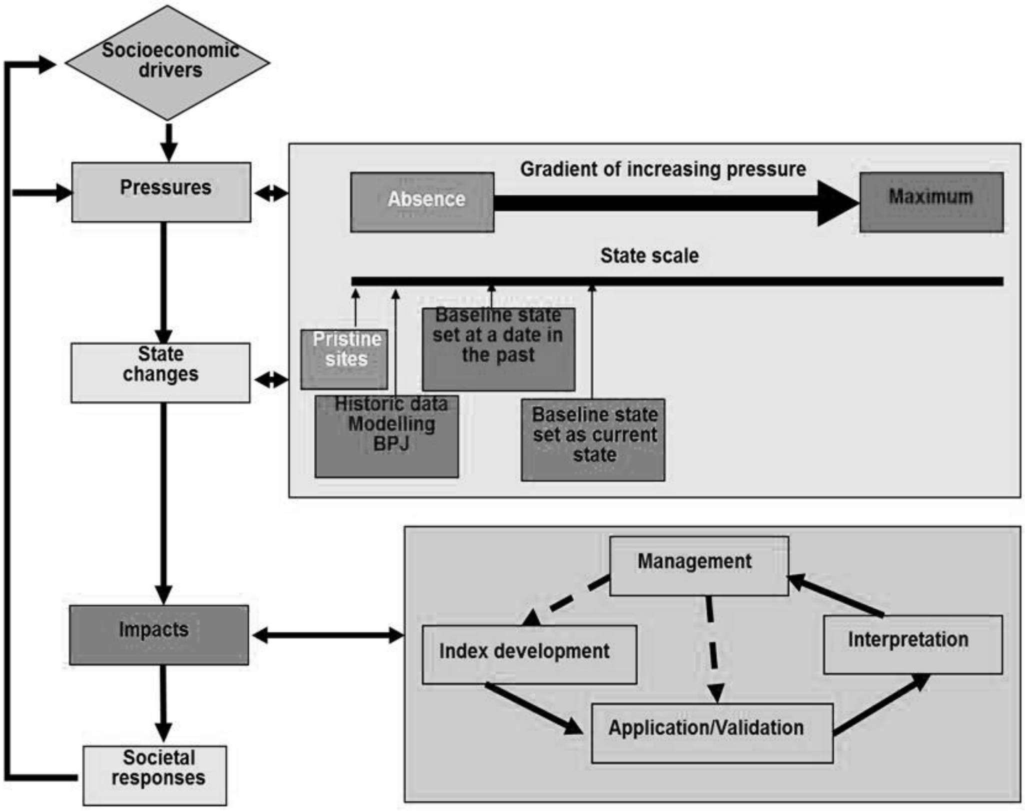

Figure 2: Conceptual model linking the DPSIR (Drivers-Pressures-state Changes-Impacts-Responses) approach to the assessment indices development and validation, response to pressures and setting reference conditions, as it was undertaken during the WISER project in marine and transitional waters. BPJ: Best Professional Judgment.

Develop and validate new indicators and multi-metric indices

Each WP evaluated the need of developing new assessment methods, taking into account the lack of methods for some BQEs (e.g. macroalgae), ecotypes (e.g. hard substratum, lagoons, etc.), development of new metrics (e.g. size spectra in phytoplankton and macrobenthos) or use of new methods in sampling or identification (e.g. FLOWCAM (Garmendia et al., in prep); and satellite derived assessment (Novoa et al., in prep), in phytoplankton). Hence, at least four new assessment methods have been developed for phytoplankton (Lugoli et al., in prep), macroalgae (Díez et al., 2012; Neto et al., in press) and macrobenthos (Basset et al., 2012). Additionally, two papers reviewing the indicators for angiosperms (Marbà et al., in prep.) and fishes (Pérez-Domínguez et al., submitted) have been prepared.

Identification of human pressure-response relationships

Module 4 has evaluated a large number of existing datasets, performing a joint field exercise of all WPs to obtain a common database covering several BQEs and coastal/transitional water types in Europe (see Figure 1). When studying the response of BQEs to human pressures, the major stressors considered were: (i) Hydromorphological pressure, mainly in transitional waters (e.g. structural changes, residence and flushing time alterations), including the assessment of the good ecological potential of Heavily Modified Water Bodies (HMWBs) (e.g. in the case of fishes); (ii) Eutrophication (restricted to selected BQEs, such as phytoplankton, macroalgae and angiosperms); and (iii) Pollution (metals and organic compounds), affecting disturbance-sensitive species, such as in benthic macroinvertebrates.

These major stressors have been considered under different pressures (presence of ports, aquaculture, urban and industrial discharges, etc.) and some papers have been published for macroalgae (Dromph et al., in prep. (b)), angiosperms (Dromph et al., in prep. (b)) and macrobenthos (Borja et al., 2011).

Definition of reference conditions

During this project, Borja et al. (2012) have stressed the importance of setting targets and reference conditions in assessing marine and transitional ecosystems quality. Hence, this has been a major task for all BQEs and several papers are under preparation.

Evaluate uncertainty on the use of assessment methods

Uncertainty assessment and sensitivity analysis was a major component of all WPs and BQEs, using the data compiled from the centrally organised field exercise, as well as other existing data. These analyses included the assessment of different sources of uncertainty (sampling, processing, natural spatial and temporal variation, calculation of metrics and estimation of response curves), as a basis for the identification of good indicators (sensitive to pressure and high precision). Combined uncertainty analyses were used to assess the risk of misclassification, in particular across the good/moderate boundary. The sources and magnitude of uncertainty were examined to develop guidance on sampling frequency (temporal variability), number of sampling sites (spatial variability) and analytical methods (harmonised versus non-harmonised).

Hence, some papers have been published or are in preparation (Bennet et al., 2011; Dromph et al., in prep. (c); Mascaró et al., in prep).

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Transitional and coastal water management, restoration and the impact of global and climate change

Jacob Carstensen[1], Dorte Krause-Jensen[1], Carlos Duarte[2], Nuria Marba[2], Angel Borja[3] and Michael Elliott[4]

[1] Department of Biosciences, University of Aarhus
[2] Department of Global Change Research, Instituto Mediterráneo de Estudios Avanzados, CSIC-UIB
[3] Marine Research Division, AZTI-Tecnalia
[4] Institute of Estuarine and Coastal Studies, University of Hull

Email address of corresponding author: jac@dmu.dk

Key words: eutrophication, oligotrophication, regime shift, trajectory, typology

Abstract

In his seminal paper, Cloern (2001) introduced three phases of our conceptual understanding of eutrophication in transitional and coastal waters (both referred to as coastal in the following): 1) the early view that responses to eutrophication were expected to be proportional and universal across all coastal ecosystems, 2) the contemporary view recognizing that system-specific attributes modulate responses of coastal ecosystem to eutrophication, and 3) the more recent acknowledgement that multiple pressures, besides eutrophication, affect the various components in the complex mosaic of interacting processes in coastal ecosystems. We will add yet another phase to this complexity, based on more recent studies, suggesting that coastal ecosystems do not necessarily respond in a gradual and predictable manner but may shift suddenly between alternative stable states when critical thresholds are exceeded. We will put results from WISER into this framework and discuss the management implications.

The simple eutrophication models

The first conceptual models for coastal eutrophication (Phase I) were derived from earlier studies of lakes, where significant correlations in phytoplankton biomass to normalized inputs of phosphorus were reported (e.g. Vollenweider 1968). Both the lake and coastal studies were comparing indicators of eutrophication versus nutrient status across many systems, implicitly assuming relationships to have universal character. Most commonly, chlorophyll levels were compared to nutrient concentrations (e.g. Meeuwig et al. 2000), but similar simple relationships were also developed for benthic vegetation (e.g. Nielsen et al. 2002) and benthic macrofauna (e.g. Josefson & Rasmussen 2000). These relationships were further supported by experiments of nutrient additions. The pervasive belief of a uniform response across coastal ecosystems and spatial relationships to be translatable to temporal ones penetrated into many legislative frameworks. It was assumed that ecosystem restoration is a fully reversible process identical to the degradation process, i.e. that oligotrophication can be achieved along the same response trajectory as the eutrophication occurred.

Downscaling ecosystem responses

Following the first phase of eutrophication models it was realized that coastal ecosystems do not necessarily behave similarly, and that there could be system-specific characteristics that would modulate responses to eutrophication and consequently, also to oligotrophication (Phase II). Tidal mixing, retention times, stratification, benthic-pelagic coupling and so forth are important factors governing the sensitivity of coastal ecosystems to eutrophication. This implies that relationships developed under Phase I could be biased by differences among coastal ecosystems. As an example using monitoring data from 35 coastal sites in Denmark, the slope in a generic chlorophyll-TN relationship (log-log scale) was 0.92, whereas it was only 0.53 if a site-specific intercept was included (Carstensen & Henriksen 2009). The bias in the generic chlorophyll-TN relationship was mainly due to differences in the bioavailability of TN across sites. Carstensen et al. (2011) analysed 28 coastal systems from 4 regions and showed a large diversity in relationships (Fig. 1A). System-specific characteristics are, to some extent, included in the Water Framework Directive, which suggests that coastal ecosystems should be organised and compared within typologies, including as a minimum different classes of salinity and tidal mixing. However, these typologies do not constitute distinct categories but rather arbitrary thresholds along a continuum of salinity and tidal mixing regimes. Moreover, several studies have revealed that the proposed typology is inadequate for describing differences between coastal ecosystems.

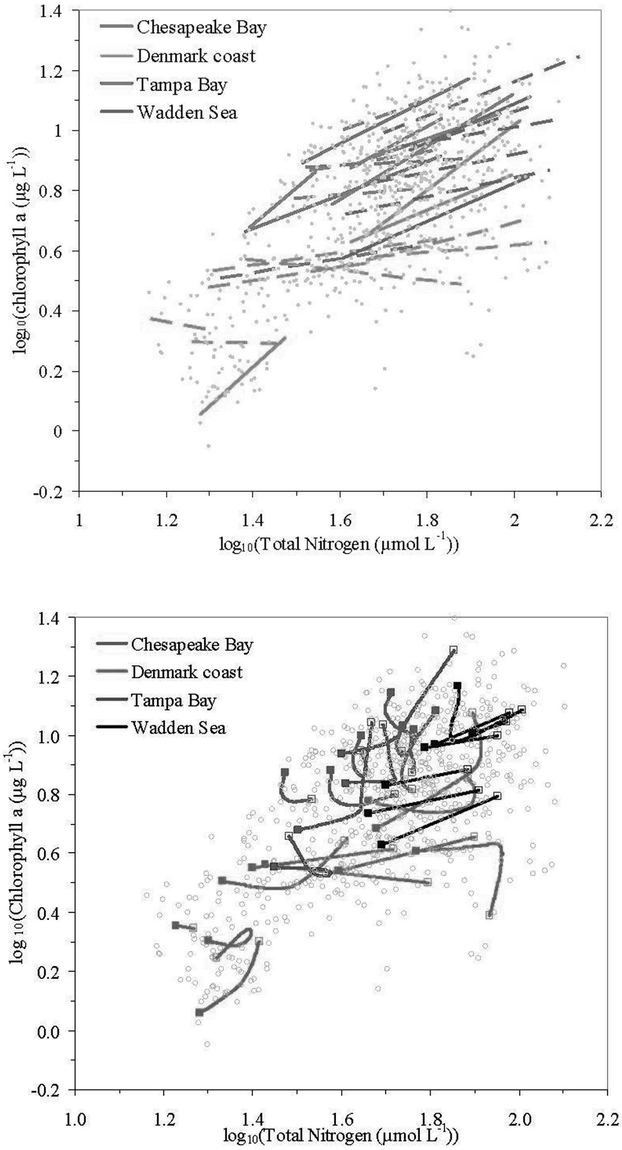

Figure 1: Analysis of chlorophyll-TN relationships for 28 coastal ecosystems by means of static linear regression (A) and time trajectories (B). From Carstensen et al. (2011)

Ecosystem trajectories under multiple stressors

The simpler conceptual models in Phase I and II rested on the assumption that eutrophication was the dominant pressure on the majority of coastal ecosystems. However, within the recent decade the importance of other stressors (Phase III), such as overfishing, climate change and contaminants, have also been acknowledged (Table 1). The consequence of managing just one stressor out of several was that eutrophication was not reversed to the expected status after reducing nutrient inputs (Duarte et al. 2009). As a consequence, trajectories of chlorophyll versus TN over time did not exhibit simple and predictable patterns, but rather convoluted trajectories that for many systems resulted in higher chlorophyll levels despite reductions in TN (Fig. 1B). Similar idiosyncratic trajectories have been observed for secondary eutrophication effects, such as the relationship between chlorophyll and oxygen concentrations in bottom waters (Steckbauer et al. 2011). The apparent lack of recovery, caused by shifting baselines associated with other stressors, has led to frustrations among ecosystem managers. However, it is important to recognise that despite an apparent failure in restoring a historic status for the coastal ecosystem, a reduction in chlorophyll levels has in essence been realised since concentrations of chlorophyll would have been even higher if nutrient levels had not been reduced.

Non-linear responses and regime shifts

Responses of coastal ecosystems in the conceptual models described above (Phase I-III) were assumed to be the same for both eutrophication and oligotrophication processes, given that other stressors were not changed. However, the complex interaction of biological and biogeochemical processes in coastal ecosystems may lead to hysteresis responses, due to significant feed-back processes that can sustain alternative stable states. There are a number of mechanisms that can lead to such important feed-back processes. Hypoxia is a secondary response of eutrophication that alters the benthic community and as a consequence enhances the internal input of nutrients from the sediments (e.g. Conley et al. 2009). Such a positive feed-back of nutrients can maintain the coastal ecosystem in a hypoxic state for extended periods, despite that nutrient inputs from land are reduced, and can explain declines in oxygen concentrations relative to chlorophyll (Steckbauer et al. 2011). On the other hand, recovery can also occur relatively fast, once a well-functioning benthic community has been established (Norkko et al. 2011). Eutrophication reduces the light reaching the bottoms, outshading benthic vegetation. However, an important ecosystem service of the benthic vegetation is the stabilisation of sediments and enhanced sedimentation of particulate matter. Thus, loss of benthic vegetations may sustain a self-enforcing turbid state despite nutrient reductions. The management implication, when regime shifts are probable, is to identify critical thresholds associated with the regime shift and ensure that they are not exceeded. The critical thresholds are likely to change with other stressors affecting the system. If eutrophication has led to a collapse of the coastal ecosystem, nutrient inputs and other stressors on the system must be alleviated to a level allowing the desired regime to be re-established.

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Cyanobacterial responses to phosphorus concentrations and their application to recreational health thresholds

Laurence Carvalho[1,2], Claire McDonald[2], Caridad de Hoyos[3], Ute Mischke[4], Geoff Phillips[5], Gábor Borics[6], Birger Skjelbred[7], Anne Lyche Solheim[7], Giuseppe Morabito[8] & Ana Cristina Cardoso[1]

[1] Institute for Environment & Sustainability, European Commission Joint Research Centre (JRC), Ispra, Italy
[2] Centre for Ecology & Hydrology (CEH), Bush Estate, Penicuik, EH26 0QB, UK
[3] CEDEX, C/ Paseo Bajo de la Virgen del Puerto nº 3; 28005 Madrid, Spain
[4] IGB, D-12489 Berlin, Justus-von-Liebig-Str. 7, Germany
[5] Environment Agency, Kings Meadow Road, Reading, Berkshire, RG1 8DQ, UK
[6] Balaton Limnological Research Institute, Department of Tisza River Research, Klebelsberg Kuno U. 3, Tihany, H-8237, Hungary
[7] NIVA, Gaustadalléen 21, N-0349 Oslo, Norway
[8]CNR, Largo Tonolli 50, 28922 Pallanza (VB)- Italy

Email address of corresponding author: laca@ceh.ac.uk

Key words: algal bloom, blue-green algae, ecosystem services, freshwater, lake, phosphorus, WHO

Abstract

A safe, clean water supply is critical for sustaining many important ecosystem services provided by freshwaters. The development of cyanobacterial blooms in lakes and reservoirs has a major impact on the provision of these services, particularly limiting their use for recreation and water supply for drinking and spray irrigation. Nutrient enrichment and climate change are thought to be the most important pressures responsible for the widespread increase in cyanobacterial blooms in recent decades. Quantifying how nutrients limit cyanobacterial abundance is, therefore, a key need for setting robust targets for the management of freshwaters.

Using a dataset from over 1500 European lakes, we highlight the use of quantile regression modelling for understanding the maximum potential capacity of cyanobacteria in relation to phosphorus and the use of a range of quantile responses, alongside World Health Organisation (WHO) health alert thresholds for recreational waters, for setting robust phosphorus targets for lake management in relation to water use.

The analysis shows that cyanobacteria exhibit a non-linear response to phosphorus with the sharpest increase in cyanobacterial abundance occurring in the TP range from about 20 μg L^{-1} up to about 100 μg L^{-1}. The likelihood of exceeding the WHO 'low health alert' threshold increases from about 5% exceedance at 16 μg L^{-1} to 40% exceedance at 54 μg L^{-1}. About 50% of lakes remain below the WHO low threshold, irrespective of increasing TP concentrations, highlighting the importance of other limiting factors affecting population growth and loss processes, such as high flushing rate.

Developing a more quantitative understanding of the limiting effect of nutrients on cyanobacterial abundance in freshwaters provides important knowledge for restoring and sustaining a safe, clean water supply and can also support mitigation strategies in relation to the less manageable pressure of climate change. Our results can be used to set nutrient targets to sustain recreational services and provide different levels of precaution that can be chosen dependent on the importance of the service provision.

A fish index to assess ecological status of European lakes

S. Caussé[1], M. Gevrey[1], S. Pédron[1], S. Brucet[2], K. Holmgren[4], M. Emmrich[3], J. De Bortoli[1], C. Argillier[1]*

[1] Pôle études et recherches Hydroécologie-Plans d'eau ONEMA – CEMAGREF, Hydrobioly Research Group, 3275 Route de Cézanne, CS 40061. 13182 Aix-en-Provence Cedex 5, France
[2] European Commission, Joint Research Centre, Institute for Environment and Sustainability, 21027 Ispra, Italy
[3] Leibniz-Institute of Freshwater Ecology and Inland Fisheries, 12561 Berlin, Germany
[4] Swedish University of Agricultural Sciences, Department of Aquatic Resources, Institute of Freshwater Research, Stångholmsvägen 2, SE-178 93 Drottningholm, Sweden

* Email address of corresponding author: christine.argillier@cemagref.fr

Key words: lakes, fish multi-metric index, eutrophication, hindcasting

Introduction

The Water Framework Directive requires that each European country assess the ecological status of its lakes by using fish as ecological indicators, but very few have developed fish indices so far. One difficulty often encountered in the development of fish based indices is the heterogeneity of the environmental characteristics of the lakes in comparison with a relatively low number at national scale. The lack of comparable fish data have so far prevented the development of fish based indicators at larger geographical scale.

In this context, as part of the WISER project, a huge work was provided by European countries to assemble fish data from lakes across Europe into a common database. Our objective was then to take advantage of this database to develop a common fish-based index for European lakes exposed to eutrophication pressures.

Table 1: Total number of lakes in each member state used to build the index.

Member State	Number of lakes included in the final dataset
Denmark	*49*
Estonia	*21*
Finland	*89*
France	*40*
Germany	*69*
Ireland	*33*
Italy	*4*
Norway	*1*
Romania	*1*
Slovenia	*2*
Sweden	*143*
United Kingdom	*3*

Dataset

The dataset included 445 natural lakes located in 12 European countries (Tab. 1).

The lakes are mainly distributed in the northern part of Europe. The Central Baltic area was represented by 145 lakes with a very patchy distribution. Only 27 lakes are located in the Alpine area, one lake in the East Continental region and one lake for the Mediterranean part of Europe. Among these lakes, 101 were considered in reference condition or weakly disturbed.

Fish data were obtained in application of the European standard for sampling fish in lakes with multi-mesh gillnet (C.E.N. 2005); Only fish caught by the benthic gillnets were used. Lakes with less than tree species were discarded.

Metrics tested were related to composition and abundance of fish communities. The candidate metrics were selected considering their ecological meaning and values' distribution.

The environmental variables included in the analyses were maximum depth and lake area; altitude; catchment area; geology and mean monthly air temperatures obtained from the climate CRU model (New et al. 2002). Two temperature variables were derived:

(i) Average temperature = (TJanuary+TFebruary+…+TDecember)/12

(ii) Temperature amplitude = TJuly – TJanuary

The percentage of non-natural land cover and total phosphorous concentration in the lake were used as surrogate of nutrient loading in the lakes (proxies to assess eutrophication pressure).

Statistical approach

The development of this index was based on a site specific approach leading to a statistical modelling of the metric responses by using a stepwise multi-linear regression. Moreover, the metrics values at reference conditions were modelled following a hindcasting procedure (Baker et al. 2005, Kilgour and Stanfield 2006). The principle is first to select the best models which contain anthropogenic factors in addition to environmental parameters as predictor variables. Then, these models are used to recalculate fish community metrics under reference conditions by artificially setting the pressures to a value of reference condition. The model output represents the expected metric in that lake in the absence of significant pressures.

The difference between the observed metric values (obs_metric) and the predicted metric values by the hindcasting procedure (hind_metric) corresponds to the deviation between the observed values into disturbed conditions and the predicted values i.e. in case of no disturbance and describe the response range of the metric metric, whatever the natural environmental variability. This deviation score has been expressed as an Ecological Quality Ratio (EQR) (numerical value between 0 and 1).

The complete set of available metrics should then successfully pass through the statistical modelling steps presented above to be selected (Hering et al. 2006):

– a model cross-validated with a r>0.7 between the predicted and the observed values, in order to assess the goodness of fit of the designed model and estimate the generalization performance of the selected model

– an adjusted R^2 of the stepwise model > 0.3

– at least one significant stressor retained in the model

– metric should show a strong correlation to the stressor gradient

Moreover, the observed trend of the selected metric on the pressure gradient should correspond to the expected one when it was known. If metrics showed a strong redundancy (r Spearman>0.8, according to Hering et al. 2006), only the one with the highest correlation to the stressor was selected.

The multi-metric index is finally the average value of core metrics.

Results

A model of three metrics passed the cross-validation step: BPUE (biomass per unit of effort), CPUE (catch per unit of effort) and OMNI (relative number of omnivorous individuals). These metrics were significantly correlated to non-natural land cover (Spearman r: -0.43, -0.44 and -0.40 respectively) and OMNI metric was correlated to the total phosphorus stressor.

The three metrics could be considered as non redundant and were all proposed as core metrics.

The multi-metric index composed of these three metrics was correlated to the stressor percentage of non-natural land cover at catchment scale (Spearman's r: -0.5; P<0.001).

The class boundaries have been defined by fixing the high/good (H/G) boundary and then divided the remaining part into four equal parts. The 25% percentile of the "reference or weakly disturbed" lakes was proposed to define the threshold between the high and the good classes (Fig. 1).

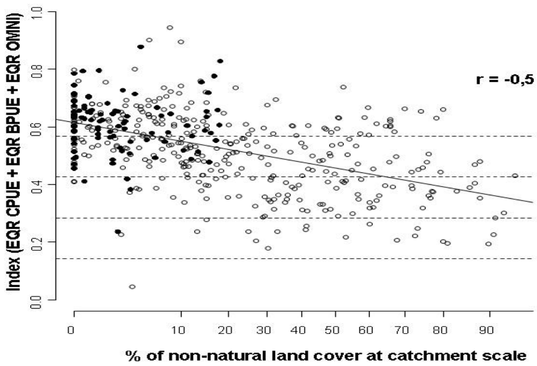

Figure 1: Relationships between the percentage of non natural land-cover (into arcsine square root scale) and the Index (mean of BPUE, CPUE and OMNI). Horizontal lines indicate the class boundaries fixed with the high/good boundary using the 25% percentile of the reference or weakly disturbed lakes (filled dot) and the others below the high-good boundary into four equal width classes.

This value of percentile corresponds to the value of the stressor often used to distinguish reference lakes from non-reference lakes (10 % for the percentage of non-natural land cover) and is in agreement with the annexe IV of guidance on the intercalibration process (European Community 2011). Following this method, the thresholds defined are given in Tab. 2.

Table 2: Class boundaries defined for the European fish index

Classes	Thresholds
H	*> 0.57*
G	*0.57 - 0.43*
M	*0.43 - 0.28*
P	*0.28 - 0.14*
B	*< 0.14*

Discussion and conclusion

Despite the large number of difficulties encountered during this study as the large environmental heterogeneity of the lakes, the differences in data availability from the different countries, the doubts and discussions around the disturbance conditions of these lakes, the lack of details in the fish description, the poor number of environmental and stressor descriptors used…etc, several output has emerged and the result are promising.

First, a huge effort has been done in collecting the European data and the first important output is the compilation of European fish data for lakes that is so far the only existing database at this scale. These data have been used for the development of this index. They permit also to test the relevance of size structure (age) metrics at large scale (Emmrich et al. in preparation) and were used to explore the climatic, biogeographical and anthropogenic factors structuring fish communities at the European scale (Brucet et al. submitted). Applied use can also be expected in the intercalibration groups and will be probably useful for future studies.

Second, in front of discordances in the criteria used to define whether a lake is assigned in reference status or not, an innovative modelling approach has been tested and proposed.

Finally, a European multi-metric fish index for natural lakes has been developed which respond significantly to the anthropogenic stressors. The targeted pressures are eutrophication and "general degradation"; indeed, non natural land cover in the catchment that contributes to explain the selected metrics can be related to different types of pressures (organic matter loading, sedimentation…). Moreover, the index includes, as required by the normative definition of the Water Framework Directive, abundance and composition metrics. This index can be considered as a temporary assessment method of the lakes located in countries where national methods have not yet been developed.

However, among the 1922 lakes with fish data currently listed in the European database, comparable environmental data are available for a quarter of them only. Further data collection, in order to better consider natural variability and pressures, would greatly contribute to improve these results.

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WISERBUGS (WISER Bioassessment Uncertainty Guidance Software) tool for assessing sampling confidence of estimated WFD ecological status class of water bodies

Ralph T. Clarke

School of Applied Sciences, Bournemouth University, Talbot Campus, Poole, Dorset, BH12 5BB, UK

Email address of corresponding author: rclarke@bournemouth.ac.uk

Key words: confidence, uncertainty, WFD, ecological status class, sampling variation, multi-metric indices, multi-metric rules

Overall Purpose

The aim of the software program WISERBUGS is to assist Users in quantifying the effects of (previously-estimated) sampling and other methodological uncertainties on the confidence of estimates of the ecological status of individual water bodies (lakes, rivers stretches, transitional (estuarine) or coastal waters), as required of Member States by the European Water Framework Directive (WFD, 2000).

WISERBUGS software and User Manual was produced by me as WISER Deliverable D6.1.3

The Articles of the WFD (Annex V, section 1.3) require that "Estimates of the level of confidence and precision of the results provided by the monitoring programmes shall be given in the (monitoring) Plan". Thus, water body monitoring and management organisations need to have some understanding and estimates of the confidence to which an individual water body (WB) can be assigned to an ecological status class based on their selected field sampling methods, sample processing protocols and choice of metrics, multiple metric and (optionally) multiple biological quality element (BQE) assessment scheme.

A core part of the WISER project was to collect standardised field sample and survey information on each BQE (phytoplankton, aquatic macrophytes, macroinvertebrates, fish and aquatic habitats) at each of a wide range of lake, transitional and coastal water body sites across Europe. One important reason for this was to improve understanding and provide estimates of the sampling uncertainty (replicate, sub-sample, spatial and temporal) associated with specific sampling/surveying methods, individual metrics, multi-metric indices (MMIs) and multi-metric classification rules. When used with WISERBUGS, this and similar sampling information can help assess which metrics, multi-metric rules and also combination of BQEs provide the most precise (in terms of sampling uncertainty) assessments of water body status class.

User-specified metrics and mult-metrics and mult-BQE Classification rules

WISERBUGS is designed to be as generic as possible. Therefore, the User has almost a completely free choice (and therefore requirement) to specify the:

i. metrics to be used in the water body assessments,

ii. chosen rules for combining metrics into multi-metric indices (MMI) - examples could be the weighted Inter-Calibration common metrics indices (ICMi)

iii. ecological status class limits for individual metrics, EQR or MMI

iv. rules for combining estimated status classes:

- from individual metrics into a higher level class
- from (stressor-specific) metric groups into an overall class using that BQE
- from each BQE into an overall multi-BQE assessment class for the WB (for example using the worst case rule)
- the combination rules allowed are worst case (one-out-all-out), average (rounded up or down) and median (rounded up or down).

WISERBUGS can usefully 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).

Specification of sampling uncertainty estimates for WB metrics

In order for the software to assess confidence of class based on the User-specified metrics and metric combination rules for a list of specific sampled water bodies, the User must input prior estimates of the relevant sampling uncertainty for each estimate of observed metric values or derived EQR for each metric to be involved in the WB assessments. In practice, the estimates of the sampling standard error (SD) for each metric for each water body or site to be assessed within WISERBUGS must be based on best-available information from replicated sampling studies on this or environmentally-similar water bodies, such as as those in the WISER extensive field sampling study. Even where reliable estimates of the sampling uncertainty are not avalable, the User can increase understanding of the consequences for confidnece of class by using a range of trial estimates to represent monitoring sampling schemes of different intensity and thus cost.

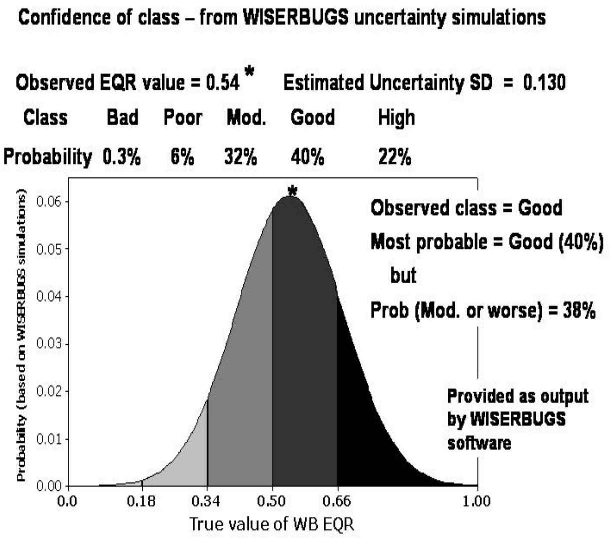

Figure 1: Example of confidence of class estimates from WISERBUGS simulations.

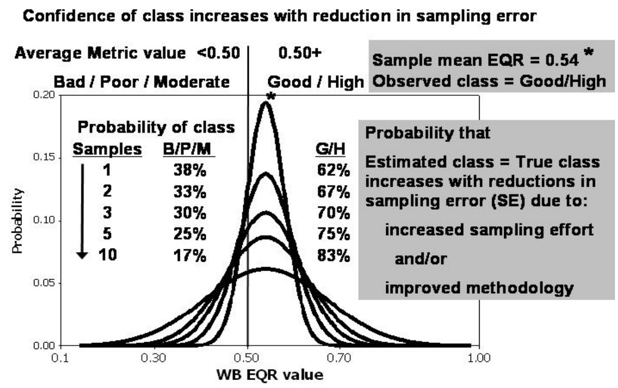

Figure 2: Example showing how confidence of observed class increases with reductions in sampling error.

As an illustrative, (based on WISERBUGS), suppose assessments for a lake are to be based on a single metric EQR for which the status class lower limits are 0.66 (high), 0.50 (good), 0.36 (moderate) and 0.18 (poor). If the lake WB has an estimated observed EQR value of 0.54 based on a single sample and site and the (previously-estmated) sampling uncertainty SD is 0.130, then, although the observed status class is good, with a probability of 40%, there is an estimated 32% chance that the true WB class (based on the average of all possible samples from the lake) is moderate and a 6% probability it is poor (Fig. 1).

However, if the lake mean value of 0.54 was based on more (2,3,5,10) sampling sites around the lake, then our confidence that the true status of this WB is equal to the observed good or better status increases from 62% with one sample to 83% with 10 samples (Fig. 2).

In WISERBUGS, all of the above information on metrics, class limits and rules is supplied by the User in a 'Metric Specification File' in EXCEL format. Detailed help is provided in the WISERBUGS User Manual.

For each set of water bodies to be assessed, the program reads the observed values of each metric or EQR to be used from a User-specified 'Observed metric values' EXCEL file.

The observed metric (or derived EQR) values must have been calculated previously, outside of program WISERBUGS. The layout of this input file (metrics in rows, WB (or samples) in columns was designed to provide immediate compatibility with the metric values EXCEL files derived and output from the freshwater macroinvertebrate sample software known as 'ASTERICS' (obtainable www.eu-star.at).

The ecological status class assessment for individual metrics can be based on:

i. input observed (O) metrics values

ii. input pre-calculated oserved (O) EQR values

iii. EQR values derived within the software from the input (O) values as:

$$EQR = \frac{O - E_0}{E_1 - E_0}$$

where E_1 = Reference Condition value (= value of metric for which EQR = 1), E_0 = anchor value of metric for which EQR = 0 and E_1 and E_0 are supplied by the User (potentially for each WB) in separate input files.

Case (i) and (iii) require uncertainty SD estimates for observed metric values, while case (ii) requires uncertainty SD estimates for pre-calculated EQR. Sampling variability correlations between metrics can also be incorporated to allow for the effect of involving metrics which respond very similarly between samples from the same WB.

Use of simulations to provide estimates of confidence of class

WISERBUGS uses the uncertainty estimates for each metric to simulate a large number of other possible observed metric or observed EQR values which could have been obtained for the same water body using this same sampling monitoring scheme. Non-normally distributed sampling variability of metric values is allowed for by appropriate mathematical transformations in the simualtions.

For each simulation, the same User-specified rules that were used for determining single and combined metric status classes from the observed sample WB data values are applied to the simulated observed metric and EQR values to lead to a status class based on each individual metric and combination of metrics in exactly the same way. The resulting frequency distribution of (typically 10000) simulated values and the probabilities of the derived classes are used to derive 95% confidence limits for metric values and, most importnatly, to provide estimates of the probability of each status class (high, good, moderate, poor and bad) being the true (i.e. complete sampled) class of this WB; based on this user-specficied choice of sampling metrics and assessment rules.

The results are all provided in an EXCEL output file, with one line per WB, each with all of the estimates of confidence of each class and EQR confidence limits, for each level of hierarchical grouping of metrics and BQEs, beginning left-most with the overall assessment for the WB and then in increasing detail, all in standard column format. The User can then easily extract the results for thier own management or publication purposes.

This provides the type of confidence of status information on single, mult-metric and multi-BQE water body assessments required by the WFD.

Disclaimer and Caveat: Assesses precision not accuracy

Uncertainties in estimates of the ecological quality and status class of a site or water body are potentially due to many factors, ranging from the field sampling and sample processing methodology to the choice of high quality sites or metric values to represent the biological Reference Conditions for the site/waterbody.

The approach to assessing 'uncertainty' in program WISERBUGS is simply to estimate the range or variability of estimates of ecological status that could have been obtained using the chosen sampling methods and protocols.

Because the 'true' status class of a site/waterbody is not known, the approach does not try to estimate Type I or Type II errors, but merely to quantify the inherent variability in the methods used to estimate site/waterbody ecological quality. The approach cannot assess whether the metrics used in the bioassessment are good indicators of true ecological quality, but merely whether they give repeaTab. results. External practical experience with using particular metrics or multi-metric assessments systems must be used to judge their usefulness and reliability to detect the range of biological conditions. Thus the program only assesses aspects of 'precision' rather than 'accuracy'.

The error assessment software must, of necessity, be based on the best available estimates of the various sources of variation and errors in observed metric values and EQRs, as provided by the User (from the WISER project or elsewhere). Sources of variation for which no estimates are currently available are ignored in the error assessment program (and effectively treated as zero). In such cases, the software system will over-estimate the precision and under-estimate the true uncertainty in the assessment of status classes. Any User needs to be made aware of these obvious limitations, especially from the point of view of taking catchment management decisions. However, this software system approach provides a good framework for uncertainty assessment and is a major step forward.

The ecological status assessment of transitional waters: an uncertainty analysis for the most commonly used fish metrics in Europe and the French Estuarine and Lagoon multimetric Fish Index (ELFI)

Anne Courrat[1], Mario Lepage[1], Maria C. Alvarez[2], Angel Borja[3], Henrique Cabral[4], Mike Elliott[2], Joao M. Neto[5], Rafael Pérez-Domínguez[2], Violin Raykov[6], Ainhize Uriarte[3]

[1] Irstea, Groupement de Bordeaux, UR EPBX, Ecosystèmes estuariens et Poissons migrateurs amphihalins. 50 av. de Verdun, Gazinet - 33612 CESTAS Cedex, France
[2] Institute of Estuarine and Coastal Studies (IECS) University of Hull. Hull, HU6 7RX, United Kingdom
[3] AZTI - Tecnalia / Marine Research Division. Herrera kaia portualdea z/g. 20110 Pasaia, Spain
[4] Centro de Oceanografia, Faculdade de Ciências da Universidade de Lisboa, Campo Grande, 1749-016 Lisboa, Portugal
[5] IMAR - Institute of Marine Research (C.M.A.). Department of Life Sciences, Faculty of Science and Technology, University of Coimbra 3004 - 517 Coimbra, Portugal
[6] Institute of Oceanology, Bulgarian Academy of Sciences (IOBAS), "Parvi May" Str. 40, P.O. Box 152, Varna 9000, Bulgaria7 Bulgarian Academy of Sciences, Institute of Oceanology, Parvi Mai Str. 40, PO Box 152, 9000 Varna, Bulgaria

Email address of corresponding author: anne.courrat@irstea.fr

Key words: fish metrics, ecological assessment, transitional waters, uncertainty, multimetric index, mixed models, WISERBUGS

Introduction

In Europe, the Water Framework Directive (WFD; Directive 2000/60/EC) aims at reaching good ecological status for surface waterbodies by 2015. Consequently European countries have developed methods based on biological (phytoplankton, macroalgae, angiosperms, macrobenthos and fishes), hydromorphological and physico-chemical quality elements for the assessment and monitoring of rivers, lakes, coastal and transitional waters. In addition to the ecological status class (high, good, moderate, poor or bad) for each waterbody, the WFD requires that "estimates of the level of confidence and precision of the results provided by the monitoring programmes shall be given in the (monitoring) plan". Such estimates are especially important to avoid misclassification of water bodies in their ecological assessment, which could, in extremis, lead to challenges to the final implementation of the Directive. Many factors will affect the final outcome of the assessment exercise, such as sampling design, year(s) of sampling, operator, etc., and so the impact of these factors on the assessment must be known and quantified.

The European Framework project WISER is supporting the implementation of the WFD by testing and complementing existing assessment schemes, with a focus on the effects of uncertainty on classification strength, in order to make existing assessment methods more reliable and more defendable. The present work focuses on fish-based indicators for estuarine and lagoon (transitional waters in the WFD) quality. Previous studies highlighted the potential impact of the sampling design and estuarine natural features on the value of some fish metrics (Courrat et al. 2009; Delpech et al. 2010; Nicolas et al. 2010). However, these studies only relate to some of the metrics and do not focus on quantifying the degree of uncertainty in an assessment scheme based on these metrics. The consequences of the uncertainty at the level of fish metrics on the corresponding Ecological Quality Ratio (EQR) and Ecological Quality Status (EQS) of the water body were not assessed. Hence the present work has four main goals:

- To give an overview of all factors that may affect the value of the most common WFD fish metrics in use for transitional waters and to identify the key sources of variability for these metrics.
- To quantify the effect of these key sources of variability on the individual metrics.
- To test how that variability may impact the final result of a multimetric fish index, both in terms of EQR and EQS
- To indicate the general requirements of a sampling protocol that minimizes uncertainty for the fish-based assessment of transitional waters.

Uncertainty sources are studied and quantified based on a dataset covering several countries and several types of estuaries and lagoons.

Approach and Data Availability

Fish data: This work is based on fish surveys from five datasets, obtained between 2003 and 2010 in 39 estuaries and 14 lagoons distributed accross six countries (Bulgaria, Italy, United Kingdom, France, Spain and Portugal –Table 1 and Figure 1). Three main types of gear were used: beam trawls and seine nets (active gear) and fyke nets (passive gear). Datasets are composed of fishing events. A fishing event is described as a beam trawl haul, a seine haul or a fyke net collection. In total, the datasets contain 3249 fishing events. For each fishing event were recorded some biological data (the number of fish caught from each species), data from the sampling protocol (e.g. trawled area) and some environmental data (e.g. salinity).

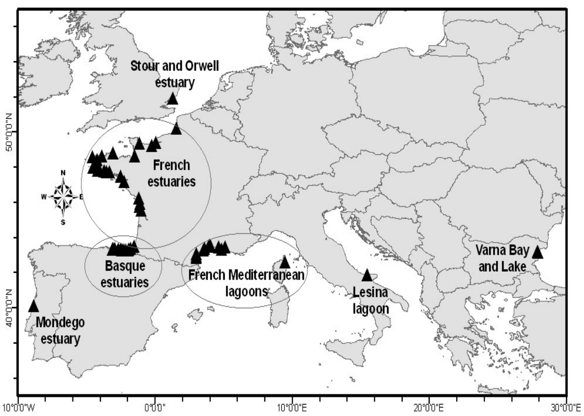

Figure 1: Map of estuaries and lagoons where fish data were available for the following uncertainty analyses.

Pressure data: A pressure index was defined from CORINE Land Cover (CLC - Commission of the European Communities, 1994) 2006, except for the Stour and Orwell estuary where only CLC 2000 was available. The pressure index was calculated including a 2000 m buffer around the estuaries and lagoons.

Data on estuarine features: Data on estuaries (such as estuarine area, continental shelf width, etc.) were also used. These data come from Nicolas et al. (2010) and were completed for the purpose of the present study.

Common species list and functional guilds: A common list of fish species was compiled based on the World Register of Marine Species (WoRMS) database (Appeltans et al. 2011) and a common assignment of "ecological guilds", "position guilds" and "trophic guilds" to fish species was agreed by the experts of the workpackage (Courrat et al. 2011).

General approach: The general approach used in the present work is outlined in Figure 2. Some of the most commonly used WFD fish metrics for transitional waters were selected together with the fish metrics composing the French multimetric Estuary and Lagoon Fish Index ELFI (Delpech et al. 2010; Courrat et al. 2011). All fish metrics were calculated at the fishing event scale to be able to analyse effects of the sampling protocol and to maximize the sample size for the models (Courrat et al. 2009; Delpech et al. 2010). A list of potential sources of variability impacting fish assemblages in estuaries and lagoons was created, based on expert knowledge and literature. Fish metrics and potential sources of variability in fish assemblages were crossed together in order to highlight the key potential

*Table 1: Structure of the datasets used in the present work (dataset description in *Uriarte et Borja 2009; **Martinho et al. 2008; ***Courrat et al. 2009, Drouineau et al. 2012 and ****Courrat et al. 2011).*

Dataset	Data source	Years of sampling	Number of estuaries and lagoons	Number of fishing events	Sampling gears
*Basques estuaries (Spain)**	*Basque Water Agency and AZTI*	*2008, 2009, 2010*	*12 estuaries*	*342*	*Beam trawl*
*Mondego estuary***	*IMAR-CMA*	*2003, 2004*	*1 estuary*	*74*	*Beam trawl*
*French estuaries and lagoons****	*French Water Agencies and Cemagref*	*From 2005 to 2009*	*12 lagoons / 25 estuaries*	*2414 in estuaries / 294 in lagoons*	*Estuaries: beam trawl and fyke net Lagoons: Cemagref fyke net for lagoons*
*Stour and Orwell EA data*****	*Environment Agency (EA)*	*2009*	*1 estuary*	*23*	*Beam trawl, fyke net and seine net*
*Wiser new field data*****	*Wiser WP44*	*2009*	*2 lagoons/ 3 estuaries*	*63 in estuaries / 39 in lagoons*	*Beam trawl, fyke net, Cemagref fyke net for lagoons, seine net*

sources of uncertainty for each of the selected fish metrics. When possible considering available fish data, the uncertainty sources were quantified using either linear mixed models (LMM) or generalized linear mixed models (GLMMs) with estuary or lagoon as a random factor. Models were run separately for estuaries and lagoons and included natural features of these as well as a pressure index based on CORINE Land Cover (Commission of the European Communities 1994). Last, the impact of variability at the fish metric scale on the final result of a multimetric index was tested and analysed using the case study of the French multimetric Estuary and Lagoon Fish Index ELFI. This analysis was carried out using the WISERBUGS (WISER Bioassessment Uncertainty Guidance Software (Clarke 2011)).

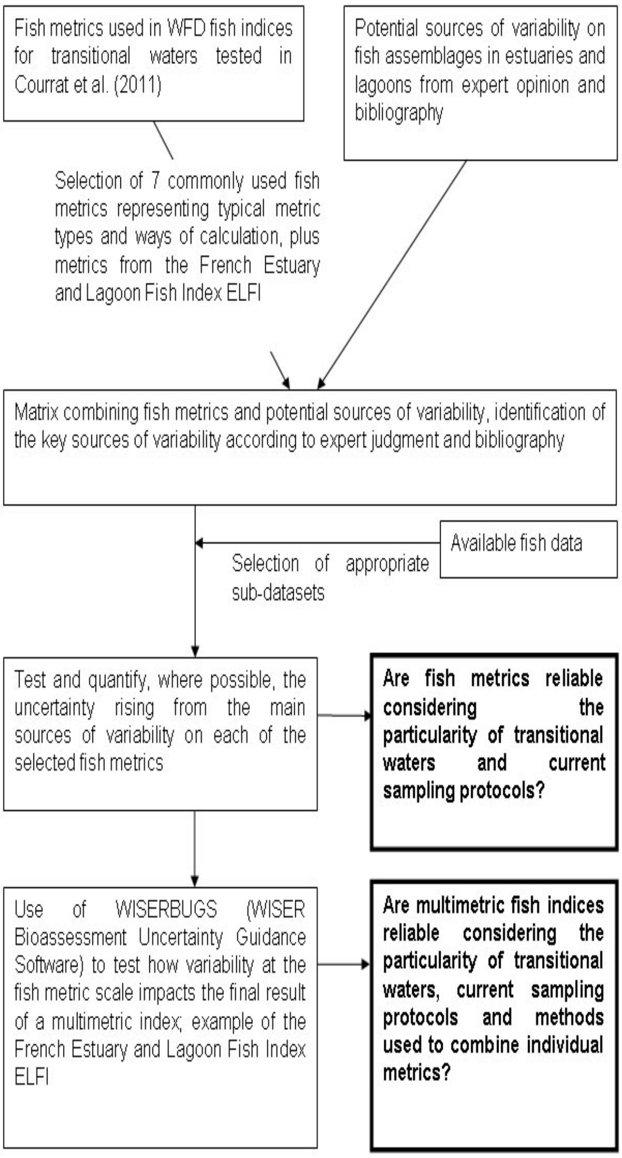

Figure 2: general approach used for the study of uncertainty in fish indices for estuaries and lagoons; bold frames: expected outcomes.

Results and discussion

The key potential sources of uncertainty for fish metrics in transitional waters are given here, together with quantification of this uncertainty when suitable data were available. Considering all potential sources of uncertainty (quantifiable and not), the reliability of common WFD fish metrics for transitional waters is discussed. The suitability of current WFD sampling protocols for fish in transitional waters is also discussed and general requirements for minimizing uncertainty are proposed.

The impact of fish metric uncertainty on the uncertainty of multimetric indices is discussed and the uncertainty assessment of the French fish index ELFI is given as example. Several metric combination rules have been tested to assess how they affect the accuracy of the indices and recommendations to minimize uncertainty at the scale of the multimetric index are provided. The lack of data for a full uncertainty analysis including all key sources of uncertainty is highlighted and a focus is made on which data are requested for similar analyses in the future. The results of our uncertainty analysis highly depend on the fish metrics considered and on the way they are calculated (e.g. per fishing event or averaged per year or season with sampling effort requirements) and on the type of estuaries and lagoons where the fish data has been obtained. The present study gives an overview of potential sources of uncertainty occurring both at the level of fish metrics and at the level of derived multimetric indices. However the authors recommend the inclusion of uncertainty analyses in the general validation of fish indices using data from the area where the index is to be used.

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Sources of uncertainty in assessment of phytoplankton communities

Karsten Mikael Dromph[1], Susana Agusti[2], Alberto Basset[3], Javier Franco[4], Peter Henriksen[1], John Icely[5], Sirpa Lehtinen[6], Snejana Moncheva[7], Marta Revilla[4], Kai Sørensen[8]

[1] Department of Bioscience, Aarhus University, Frederiksborgvej 399, PO box 358, DK-4000, Roskilde, Denmark.
[2] Instituto Mediterráneo de Estudios Avanzados (IMEDEA), Miquel Marqués 21, 07190 Esporles, Islas Baleares, Spain
[3] University of Salento, Dipartimento di Scienze e Tecnologie Biologiche ed Ambientali, 73100 Lecce, Italy
[4] AZTI-Tecnalia, Marine Research Division, Herrera Kaia Portualdea /n, 20110, Pasaia, Spain
[5] CIMA Gambelas, University of Algarve, Faro 8005-139, Portugal
[6] Marine Research Centre, Finnish Environment Institute, PO Box 140, 00251 Helsinki, Finland
[7] Bulgarian Academy of Sciences, Institute of Oceanology, Parvi Mai Str. 40, PO Box 152, 9000 Varna, Bulgaria
[8] Norwegian Institute for Water Research, P.P.Box 173, Kjelsås, 0411 Oslo, Norway

Email address of corresponding author: kmd@dmu.dk

Key words: Phytoplankton, uncertainty, monitoring, pigment analysis, population size

Introduction

The phytoplankton community is an important parameter for assessing the degree of eutrophication of marine habitats. The phytoplankton reacts directly to increasing nutrient concentrations by increasing their growth (Krause-Jensen et al., 2008; Henriksen, 2009). The increased population densities indirectly affects the benthic communities as they lead both to increased light attenuation in the water column and enhanced sedimentation of organic matter, which may lead to increased shading by settling on the vegetation and anoxia through oxygen consumption during decomposition (Krause-Jensen et al., 2008). Phytoplankton populations are assessed by direct identification, the counting of cells in water samples, and indirectly by measuring concentrations of pigments associated with phytoplankton (Paerl et al., 2003). A measurement of any parameter is associated with uncertainty, for the parameters used in monitoring phytoplankton communities, several sources contribute to the total variation of an estimate of the parameter for a particular water-body this includes: variation between stations, variation between samples, variation in processing of samples, and temporal variation (Clarke and Hering, 2006).

The aim of the present study was to assess the variation in pigment concentrations and population densities attributed to water-body, station, and sample levels and in the following processing of samples across water-bodies in Europe.

Materials and Methods

Sampling of parameters for characterisation of the phytoplankton communities, pigments and enumeration of cells, was performed in seven European water-bodies. Within each water-body sampling was carried out at a number of stations (Tab. 1). At each station there was taken between three and seven water samples of which one was split into two subsamples.

Pigment concentrations were measured in water samples filtered onto Whatman GF/F filters, which were immediately frozen and shipped to Denmark on dry ice. Pigments were extracted by submerging filters in methanol, followed by incubation for 24 h in the dark at -20° C. After incubation the filters were sonicated for 15 sec on ice and 1 ml of 0.2 µm filtered extract was diluted with 250 µl water before analysed by HPLC (Wright et al., 1991). Pigments were identified by retention times and absorption then quantified against standards. Chlorophyll a was measured in Helsinki Bay on two integrated samples from each of the stations, covering depth to 0-10m. 100 ml of each sample was filtered on a 47mm GF/G-filter, and two replicates were made. Samples were extracted with 96% ethanol for 24h in the dark before they were analysed using a spectrofluorometer.

Samples for phytoplankton analysis were fixed immediately, and stored cold and in the dark until analysis. Direct counts and measurements of dimensions of phytoplankton were made using an inverted microscope

Table 1: Number of stations and parameters measured for each water-body.

Water-body	Location	No stations	Pigments	Chla A	Cell counts
Helsinki Bay	*South of Finland*	*4*	-	+	+
Lesina Lagoon	*Coast of Puglia region in southeast Italy*	*3*	+	-	+
Mallorca	*North-east coast of Mallorca*	*3*	+	-	+
Mompás-Pasaia	*Coast of Basque region in north Spain*	*3*	-	-	+
Mondego Bay	*North-vest coast of Portugal*	*9*	+	-	-
Oslo Fjord	*South-east coast of Norway*	*3*	+	-	-
Varna Bay	*Bulgarian Black Sea Coast*	*8*	+	-	+

on 50 ml settled fixed samples according to Uthermöhl (1958). In order to assess the variation between taxonomists counting the samples, a second person counted one sample from each station.

The proportion of variation explained by the variance components water-body, sample, sub-sample and for the cell counts taxonomist was estimated by fitting hierarchical mixed effect models. The models contained a fixed intercept while the variance components water-body, station, sample, sub-sample were estimated as random effects with a hierarchical structure where stations were nested within water-body, samples within stations and sub-samples within samples. The models were parameterised using the MIXED procedure in the statistical software package SAS/STAT 9.2 (SAS Institute Inc, 2009).

Results

The variation in pigment concentrations explained by each of the variance components are listed in Tab. 2. The main proportion of the variation between pigment measurements was explained by the variation between stations (10-68% of variation) followed by the variation between water-bodies (6-52% of variation). The diverging picture for the two pigments Diatoxanthin and Neoxanthin was caused by a high variability between samples from stations within one of the water-bodies for each pigment, respectively Varna Bay and Mondego Bay. For chlorophyll a measured in CTD samples more than 90% of the variation was explained at the station level. However, data for this parameter were only available from a single water-body.

Table 2: Estimated percentages of total variance of pigments in a hierarchical mixed effect model due to selected variance omponents.

Pigment	Water Body	Station	Sample	Sub-sample	Residual
Niskin bottle samples					
Alloxanthin	*19.13*	*68.98*	*0.00*	*7.99*	*3.91*
Chlorophyll a	*11.48*	*67.58*	*0.00*	*0.00*	*20.94*
Chlorophyll b	*17.21*	*74.80*	*0.00*	*0.00*	*7.99*
Chlorophyll C1 C2	*11.19*	*66.34*	*0.00*	*0.00*	*22.47*
Diadinoxanthin	*7.66*	*68.48*	*0.00*	*0.00*	*23.86*
Diatoxanthin	*6.44*	*37.70*	*0.00*	*53.04*	*2.82*
Fucoxanthin	*52.58*	*43.65*	*0.00*	*0.00*	*3.77*
Lutein	*10.68*	*48.72*	*0.00*	*0.00*	*40.61*
Neoxanthin	*19.65*	*10.12*	*67.87*	*0.00*	*2.36*
Peridinin	*6.31*	*66.27*	*0.00*	*0.00*	*27.42*
Zeaxanthin	*51.06*	*39.58*	*1.54*	*7.36*	*0.46*
9-But-Fucoxanthin	*0.00*	*60.50*	*0.00*	*0.00*	*39.50*
β-Carotene	*22.90*	*65.01*	*2.29*	*0.00*	*9.79*
CTD samples					
Chlorophyll a	-	*93.87*	*0.33*	*1.38*	*4.43*

Table 3: Estimated percentages of total variance of number of taxa and total cell counts in a hierarchical mixed effect model due to selected variance components.

Variable	Water Body	Station	Sample	Sub-sample	Taxonomis	Residual
Number of taxa	*83.07*	*0.00*	*0.00*	*0.00*	*1.55*	*15.38*
Total density	*10.17*	*0.46*	*0.00*	*0.00*	*34.70*	*54.67*

The variation in number of taxa and total number of cells recorded explained by the variance components are listed in Tab. 3. The main proportion of the variation between numbers of taxa recorded was found to be explained by the variation between water-bodies (83%) followed by taxonomist (2%). For the density of phytoplankton recorded as number of cells l^{-1} the picture was reversed with the main proportion of the variation between densities of cells recorded explained by the variation between the taxonomists counting the samples (55%), followed by the variation between water-bodies (10%).

Discussion

For both sampling methods for pigments the main local variation was found to be at station level. In order to increase the precision of estimates of pigment concentrations for a specific water-body it will therefore have the greatest impact to increase the number of stations, while increasing the number of samples per station will only have a minor effect.

When the phytoplankton community was characterised as number of cells l^{-1} or number of taxa recorded, the main proportion of the variation was at the level of taxonomist for the cell counts and of water-body for number of taxa. The high proportion of variation in the number of taxa recoded at the water-body level is likely to be a reflection of the variation between the communities found at different sites. The high proportion of variation explained by taxonomists is a result of variation in sample processing between staff. Changes in staff are known to potentially cause significant changes in the reported structure of phytoplankton communities, even when the methodologies remain identical (Peperzak, 2010). In order to minimise the variations in sample processing between staff involved in analysing samples as part of monitoring programs, the continuous training and inter-calibration of staff is recommended.

Acknowledgement

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Extraction of data from WISER databases

Bernard Dudley[1], Jannicke Moe[2], Astrid Schmidt-Kloiber[3], Laurence Carvalho[1]

[1] CEH Edinburgh, Bush Estate, Penicuik, Milothian, EH26 0QB, UK
[2] Norwegian Institute for Water Research (NIVA), Section for Climate and Environmental Modelling, Oslo Centre for Interdisciplinary Environmental and Social Research (CIENS), Gaustadalléen 21, NO-0349 OSLO, Norway
[3] BOKU - University of Natural Resources & Life Sciences, Max Emanuel-Straße 17, 1180 Vienna, Austria

Email address of corresponding author: <bedu@ceh.ac.uk>

Introduction

The WISER project has compiled and created a large quantity of data from a wide variety of sources into a common format (see nearby posters – Moe et al. 2011; Schmidt-Kloiber et al. 2011). The bulk of the combined WISER dataset comprises: (i) biological data from diverse organism groups, at various taxonomic levels, using multiple units of measure; (ii) environmental data intended to match the biological data, though often collected at different times and locations to the biological data; and (iii) waterbody-level data. In general, data users in the WISER project required these three types of data to be compiled together, so that matched biological and environmental data could be analysed, usually at the waterbody scale, along with the waterbody-level data. Users' needs in terms of taxonomic and temporal scales were complex and diverse and it was desirable to allow users to extract their own data as and when they needed it, rather than having to wait for someone to perform extractions for them. Therefore a data extraction tool was constructed that was capable of extracting data from all databases using the WISER common structure, and which allowed users many choices in the data selected, and the way data were aggregated in time and space as well as taxonomically. The tool was also developed to produce various biological metrics from the aggregated data.

Methods

The data extraction tool was built from a Microsoft Access database. To utilise data contained in WISER user databases, tables from the user databases were manually linked into the extraction tool database prior to first use of the tool. Forms provided users with a series of choices for the data to be extracted, the level of aggregation required in both time and space, and the calculation of metrics (Figure 1). Users of the tool were able to choose:

– which biological groups to include,
– which environmental determinands to include,
– whether to use data from all months or only from a particular (user defined) season,
– which waterbody-level information to include,
– how to aggregate data in space,
– how to aggregate data in time (two levels of aggregation were allowed so that, for example, annual averages of monthly averages could be calculated),
– what level of taxonomy to use,
– whether to calculate biological metrics.

Available metrics included Shannon's J evenness and taxonomic richness (applied to all groups at the taxonomic scale selected by the user, for any biological group), trophic metrics for both macrophytes and phytoplankton, and two morphological metrics for phytoplankton. Rank and abundance of the ten most abundant taxa in each aggregated 'sample' were also available.

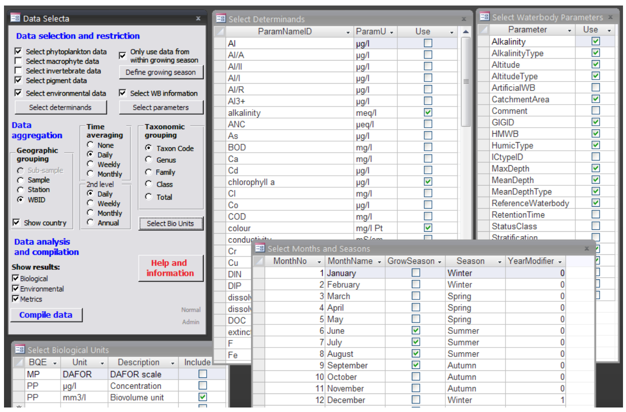

Figure 1: Screenshot of some parts of the Microsoft Access Data Extracta database user interface. The window at the top left of the picture is the main selection panel where options for selection, aggregation, analysis and compilation are provided. The other windows allow selection of environmental parameters, waterbody-level parameters, biological units, and definitions of seasons.

Once the users were satisfied with their selections, a series of interdependent queries were executed (Figure 2) to perform the following steps:

– selection of appropriate data,
– aggregation of data in time, space and taxonomy (generally as averages),
– computation of biological metrics, and
– presentation of extracted data in a crosstab format.

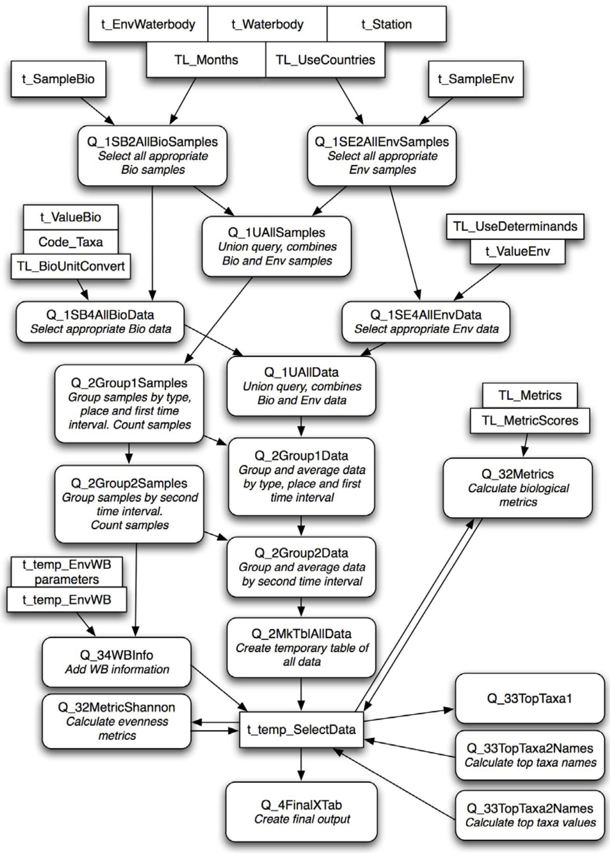

Figure 2: Diagram of the tables and queries used in data selection, aggregation, metric calculation and data presentation. Boxes with squared corners represent tables, boxes with rounded corners represent queries and arrows represent dependencies (A>B means that B requires A).

Results and Discussion

The extraction tool was used by several workpackages, primarily WP3.1 and WP3.2 to extract lake phytoplankton and macrophyte data. An example of the output from the tool is shown in Table 1, where phytoplankton and environmental data were selected at the lake, month and biological class levels, for the growing season only (June to September in this case), and the metrics richness and Shannon's J evenness were calculated on the aggregated data. Multiple peer-reviewed papers are in preparation by WISER project partners, based on the outputs from this tool.

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Table 1: Example of data extraction tool output.

Place	Date	A_AlkalinityType	A_MaxDepth	A_MeanDepth	A_MeanDepthType	A_ReferenceWaterbody	A_SurfaceArea	A_SurfaceAreaType	BPP_Bacillariophyceae_mm3/l	BPP_Chlorophyceae_mm3/l	BPP_Chrysophyceae_mm3/l	BPP_Conjugatophyceae_mm3/l	BPP_Cryptophyceae_mm3/l	BPP_Cyanophyceae_mm3/l	BPP_Dictyochophyceae_mm3/l	BPP_Dinophyceae_mm3/l	BPP_Euglenophyceae_mm3/l	BPP_Klebsormidiophyceae_mm3/l	BPP_Prasinophyceae_mm3/l	BPP_Prymnesiophyceae_mm3/l	BPP_Ulvophyceae_mm3/l	BPP_Unknown_mm3/l	BPP_Xanthophyceae_mm3/l	E_Alkalinity_meq/l	E_chlorophyll a_µg/l	E_colour_mg/l Pt	E_PO4-P_µg/l P	E_total N_mg/l N	E_total P_µg/l P	MPP_PPR_mm3/l	MPP_SHJ_mm3/l
UK_8286_Loch of Spiggie	2007/07	M	12	3.5	S	0	0.97	S	0.125	0.134	0.001	0.016	0.420	0.463	0.002	0.049		0.000		0.001	0.035	0.011		1.07	11.0		8		17.6	12	0.62
UK_8286_Loch of Spiggie	2007/08	M	12	3.5	S	0	0.97	S	0.330	0.179		0.007	0.320	1.425	0.000	0.052	0.003	0.003		0.000	0.010	0.022								12	0.50
UK_8286_Loch of Spiggie	2007/09	M	12	3.5	S	0	0.97	S	0.489	0.125	0.005	0.016	0.080	2.369	0.002	0.024	0.024	0.004		0.026	0.000	0.045		1.12	16.1		8		27.1	13	0.37
UK_8286_Loch of Spiggie	2008/07	M	12	3.5	S	0	0.97	S	0.197	0.358	0.087	0.006	0.181	25.453	0.157	0.007						0.009		1.08	34.6		16		36.2	9	0.10
UK_8286_Loch of Spiggie	2008/09	M	12	3.5	S	0	0.97	S	0.009	0.078	0.028	0.037	0.259	11.220		0.018	0.002	0.001		0.164		0.029		1.18	19.2		19		28.6	11	0.12
UK_8289_Loch of Harray	2007/07	H	4	2.7	VS	-1	9.81	M	0.086	0.091	0.012	0.001	0.385	0.026	0.005	0.024	0.005	0.000	0.003	0.001		0.001		2.18	4.6		8	0.57	19	13	0.51
UK_8289_Loch of Harray	2007/08	H	4	2.7	VS	-1	9.81	M	0.045	0.078	0.008	0.013	0.083	0.031	0.002	0.024	0.003	0.000		0.000		0.004		2.74	2.7		8	0.43	16.7	12	0.74
UK_8289_Loch of Harray	2007/09	H	4	2.7	VS	-1	9.81	M	0.033	0.046	0.050	0.000	0.026	0.009	0.000	0.025	0.006		0.000	0.000		0.005	0.003	3.02	3.6		44	0.54	44	13	0.75

Uncertainty in macrophyte metrics used in calculating the ecological status of lakes

Bernard Dudley[1], Michael Dunbar[2], Ellis Penning[3], Agnieszka Kolada[4], Seppo Hellsten[5]

[1] CEH Edinburgh, Bush Estate, Penicuik, Milothian, EH26 0QB, UK
[2] CEH Wallingford, Benson Lane, Crowmarsh Gifford, Wallingford, Oxfordshire, OX10 8BB, UK
[3] Deltares, P.O. Box 177, 2600 MH Delft, The Netherlands
[4] Institute of Environmental Protection, Department of Freshwater Assessment Methods and Monitoring, Kolektorska 4, 01-692 Warsaw, Poland
[5] Finnish Environment Institute, P.O.Box 413, FI-90014 University of Oulu, Finland

Email address of corresponding author: <bedu@ceh.ac.uk

Introduction

Managers of natural water resources in Europe are required to assess the water quality of lakes under the terms of European legislation adopted in 2000: the Water Framework Directive (WFD). This assessment must be conducted in terms of biological quality elements (BQEs), which include macrophytes (aquatic plants), and which have inherently highly complex and variable distributions. The complex variability in the distribution of these organisms creates uncertainty in the biological quality assessment methods used to assess them. Consequently, creating reliable standard methods for these biological quality elements has been a major challenge for the responsible agencies across Europe since the adoption of the WFD. Generally, assessment methods condense the taxonomic and distributional information gained from macrophyte surveys into metrics, which are usually designed to reflect water quality in terms of the water's biota. This paper uses data collected from customised surveys of more than 30 lakes across Europe to define the uncertainty in several of the metrics that might be used to define the status of macrophytes in lakes. The study demonstrates the complex spatial variability in macrophyte communities, the effect of this variability on the metrics, and the implications to water managers, especially in relation to appropriate survey design. Although the study focuses on the assessment of plants in European lakes, the results have implications for all BQEs in all WFD waterbody types, and indeed for any assessment of the quality of a biological community that uses a metric derived from taxonomic data, anywhere in the world.

Several specific research questions were formulated:

– How does the choice of using presence-absence data or abundance data affect metric results and their uncertainty?
– How does the choice of the species list that is used (i.e. the inclusion or exclusion of helophytic taxa) affect the results of the metric?
– How does surveying 0-1 m depth zone compare to surveying the whole depth range of potentially colonized area?
– How variable are metrics between lake types, between lakes, and between transects within a lake?

A practical aim of this work was to give recommendations on appropriate sampling design and analysis methods that are most likely to reduce uncertainty in the assessment of the status of lake macrophytes. This study does not address the effects of probability of misclassification of water bodies in status classes as common status boundaries have not yet been defined for the metrics used in this study.

Methods

Data were collected from a customised field survey of 28 lakes, which are listed and characterised in Table 1.

A common sampling procedure was devised, based on boat transect methods. Within each selected lake, six localities evenly distributed along a shoreline were identified (the first assigned arbitrarily, and the other five at regular intervals around the shore). Within each locality three parallel transects were surveyed, each being 5m from its neighbour and each starting at the shore and proceeding towards the centre of the lake (Figure 1). Each transect was divided into depth zones of 1 m depth intervals down to the macrophyte colonisation depth limit and in each depth zone five macrophyte sampling sites were used. At each sampling site a single sample was gathered from a rake dragged along the bottom for approximately 2 m, and supplemented by observation through a bathyscope, where this was possible. In each sample all species were identified and their abundance was estimated using a continuous percentage scale.

Table 1: List of lakes surveyed for macrophytes in 2009 for uncertainty analyses. Information is as per Water Framework Directive Intercalibration typology definitions. Where information was not available this is denoted with a dash.

Country	Lake Name	GIG Region	GIG Type	Alkalinity Type	Provisional Status	Eutrophication pressure	Hydromorpho-logical pressure
Germany	Roofensee	CB	LCB1	High	H/G	Low	Low
	Grienericksee	CB	LCB1	High	G/M	Medium	Medium
	Glindower See	CB	LCB1	High	P/B	High	Medium
Denmark	Fussingsø	CB	LCB1	High	-	Medium	-
	Nordborgsø	CB	LCB1	High	-	High	-
Estonia	Saadjärv	CB	LCB1	High	H/G	Low	Low
	Viljandi	CB	LCB1	High	G/M	Low	Medium
Poland	Kiełpińskie	CB	LCB1	High	G	Medium	Low
	Rumian	CB	LCB1	High	M	Medium	Low
	Lidzbarskie	CB	LCB1	High	P/B	High	Low
United Kingdom	Rostherne Mere	CB	LCB1	High	P/B	High	Low
	Loweswater	N	LN2a	Medium	M	Medium	Low
	Grasmere	N	LN2a	Medium	M	Medium	Low
Finland	Sääksjärvi	N	LN1	Medium	G/M	Low	Medium
	Vuojärvi	N	LN1	Medium	M/P	High	Medium
	Iso-Jurvo	N	LN2a	Low	H/G	Low	Medium
Norway	Nøklevann	N	LN2a	Low	H	Low	Low
	Longumvatnet	N	LN2a	Medium	G/M	Medium	Low
	Temse	N	LN2a	Medium	M	Medium	Low
Sweden	Västra Solsjön	N	LN2a	Low	H	Low	Low
	Fiolen	N	LN2a	Low	M	Medium	Low
	Skirösjön	N	LN1	Medium	P	High	Medium
France	Aulnes (étang des)	Med	L-M1	High	M	High	Low
	Salagou (lac du)	Med	-	High	M	Medium	Medium
Italy	Segrino	Med	AL5	High	H/G	Low	Medium
	Lago di Monate	Med	AL5	Medium	G	Medium	Medium
	Candia	Med	AL5	Medium	G/M	Medium	Low
	Alserio	Med	AL5	High	M/P	High	Low

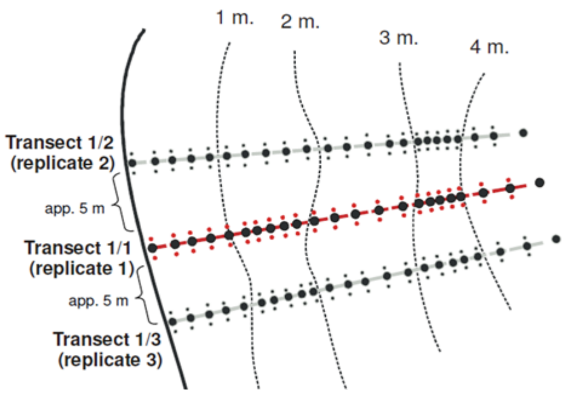

Fig. 1: Diagram of sampling design used in the WISER common field sampling protocol in 2009.

The WISER lake macrophyte data were used to examine variability associated with the varying levels of the hierarchical sampling scheme: transects within stations within waterbodies within countries. We assessed this for several response metrics (Kolada et al. 2011), including:

– Intercalibration Metric for lake macrophytes (ICM-LM),
– modified Ellenberg-N score,
– maximum growing depth (C_max),
– species richness.

Each metric was calculated for each transect. We examined correlation between metrics at the various levels in the sampling hierarchy, and correlations between explanatory variables. Uncertainty analyses were conducted using linear mixed effects models, fitted using the nlme package in the R environment for statistical computing. The levels of the sampling hierarchy were specified as nested random effects, with the lowest level, variation between transects, forming the residual. We used TP and alkalinity, measured at the lake level, as explanatory variables in all analyses because of their strong relationship to the metrics studied.

Results and Discussion

How does the choice of using presence-absence data or abundance data affect the metric results and their uncertainty?

Compared to presence/absence ICM-LM, the abundance weighted ICM-LM resulted in a steeper (0.46 vs 0.38) but slightly less precise (standard error of 0.30 vs 0.28) response to TP, while response to alkalinity was very similar. Presence/absence ICM-LM showed greater variance at waterbody scale than weighted ICM; variances were progressively similar at station and transect level. This supports the use of abundance weighted averages as they provide a stronger relationship with the nutrient pressure (TP), but this result contradicts previous work in the REBECCA project (Penning et al. 2008), where weaker relationships between pressure and metric were obtained when using abundance data. Unlike in WISER, however, the REBECCA analyses were performed on harmonised data obtained by various research groups, using disparate survey methodologies. We recommend therefore, that abundance data should be used in calculating macrophyte metrics, but only in cases where all data has been collected using a common methodology, as was done in WISER.

How does the choice of the species list that is used affect the results of the metric?

Metrics based on submerged taxa only showed a stronger relationship with the pressure variable TP, and are also more closely related to alkalinity. Helophytes are less affected by water quality as their environment is not sub-aquatic, and their response to eutrophication is obscured by soil trophic characteristics, exposure, shoreline management and especially water level fluctuation dynamics as noted in several studies (e.g. Coops et al. 1994). Although we would expect that the use of data from the WISER intensive field campaign should provide a strong basis to answer this question, it should be noted that the current survey design may not have been appropriate to properly represent the high variability found in shoreline plant communities.

How does surveying 0-1 m depth zone compare to surveying the whole depth range of potentially colonized area?

ICM-LM scores for deeper water were lower than for shallower water (intercept at 4.51 vs 5.05) indicating that species in the shallow zone are more often representing higher trophic status. Conversely, large perennial isoetids, for example, which prefer deeper waters, are indicators of low trophic status. If an assessment method uses only shore-based data (obtained by wading), it is likely to result in an assessment of condition that is worse and less precise than if the method used data from deep water as well (obtained by boat). It was also apparent that metrics calculated on data from deeper samples were more responsive to changes in the pressure (TP). Therefore it seems to be important to include the deeper sites in the survey to get a more precise response to a TP pressure gradient.

How variable are metrics between lake types, between lakes, within a lake, and between transects?

Table 2 compares the relative proportion of variance in the selected metrics at each level of the sampling hierarchy, and summarised this variance as proportions between and within waterbodies. Results are presented per metric for models with and without TP and alkalinity as explanatory variables. Not surprisingly, both ICM-LM and Ellenberg show similar behaviour, with 70-75% of variance in the metric occurring between waterbodies and countries in the null model (without explanatory driver variables). Including TP and alkalinity in the models reduces this variance to 40-50%. In these latter models, ICM-LM, compared to Ellenberg, illustrates a slightly higher proportion of variance between waterbodies + countries, with correspondingly less variance within waterbodies. Maximum growing depth also behaves similarly to ICM-LM, although the covariates appear slightly more successful in explaining between-waterbody variance. The richness metric follows a completely different behaviour; introduction of the covariates reduces the variance between waterbodies but accentuates the variance between countries and total between waterbody variance remains roughly constant.

The results illustrate that differences in the number of transects for which metrics may be calculated can have a strong influence on the results. In particular, as TP levels increase, richness decreases, but numbers of taxa for which metrics such as ICM-LM can be calculated decrease even more rapidly. It is a general rule in statistical modelling that the data points at the extremes of the explanatory variables have most influence on the response relationships. Increased imprecision of metrics associated with low richness of indicator taxa, and at the most extreme, non-calculability of such indices can have a significant influence on perceived metric performance. Therefore, to maintain the same degree of uncertainty, more sampling is required at either end of the trophic scale, when there is less vegetation to be sampled.

Table 2: Proportions of variance at different levels of the sampling strategy for four different metrics and two formulations of the model: with and without TP/alkalinity.

Metric	Model	Country	Waterbody	Station	Transect	Total Between	Total Within
ICM-LM (weighted)	*Null*	*0.11*	*0.61*	*0.19*	*0.08*	*0.72*	*0.28*
	TP + Alk	*0.00*	*0.47*	*0.37*	*0.16*	*0.47*	*0.53*
Ellenberg	*Null*	*0.31*	*0.42*	*0.18*	*0.08*	*0.74*	*0.26*
	TP + Alk	*0.00*	*0.41*	*0.40*	*0.19*	*0.41*	*0.59*
Max growing depth C_max	*Null*	*0.39*	*0.31*	*0.21*	*0.08*	*0.70*	*0.30*
	TP + Alk	*0.01*	*0.38*	*0.44*	*0.17*	*0.39*	*0.61*
Richness	*Null*	*0.18*	*0.19*	*0.45*	*0.18*	*0.37*	*0.63*
	TP + Alk	*0.28*	*0.10*	*0.44*	*0.18*	*0.38*	*0.62*

Acknowledgement

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River management, restoration and the impact of global and climate change

Christian K. Feld[1], Veronica Dahm, Daniel Hering, Helga Kremser, Maxime Logez, Armin W. Lorenz, Anahita Marzin, Andreas Melcher, Didier Pont, Piet F.M. Verdonschot and Hanneke E. Keizer-Vlek

[1] University of Duisburg-Essen, Department of Applied Zoology / Hydrobiology

Email address of corresponding author: christian.feld@uni-due.de

Key words: River Basin Management, streams, rivers, land use, hydromorphology, water temperature, connectivity, recovery, fish, benthic invertebrates, macrophytes, benthic diatoms

Background

Although the detection and assessment of environmental impacts have been subject to river assessment and monitoring for more than three decades, this field was particularly boosted through the WFD in 2000. Since then, scientists have spent tremendous efforts on the development of new assessment systems, capable of monitoring the implications of multiple environmental impacts that threaten the riverine flora and fauna. These efforts resulted in a considerable improvement of biological assessment as a reliable ground for integrated monitoring schemes based on the conditions of aquatic assemblages: fish, benthic invertebrates, macrophytes and benthic algae. With climate change, however, yet another stressor came into focus that is likely to influence (and maybe confound) river assessment and management, for instance through changing temperature and precipitation/discharge regimes.

Despite the achievements in understanding the multiple pathways of stressor impacts (i.e. the biota's response to degradation), the opposite pathway (i.e. biota's recovery following restoration and management) continues to be poorly understood. Since the early 1990s, restoration ecology seeks to understand the various effects of evenly various measures of restoration and mitigation on the riverine biota. Yet, even after two decades of restoration science, the mechanisms that control ecological recovery after restoration often remain hypothetical. Scientists and practitioners in river management frequently face a situation where profound improvements, for instance, of the riverine habitat structure do not show the desired improvements of aquatic assemblages. Alike ecological quality assessment, river restoration too is likely to be affected by climate change and requires adaptive strategies to account for its potential impacts on recovery.

Objectives and approach

WISER addressed these knowledge gaps and sought to highlight the response of aquatic assemblages to both degradation and restoration. The survey along degradation pathways made use of nearly 4,350 monitoring samples from Central and Western Europe and including the four organism groups. This analysis focussed on the comparison of individual assemblage's response to environmental stress (water quality, hydrology, morphology, land use). In contrast, due to the general lack of restoration monitoring data, restoration pathways were primarily derived from the restoration literature. A comprehensive review of nearly 160 peer-reviewed publications provided insight in the effects – and non-effects – of hydrological and morphological restoration measures, such as riparian buffer instalment or weir removal. In addition, a set of 40 German restoration measures was used to empirically identify ecological changes after restoration, partly also in light of possible adverse effects by stressors upstream in the catchment (e.g. land use, structural modification).

Using regression models, the potential impact of global warming was exemplarily forecasted for a selection of cold water- and warm water-adapted fish species in France. The forecasted changes in species distributions were subsequently used to estimate the effects of species turnover due to climate change on the reference conditions as defined for fish-based assessment. In addition, time series data from two case study catchments in Austria (River and Lake Traun) and France (River Seine) provided an empirical basis for the detection of long-term trends in aquatic community compositions due to changing climate factors.

The individual results were combined in order to provide practical guidance on ecologically effective river restoration and management in light of climate change.

Results

Stressors act hierarchically and at different spatial scales

Comparative analysis of individual stressor's relationship to individual BQEs revealed a dominant role of catchment and stretch-scale impacts (land use, eutrophication) over local hydrological and morphological habitat conditions (barriers up-/downstream, bed and bank structure) (Table 1). Except for macrophytes, all BQEs revealed a strong, yet more or less sensitive response to agriculture (Table 2). In concert with previous research, the results support the dominant role of agriculture in the catchment above a site.

Fish and invertebrates (data from France and Germany) were found to strongly respond to catchment and stretch-scale agriculture in mountain streams and rivers (Figure 1), while the strongest correlations were found with land use patterns in near-stream buffer strips (100 m width, 5–10 km length). The trends were less obvious for lowland systems, yet support the potential role of near-stream riparian areas in buffering and mitigating agricultural impacts.

BQEs respond differently to individual stressors

Not only does the response intensity (i.e. the strength of the relationship) vary among BQEs, but also its sensitivity (i.e. the stress level at which response can be detected). The comparison of both components (French monitoring data) revealed a general response of fish, invertebrates and diatoms to mixed stressor impacts at all spatial scales (i.e. general degradation) and water quality degradation (Table 2). While fishes also revealed a comparatively strong response to other stressors, this was less pronounced for the other BQEs. Comparatively weak and insensitive responses were found to hydrological and morphological degradation, probably due to the dominant role of large-scale effects of land use and eutrophication, which may have superimposed other stressor's effects.

Restoration and recovery are controlled by upstream impacts

The spatial hierarchy of stressors is rarely addressed by restoration schemes; a review of 160 restoration studies in North America, Europe, Australia and New Zealand revealed that the majority of restoration measures were implemented at stretches less than 1 km of length (Feld et al. 2011). Such local measures are prone to landscape and land use impacts in the catchment above a restored

Table 1: Ranking (hierarchy) of sub-/catchment-scale (marked grey) and local/reach/riparian-scale impacts on fishes, invertebrates and diatoms.

Rank order	Fish	Benthic Invertebrates	Benthic Diatoms
1	*Catchment arable*	*Catchment arable*	*Catchment arable*
2	*Eutrophication*	*Eutrophication*	*Alkalisation*
3	*Catchment urban/fabric*	*Buffer arable*	*Eutrophication*
4	*Buffer agriculture*	*Alkalisation*	*Habitat structure*
5	*Barriers up-/downstream*	*Catchment heterogeneous agriculture*	*Barrier upstream*
6	*Habitat structure*	*Buffer urban fabric*	*Barrier downstream*
7	*Catchment heterogeneous agriculture*	*Buffer heterogeneous agriculture*	*Catchment heterogeneous agriculture*
8	*Buffer urban and heterogeneous agriculture*	*Habitat structure*	*Riparian vegetation modified*
9	*Alkalisation*	*Catchment urban fabric*	*Buffer heterogeneous agriculture*
10	*Buffer urban fabric*	*Riparian vegetation modified*	*Buffer urban fabric*
11	*Riparian vegetation modified*	*Barrier downstream*	*Catchment urban fabric*
12	*Barrier upstream*	*Barrier upstream*	*Buffer arable*

Table 2: Intensity (strength) and sensitivity of BQE's response to different stressors. See text for explanations.

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	*Intensity*	*high*	*medium*	*low*	*medium*	*medium*
Invertebrates	*Sensitivity*	*high*	*medium*	*low*	*low*	*medium*
Fish	*Intensity*	*high*	*high*	*medium*	*high*	*high*
	Sensitivity	*medium*	*medium*	*medium*	*medium*	*low*

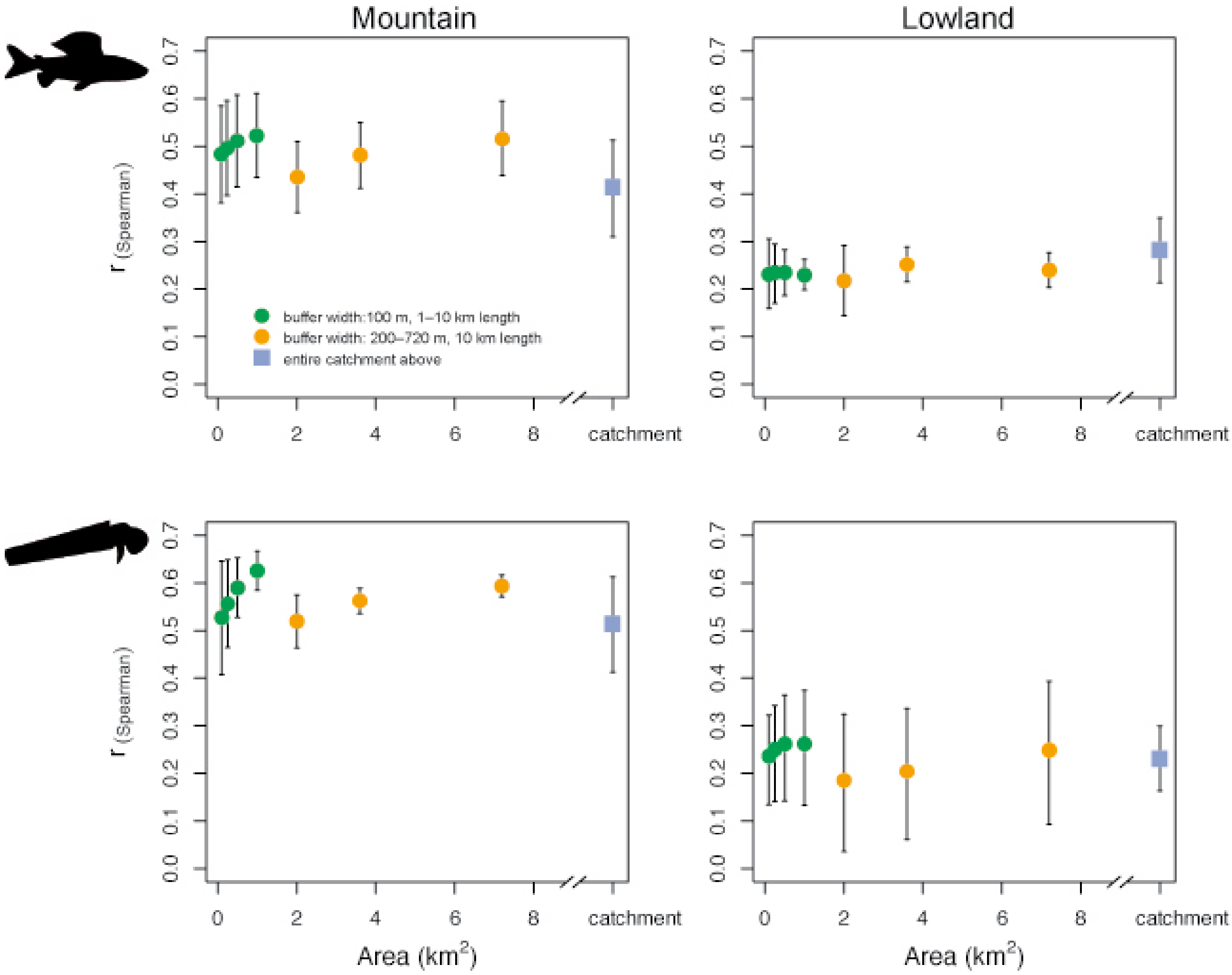

Figure 1: Correlation between the proportion of pollution intolerant fish (upper row), the number of Ephemeroptera-Plecoptera-Trichoptera taxa (benthic invertebrates, lower row) and % area as agriculture or forest (absolute mean ± SD) in different buffers upstream. The analysis was based on 500 sampling stations in France and Germany..

Table 3: Spearman rank correlations of EQRs (acc. to the German National Monitoring Standards) and physical habitat quality upstream of the sampling sites (buffer length). Significance levels refer to differences in correlation coefficients (N = number of sites; significant correlations in bold).

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

site or stretch. Hence, local habitat restoration is often found to be spoiled by fine sediment entries from crop agriculture above (Allan 2004). Fertilizer and pesticide applications above may also affect site-scale water quality (Allan 2004). Flash floods, induced by a high degree of impervious areas in urban settlements above may affect local hydromorphology (Paul and Meyer 2001).

Consquently, local restoration is unlikely to initiate recovery unless the multifaceted impacts in the catchment above (and with regard to migration barriers also below) are too addressed by restoration. The survey of nearly 40 pairs of unrestored (control) and restored (impact) sites in Germany supported the adverse impacts from land use and hydromorphological degradation above the restoration. The BQE's response to the physical habitat quality above the sites is shown in Table 3 for control and impact sites. While fishes and macrophytes were significantly correlated with habitat degradation up to 10 km above, invertebrates responded to upstream habitat conditions up to 2.5 km only.

Climate change affects the distribution of cold water-adapted fish

The use of four climatic scenarios to project the distribution of 23 widespread fish species (French monitoring data) using species distribution models (Logez et al., accepted) revealed important changes in species distribution due to global warming. Numerous local populations, for instance, of brown trout and grayling are predicted to go extinct as early as 2050–2060 (Figure 2) due to the projected increases in water temperature. This pattern is in line with the findings of Buisson et al. 2008 and supported by the long-term study conducted on a section of the Traun River in Austria. Between 1976 and 2008, the water temperature of River Traun increased by 2.2°C on average, which lead to a sharp decrease in grayling abundance. The species has almost gone extinct until present in the heated river section. The projections, however, also revealed a spatial expansion of the distribution of warm water-adapted species.

Climate change affects the functional structure of fish assemblages

Not only assemblage composition may change due to climate change, but also the functional structure of fish assemblages. To test this hypothesis, the variability of eight (functional) traits was modelled using relevant environmental factors. Observed trait values for current environmental conditions were then compared with expected values based on forecasted climatic conditions. On average, the values of metric scores increased, which implies the decrease of the relevant metric's representation in fish assemblages in the future (Table 4). This is especially marked for the two metrics based on species intolerances: intolerance to low oxygen concentration and to habitat degradation. The results imply the shift of reference conditions for some traits and thus may influence the assessment of ecological conditions using fish.

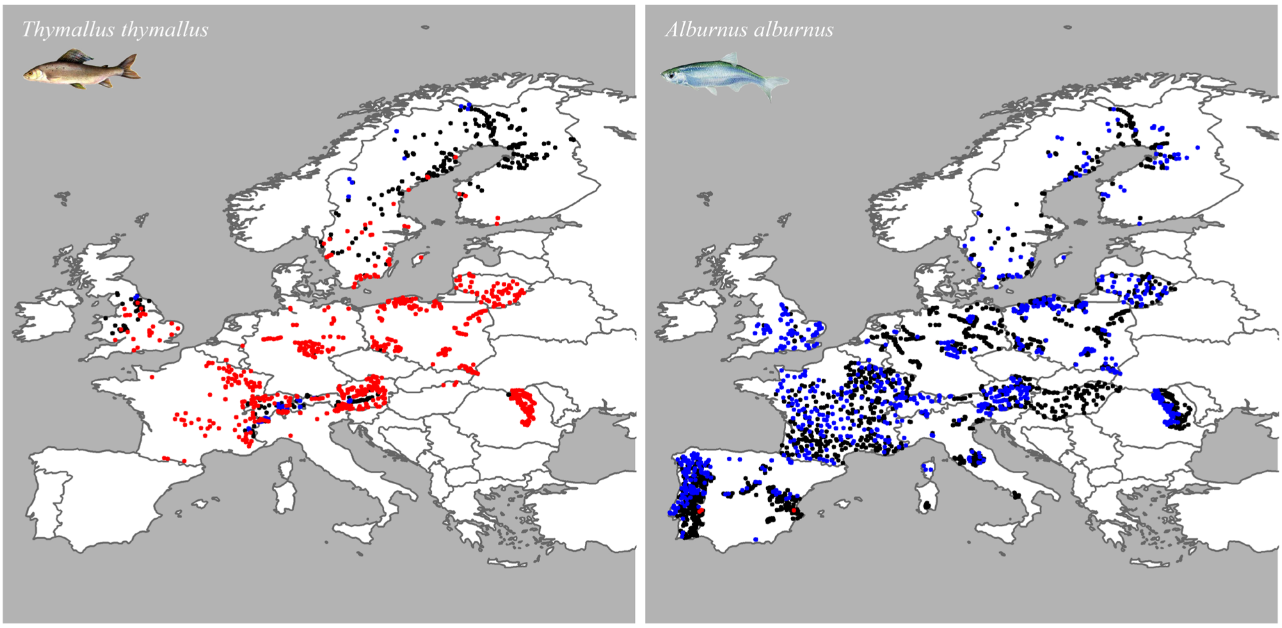

Figure 2: Projected distributions of grayling (left plot) and bleak (right plot) in the period 2050–2060. Black dots represent unchanged suitable conditions (compared to current climatic conditions), blue dots represent location with climatic conditions becoming suitable, and red dots location with climatic conditions becoming unsuitable.

Tab. 4 : Average deviation of metric scores based on the comparison of trait values expected under current environmental conditions and projected climatic conditions.

Metrics (in number of species)	2020–2030	2050–2060
Local richness	*0.150*	*0.321*
Intolerant to low oxygen concentration	*0.335*	*0.590*
Intolerant to habitat degradation	*0.302*	*0.536*
Eurytopic species	*0.039*	*0.112*
Rheophilic species	*0.232*	*0.468*
Species without spawning references	*0.093*	*0.175*
Spawning in running waters	*0.192*	*0.388*
Lithophilic species	*0.141*	*0.325*

Predicting ecological implications of climate change is uncertain

The uncertainties associated with the projections of future species' distribution are often considerably high and sometimes unacceptable. Uncertainty blurred the observed patterns and our predictive ability. Therefore, these results in general require caution in order to reliably inform water managers and restoration practitioners about the effects of climate change.

Implications

Although biological assessment and monitoring is usually implemented at the site scale, i.e. biological samples are taken at the scale of several tens (e.g. diatoms) to hundreds (e.g. fish) of metres length, the aim is to reliably assess the ecological quality at the scale of entire water bodies, i.e. at the scale of several to tens of kilometres of length or even more. Notably, the BQEs seem to meet this demand and show strong and reliable responses to broad-scale stressors such as agriculture or widespread physical habitat modifications. These stressors often act at the scale of entire water bodies.

Consequently, in order to initiate and foster recovery, and eventually to achieve the good ecological status, river restoration and management must address such broad-scale stressors and their impacts, respectively. There is evidence that restoration can lead to recovery if all relevant stressors are being addressed, but there's also tremendous evidence for unsuccessful restoration, where the individual measures implemented at a small site or short reach did not at all address the relevant stressors and their impacts appropriately. Inevitably, this means that River Basin Management must address and mitigate the multifaceted impacts of agricultural and other land uses, such as eutrophication, pollution or fine sediment loads. There is sufficient evidence that forested riparian buffer strips can effectively retain (buffer) nutrient and fine sediment entries from crop fields (reviewed by Feld et al. 2011). Yet, to present riparian buffer instalment or other mitigation measures are hardly considered in river restoration, probably because of the potentially conflicting socio-economic interests of land users. Meeting the goal of the WFD, however, is doubtful if the broad-scale impacts remain unconsidered in River Basin Management.

Moreover, water managers and restoration practitioners must account for the potential effects of global warming, for instance, when planning measures to stock or reintroduce fish species. Both species distribution models and long-term studies imply a loss of temperature-sensitive fish species due to global warming. This is likely to affect the reference conditions, which are set as assessment baselines (high ecological status), and hence will also affect ecological assessment, classification and intercalibration.

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A detailed review of biological and abiotic effects, based on ca. 160 restoration references, is available through Feld et al. (2011) and WISER Deliverable 5.1-1 (http://www.wiser.eu/results/deliverables/#D511). The conceptual models of restoration-recovery chains have been made available interactively at (http://www.wiser.eu/results/conceptual-models/). The empirical analysis of degradation and restoration pathways is available through the WISER Deliverable 5.1-2 (http://www.wiser.eu/results/deliverables/#D512). The descriptive and predictive analysis of the impacts of Climate Change is available through the WISER Deliverable 5.1-3 (http://www.wiser.eu/results/deliverables/#D513). The key messages and implications have been summarised at (http://www.wiser.eu/key-messages/).

Trends in *Posidonia oceanica* population growth rates and the plant sulphur isotopic signatures (δ34S) and sulphide intrusion

Rosa García, Núria Marbà, Carlos Duarte

Department of Global Change Research, IMEDEA (CSIC-UIB), Institut Mediterrani d'Estudis Avançats, Miquel Marquès 21, 07190 Esporles (Balearic Islands), Spain

Email address of corresponding author: rgarcia@imedea.uib-csic.es

Abstract

Coastal eutrophication has been identified among the major causes of seagrass loss, which occurs at an average global rate of 2-5% yr-1. Seagrasses are very sensitive to elevated sulphide concentrations in sediment porewaters that rise from eutrophication. The intrusion of sulphides into seagrass tissues has triggered sudden die-off events observed during the past decades in both temperate and tropical seagrass meadows. There is evidence that porewater sulphide concentration exceeding 10 μM accelerates the decline of the Mediterranean seagrass species *Posidonia oceanica* growing in carbonate rich sediments as those in the Balearic Islands (Spanish Mediterranean). Sulphide intrusion in seagrass tissues is imprinted in their sulphur isotopic signatures ($\delta^{34}S$). The proportion of total sulphur in the plant from sulphide origin (i.e. sulphide intrusion, Fsulphide) can be estimated from the $\delta^{34}S$ signatures in plant tissue, in the water column and in sediments. Annual *P. oceanica* population growth rate can be estimated from repeated shoot census in permanent plots as the balance between annual shoot recruitment and mortality rates. We examine the relationship between the $\delta^{34}S$ signatures in *P. oceanica* leaves and the plant net population growth rate across 11 meadows in years 2004, 2005, 2006, 2009 and 2010, and identify the critical leaf $\delta^{34}S$ signatures to trigger seagrass decline. Moreover, we examine the trajectories of *P. oceanica* decline along leaf $\delta^{34}S$ signatures over an 8-years period to assess the temporal responses of impact and recovery of seagrass populations to sulphide intrusion. The role of seawater summer temperature on the temporal variability of $\delta^{34}S$ and sulfide intrusion is also examined. The results demonstrate that along the Balearic Islands, $\delta^{34}S$ in *P. oceanica* leaves below 19-20‰ reflects sulfide intrusion across the leaf meristems and triggers shoot population decline.

Automatic techniques for phytoplankton abundance and size-structure characterization

Maialen Garmendia, Marta Revilla* and Lucía Zarauz

AZTI-Tecnalia; Marine Research Division; Herrera Kaia Portualdea s/n; 20110 Pasaia, Spain

* Email address of corresponding author: mrevilla@azti.es

Key words: FlowCAM, marine waters, Water Framework Directive, eutrophication, phytoplankton indicators

Introduction

Indicators based upon phytoplankton biomass, abundance, composition and blooms are needed for assessing the ecological status of the European marine coastal and transitional waters (European Commission, 2000).

Although many field studies report changes in taxonomic composition and abundance in nutrient-enriched environments, the phytoplankton response to anthropogenic inputs is not exclusive of a particular species or group (Smayda, 2004). In addition, taxonomy-based indicators have been criticized due to the risk of misclassification of small specimens and cryptic species, and because they are time-consuming and require a high degree of expertise.

Alternatively, the use of body-size distribution has been proposed (Mouillot et al., 2006). Phytoplankton size is an indicator of food quantity and quality for grazers, and can be related to habitat conditions and eutrophication (Turner, 2001; Capriulo et al., 2002). Also, recent studies suggest the incorporation of descriptors of phytoplankton size-structure in environmental monitoring programs (e.g. Buchanan et al., 2005; Sabetta et al., 2005; Lacouture et al., 2006; Lugoli et al., submitted).

An additional advantage of the size-structure approach is the availability of automatic techniques that can reduce the cost associated to extensive monitoring programmes. Over the last decades, new image analysis systems have been developed for rapid and high-resolution plankton data acquisition. In this context, the FlowCAM allows the analysis of natural samples containing particles in the nano-microplankton size range (Sieracki et al., 1998; Zarauz et al., 2007; 2009).

Taking into account the current need of developing cost-efficient tools for the assessment of water quality, the present study was planned to test if (i) the FlowCAM could be employed for counting cells and characterizing the size-structure of the dominant phytoplankton in coastal and estuarine systems; (ii) it had a potential for developing phytoplankton indicators; and (iii) it could be efficiently used in environmental monitoring networks.

Study area and methods

The Basque coast extends along 150 km in the north of Spain, south-eastern Bay of Biscay (Figure 1). The climate is temperate oceanic, with frequent freshets throughout the year. The coastal waters belong to type 1, within the Northeast Atlantic ecoregion, following the typology system of the Water Framework Directive (WFD). They are exposed, shallow (<30 m), and euhaline fully-mixed waters, with very little influence of

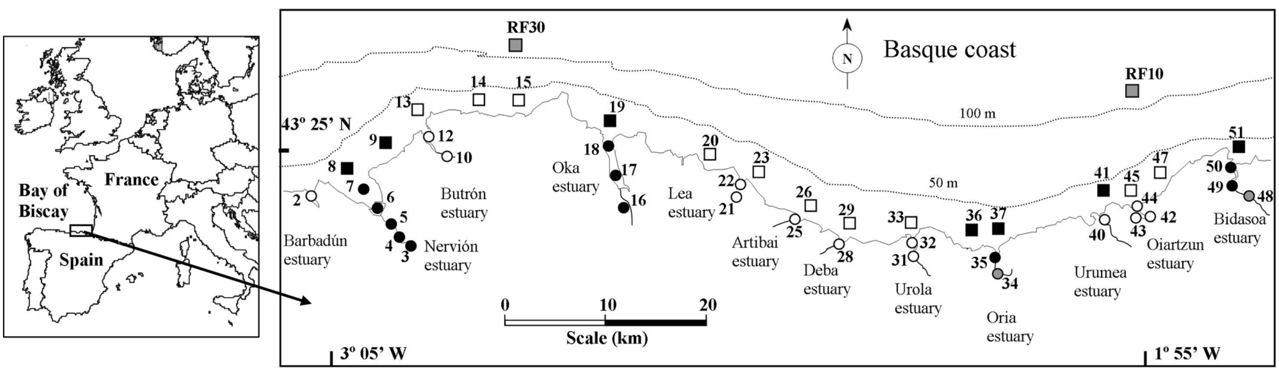

Figure 1: Study area and location of sampling stations. Key: Squares-coastal stations; circles-transitional stations; black symbols-stations sampled in 2005 and 2008; white symbols-stations sampled only in 2005; grey symbols-stations sampled only in 2008.

upwelling and large river plumes as natural fertilization factors. The estuaries are meso-macrotidal, short (2-22 km) and shallow systems (<10 m, with the exception of few outer bays).

Two WFD monitoring campaigns were conducted in 2005 and two in 2008, which corresponded to the spring and summer season, respectively. In each campaign, samples were taken in coastal and transitional waters (Figure 1). Samples for phytoplankton analysis were preserved with glutaraldehyde (0.2%) and maintained in cold and dark conditions until their analysis. Two FlowCAM analytical procedures were applied, different for the 2005 and 2008 samples (see Table 1 for details).

Both FlowCAM procedures operated in AutoImage mode, i.e. without fluorescence measurements.

Table 1: Methodological differences and similitudes between the two FlowCAM procedures.

FlowCAM procedure	2005	2008
Operation Mode	*AutoImage mode*[1]	*AutoImage mode*[1]
Objective	*4x*	*4x*
Flow-cell size	*300 μm*	*300 μm*
Lower size limit	*10 μm*	*4 μm*
Run-stop criteria[2]	*20 minutes*	*4000 particles*

[1] This mode captures an image of any particle within the specified size range (>10 μm in 2005, and >4 μm in 2008) at a constant rate, without using fluorescence detectors.
[2] The analysis can be configured to automatically stop based on the analysis duration or on the number of particles counted.

Consequently, every particle (phytoplankton, zooplankton and inorganic particles) was counted and imaged as described by Zarauz et al. (2007; 2009). The 2005-procedure was more appropriate for counting the large-size particles (10-100 μm). The 2008-procedure was more suitable for capturing the particles in the small-size range (4-50 μm). The Normalised Biomass Size Spectrum (NBSS) was used to describe the size-structure of the plankton community (Zarauz et al., 2009). Inverted microscopy was used for phytoplankton identification and counting (Utermöhl, 1958); and for comparison with FlowCAM estimations.

Among other physico-chemical variables, salinity, inorganic nutrients and optical properties were measured using standard methods (for more details, see Garmendia et al., 2011).

Results and Discussion

In general, the FlowCAM estimated a higher concentration of particles than the microscopy technique. These differences were accentuated in 2008 (Figure 2).

However, in 2005 some samples from the summer campaign showed FlowCAM/Utermöhl ratios lower than 1 (Figure 2). With the 2005-procedure only particles larger than 10 μm were detected (Table 1). But, with microscopy, cells that ranged 4-10 μm were also counted (i.e. small flagellates and diatoms) and these can be an important component of the community in the study area (Orive et al., 2004; Garmendia et al., 2011). Consequently, the actual phytoplankton concentration

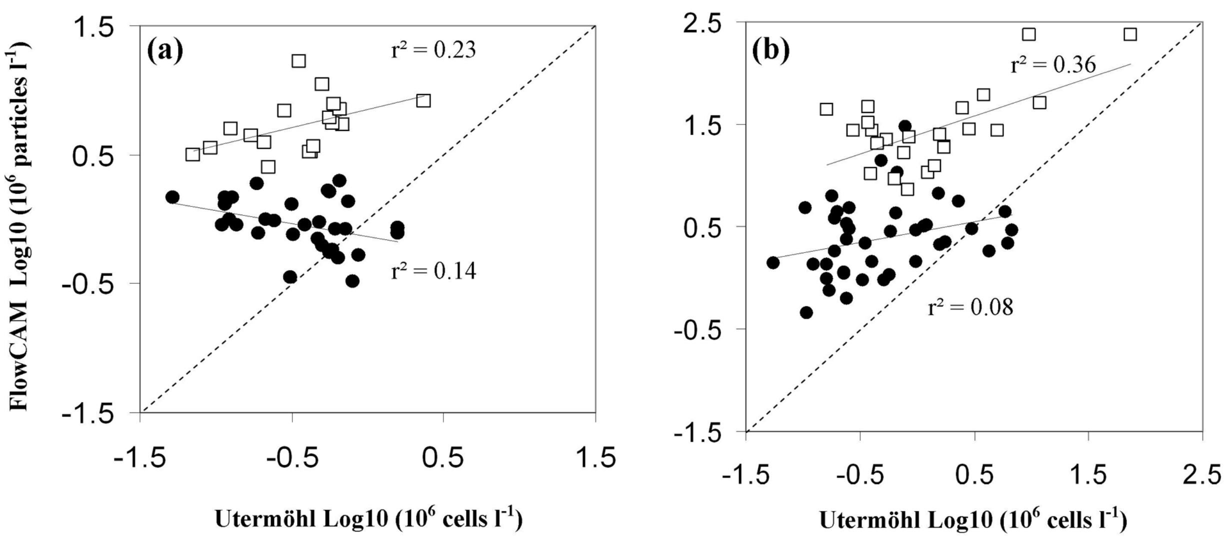

Figure 2: Correlations between FlowCAM and microscopy counts in samples from (a) coastal waters, and (b) transitional waters. The dotted line indicates the 1:1 relationship. Solid symbols represent the FlowCAM procedure for large-size particles (2005 campaigns); open symbols represent the FlowCAM procedure for small-size particles (2008 campaigns).

could have been underestimated by FlowCAM in 2005. In contrast, in 2008 the lower size limit of the FlowCAM was similar to that of the microscopy technique (i.e. 4 μm) (Table 1). However, since fluorescence was not measured and a high concentration of other particles than phytoplankton is expected in the small-size range, the abundance of phytoplankton could have been overestimated by the 2008-procedure. In addition, a smaller sample volume was processed in 2008, since the analysis stopped with 4000 particles and the small particles saturated that capacity very quickly (Table 1). Therefore, in 2008 the large-size classes were probably underestimated and the size-structure of the phytoplankton community was biased.

In summary, none of the two FlowCAM approaches showed reliable abundance and size-structure estimations to characterize the phytoplankton communities within the Basque coastal and transitional systems.

The relationships between the FlowCAM and environmental variables confirmed the interference of the suspended material in the FlowCAM estimations, since all the FlowCAM parameters were positively correlated ($p<0.05$) with suspended solids and/or turbidity (Table 2).

Indicators should reflect the effect of the pressure to be assessed, but they must not be affected by natural variability factors (Hering et al., 2010). As it can be seen in Table 2, particles concentration was the unique FlowCAM parameter which was only related to a pressure indicator (phosphate, a nutrient of anthropogenic origin in the study area). Some other FlowCAM parameters showed significant positive correlations with ammonia or phosphate (e.g. the NBSS intercept and the carbon biomass). However, these FlowCAM parameters were also related to nitrate and/or silicate, which are nutrients mainly associated to rain events and natural dilution gradients in the Basque estuaries (Valencia and Franco, 2004). Summarizing, FlowCAM parameters showed a limited potential for developing phytoplankton indicators. However, although no standardized methodology exists using phytoplankton biometric measures to discriminate among different trophic conditions (Mouillot et al., 2006), different authors (Sprules and Munawar, 1986; Marañón et al., 2001; Iriarte and Gonzalez, 2004) have underlined the potential of phytoplankton size-structure to discriminate between trophic conditions, and for their inclusion in routine monitoring.

The FlowCAM as employed in this study (AutoImage mode) was not suitable to accurately estimate the abundance and size structure of the phytoplankton communities. However, the combination with Fluorescence mode could lead to more reliable results; especially in estuaries, where the interference of the suspended material is high. Though, the inconveniences of using different working modes and magnifications, and the samples pre-treatment (filtration, dilution and concentration) for each of the working modes, can make this technique time-consuming, unless a detailed protocol is defined (Figure 3).

Conclusions

Although the FlowCAM could be eventually used for deriving cost-efficient indicators of water quality based on phytoplankton size-structure, further studies are necessary, especially in estuaries, due to the suspended material interference. Also, the standardization of the FlowCAM procedures is highly recommended to obtain reliable estimations in a practical amount of time.

Table 2: Spearman Rank Correlations between FlowCAM and physico-chemical variables. The correlation coefficient is reported, with the level of significance (p) in brackets. N = 8; data have been grouped in 8 clusters that differentiate FlowCAM procedures (2005 vs. 2008), water categories (coastal vs. transitional) and seasons (spring vs. summer). NBSS: Normalised Biomass-Size Spectrum; DIN: Dissolved Inorganic Nitrogen; n. s.: non-significant.

Group	Physico-chemical variables	Particles	Carbon biomass	NBSS Intercept	NBSS Slope
Variables mainly related to distance from land or freshets (natural variability)	*Salinity*	*-0.37 (n.s.)*	*-0.78 (p<0.05)*	*-0.95 (p<0.05)*	*0.10 (n.s.)*
	Nitrate	*0.41 (n.s.)*	*0.85 (p<0.05)*	*0.86 (p<0.05)*	*0.05 (n.s.)*
	Silicate	*0.21 (n.s.)*	*0.69 (n.s.)*	*0.83 (p<0.05)*	*0.02 (n.s.)*
Nutrients related to anthropogenic pressure	*Ammonia*	*0.24 (n.s.)*	*0.67 (n.s.)*	*0.81 (p<0.05)*	*0.00 (n.s.)*
	Phosphate	*0.88 (p<0.05)*	*0.83 (p<0.05)*	*0.69 (n.s.)*	*0.48 (n.s.)*
Nutrients that can be related to both natural variability and pressure	*DIN*	*0.21 (n.s.)*	*0.69 (n.s.)*	*0.83 (p<0.05)*	*0.02 (n.s.)*
	Nitrite	*0.59 (n.s.)*	*0.75 (n.s.)*	*0.78 (p<0.05)*	*0.30 (n.s.)*
Optical properties	*Secchi disc depth*	*-0.29 (n.s.)*	*-0.74 (n.s.)*	*-0.81 (p<0.05)*	*0.00 (n.s.)*
	Turbidity	*0.37 (n.s.)*	*0.84 (p<0.05)*	*0.99 (p<0.05)*	*-0.11 (n.s.)*
	Suspended solids	*0.95 (p<0.05)*	*0.76 (p<0.05)*	*0.38 (n.s.)*	*0.76 (p<0.05)*

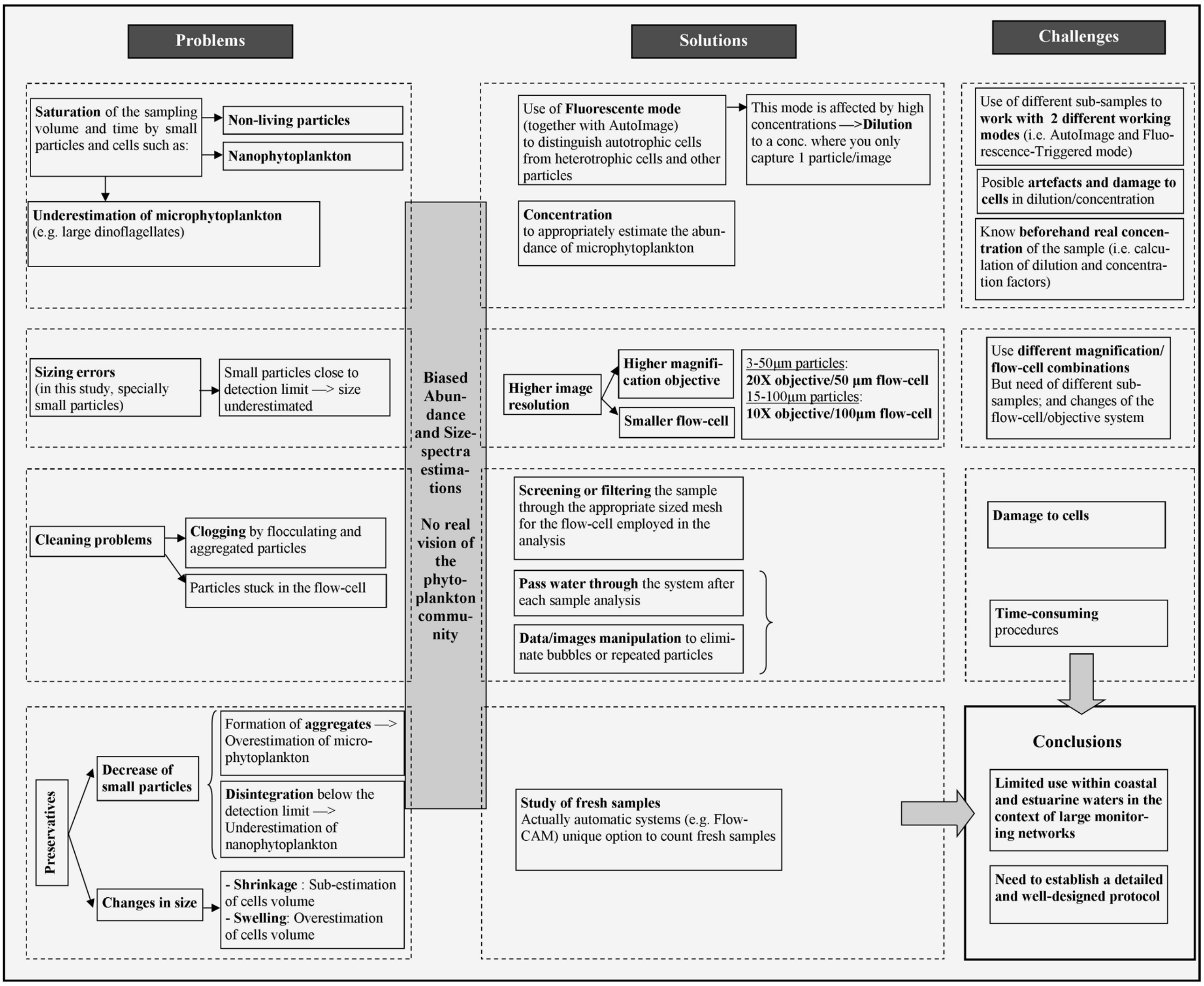

Figure 3: FlowCAM Diagram. Methodological problems found in this study, possible solutions and challenges.

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The potential for using brown trout stocking as a biocontrol agent

K.Ø. Gjelland[1, 2], P.-A. Amundsen[1] & O.T. Sandlund[2]

[1] Department of Arctic and Marine Biology, University of Tromsø, 9037 Tromsø, Norway
[2] Norwegian Institute for Nature Research, Tungasletta 2, 7047 Trondheim, Norway

Email address of corresponding author: karl.gjelland@uit.no

Abstract

Invasive species represent a great management challenge. Once an alien species is established in a new ecosystem, it may be very difficult or impossible to get rid of it. Often the only feasible management action is to use different means to control the size of the invader population. Through two stocking experiments we explored the potential for using brown trout (*Salmo trutta*) as biocontrol agent on the population of invasive vendace Coregonus albula in the Pasvik watercourse. Vendace is a planktivorous fish species often dominating the pelagic zones in lakes were it is present. The experiments were performed in lakes of similar area (about 6 km^2), but highly different lake morphology (pelagic zone to littoral zone size ratios). The planktivorous fish populations (vendace and whitefish) were monitored by gillnet fishing and echosounding surveys, whereas piscivorous fish (brown trout and pike) were monitored by gillnetting and acoustic telemetry. Stocked brown trout foraged primarily on vendace, and had a rapid growth. Recaptures and telemetry indicated a good survival of brown trout in the deep lake where the pelagic zone dominated the lake area. Brown trout survival was poor in the lake were littoral habitats constituted > 50% of the lake area. The poor brown trout survival was caused by pike predation. We conclude that stocking of native brown trout may offer a beneficial tool for targeting invasive planktivorous fish like vendace. Managers must, however, consider lake morphology and the extension of pike habitats before the initiation of a brown trout stocking program.

Using aquatic plant sub-fossils to define reference conditions in shallow eutrophic lakes: a palaeolimnological perspective

Ben Goldsmith[1], Thomas A. Davidson[2] & Helen Bennion[1]

[1] Environmental Change Research Centre, Department of Geography, University College London, Gower Street London, WC1H 6BT, UK
[2] Department of Bioscience, Aarhus University, Vejlsøvej 25, DK-8600 Silkeborg, Denmark

Email address of corresponding author: b.goldsmith@geog.ucl.ac.uk

Background

With the aim of the EU Water Framework Directive (European Union, 2000) being to achieve "good ecological quality" in all qualifying waters, there is a need to define 'good' in terms of not only the physico-chemical and hydromorphological environment, but also the aquatic biology. The WFD identifies these biological quality elements (BQE) to include fish, invertebrates, macrophytes, phytobenthos and phytoplankton with overall ecological quality judged by the degree to which the present-day assemblages deviate from those in the past, prior to anthropogenic influence. These so-called 'reference conditions' are a key element in defining the ecological status. A lake with BQEs which differ little from the reference condition will be classed as having "High" status, whereas increasing deviation from this reference will result in categories of Good, Moderate, Poor and Bad status being attributed. The determination of these reference conditions and a system for setting ecological status boundaries is therefore crucial for the implementation of the WFD and palaeolimnology is suggested as one such approach (Pollard and Huxham 1998; European Union 2000; Bennion and Battarbee 2007).

Palaeolimnological techniques have been employed extensively over the past few decades with a view to defining baseline conditions and setting restoration targets for lakes (see reviews in Battarbee 1999 and Smol 2008). A wide range of different fossil types have been used: often to reconstruct particular environmental parameters (e.g. nutrient status, acidity and temperature). With the focus of the WFD on ecological quality however, there is an advantage to retaining ecological information and thus the direct analysis of biological components in the sediment record can provide valuable information. Aquatic plant macrofossils are one such group that has been used to good effect in the palaeo-record (Birks 1980; Birks 2001) and potentially allows the structural and functional characteristics of pre-impact aquatic ecosystems to be reconstructed (e.g. Davidson et al. 2005; Salgado et al. 2010). Due to the paucity of shallow, lowland lakes of good ecological status in the UK, palaeolimnological methods may be the only way to determine pre-impact conditions (Bennion et al. 2003, Bennion and Battarbee 2007, Bennion et al. 2010).

Methods

A total of 74 sediments cores taken from 61 different lakes throughout the UK have been analysed at UCL for plant sub-fossil remains. With the original reason for taking and analysing cores being varied, the first stage of the analysis was to collate the data and assess its suitability for the purposes of defining reference conditions. A subset of 30 cores was finally used; restricted by lake type (high alkalinity, shallow) and the availability of dating back to at least 1850; this being arbitrarily chosen as the date representing pre-impact conditions in the UK (Bennion et al. 2010).

The data were harmonised and sub-fossil pseudo-species defined for each species and remain type (e.g. *Potamogeton obtusifolius* seed, leaf-tip, leaf fragment, turion etc.) and coded accordingly. Due the great disparity in total numbers of remains in a single sample (e.g. ranging from one seed, to thousands of *Ceratophyllum sp.* leaf spines), the data were square root transformed, centred and standardised, which proved sufficient for the subsequent analysis. The sub-fossil data from submerged and floating-leaved species were plotted using the specialist stratigraphic software C2 (Juggins 2003) and analysed further using CANOCO (ter Braak & Smilauer 2002) to undertake detrended canonical correspondence analysis (DCCA) (ter Braak 1986; ter Braak & Verdonschot 1995). This latter technique was used to examine the variation in pseudo-species data from all the dated cores using age as a single constraining variable. DCCA has the advantage that it scales the axis

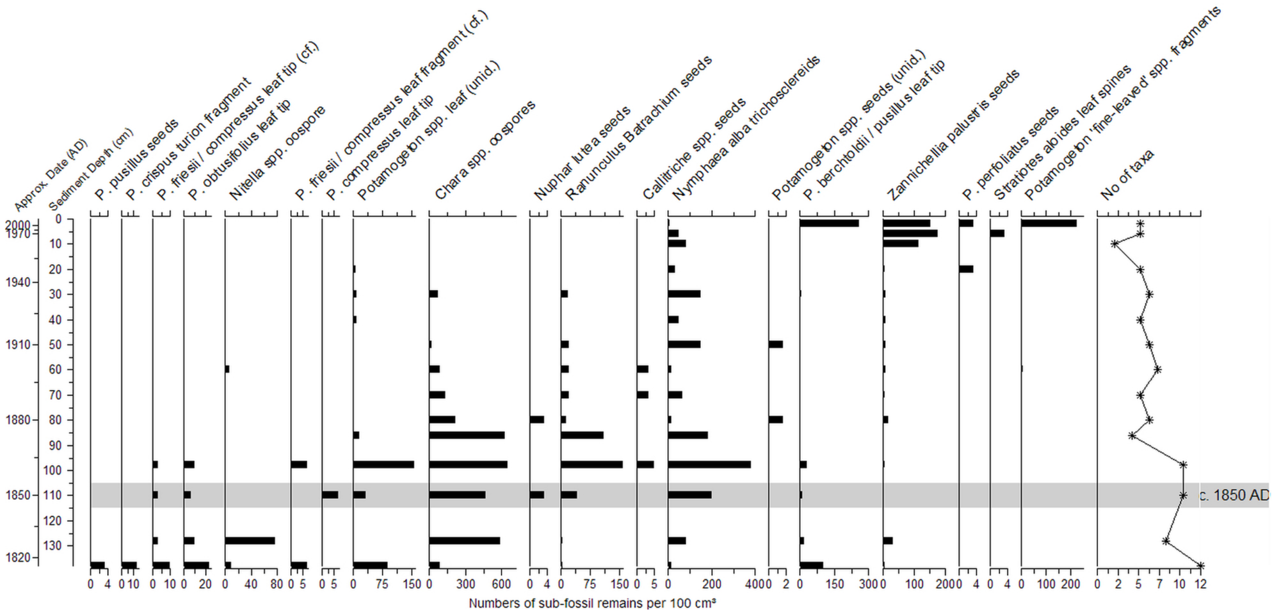

Figure 1: Aquatic macrophyte sub-fossil remains from core ORMG2; Ormesby Great Broad, Norfolk, England (adapted from Davidson et al. 2008)

scores in units of standard deviation equating to species turnover (Smol et al. 2005), thus allowing community shifts to be assessed through time. The technique measures the dissimilarity of species between samples; a low axis 1 score denotes a high level of similarity, whereas a high score indicates a low level of similarity. For samples constrained by age within the DCCA analysis, the technique examines species turnover through time as the primary axis and hence tracks the extent of change through time.

Results

Many of the sediment cores analysed from lowland lakes in the UK show clear trends from species rich mixed plant communities in the past, to fewer and more typically 'eutrophic' species towards the present. Ormesby Broad, a shallow, eutrophic water body in Norfolk, England, is a good example of this (Fig. 1)

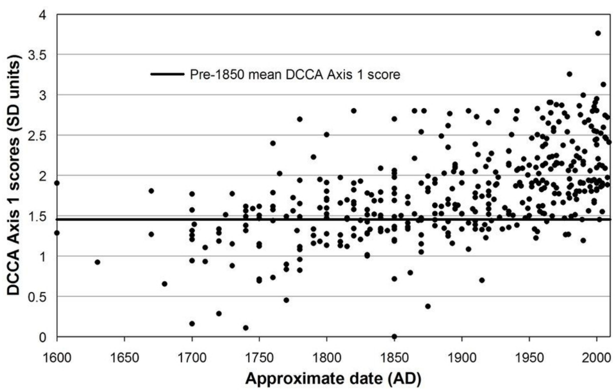

Figure 2: DCCA scores for macrofossil remains from 30 sediment cores, constrained by date.

Using DCCA with samples constrained by date the total pseudo-species turnover was estimated from the sub-set of 30 dated cores, yielding DCCA axis 1 scores of 0.00-3.76 (standard deviation units). A clear trend can be seen in the data, with higher values tending towards the more recent samples (Fig. 2). This can be broadly interpreted as an increase in species turnover since 1850 due to increased pressures on the freshwater environment.

With very few examples of modern reference sites of this lake type in the UK, we can use instead the mean DCCA Axis 1 scores based on the sub-fossil record of the pre-1850 sites as our estimate of reference conditions (mean = 1.43, median 1.45, hence 1.44). It is therefore suggested that samples with a score greater than 1.44 SD units can be said to be moving away from reference condition. The extent to which this is significant has yet to be tested, but when examined on a site by site basis, it is clear that since c.1850 there have been considerable shifts in the aquatic plant communities away from the palaeo-derived reference condition.

Figure 3 a and b show data from two heavily degraded sites which are now turbid with few or no submerged aquatic macrophytes: this is typical for the majority of lakes in the dataset. Ormesby Broad and Groby Pool (Figure 3 c and d) still support a reasonable aquatic flora, but the sub-fossil record suggests some deterioration, whereas Kilroosky Lough and Kenfig Pool (Figure 3 e and f) are lowland lakes which still have a diverse aquatic flora and maintain clear water conditions. These latter two sites are the best examples of UK

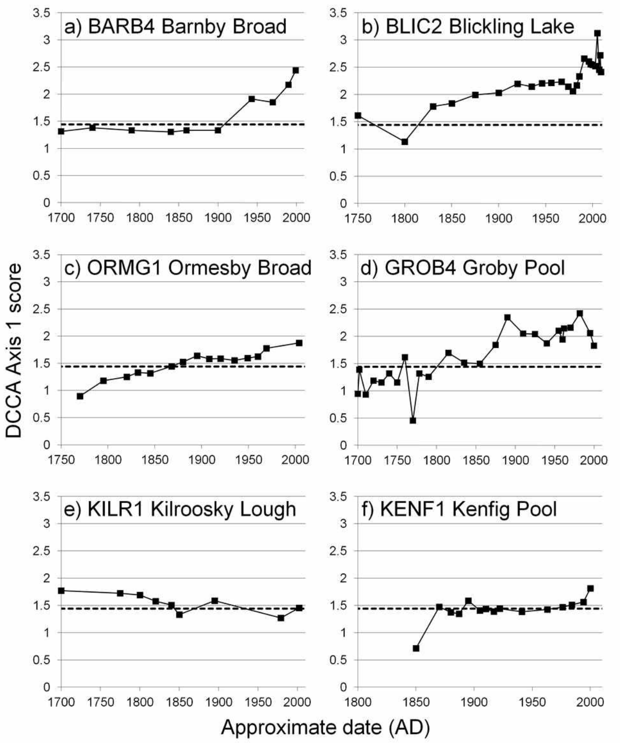

Figure 3: Individual sites showing DCCA Axis 1 scores tracked through time. The dashed line is the mean score based on pre-1850 'reference' samples

lowland lakes within the current dataset with respect to their modern flora and show the least deviation from the palaeo-derived reference condition.

Conclusion

In conclusion, aquatic plant macrofossils provide an insight not only into some of the past species assemblages that might have populated 'reference' communities in lakes, but also it can be demonstrated that these assemblages have in many cases shifted away from the relatively stable conditions of pre-1850. Analysis of the macrophyte remains from lake sediments is a valuable technique for examining the direction and magnitude of change away from pre-disturbance conditions and therefore this palaeo-limnological tool has the potential for both defining reference conditions and for setting restoration targets for lakes.

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Overview and outcome of WISER, future research needs and obstacles

Daniel Hering, Sebastian Birk, Christian K. Feld & Jörg Strackbein

University of Duisburg-Essen, Department of Applied Zoology / Hydrobiology

Email address of corresponding author: daniel.hering@uni-due.de

Introduction

WISER was probably the last of many European Union funded projects on the definition of ecological status and the development of assessment and classification systems. At the same time, it was one of the first large international projects dealing with restoration of different aquatic ecosystem types (rivers, lakes, transitional and coastal waters). This contribution gives an overview of the main results achieved by the project and places them in the context of the implementation process of the Water Framework Directive (WFD). Still existing knowledge gaps in the fields of ecological assessment and restoration are outlined and future research needs are defined.

WISER experiences with data handling and availability

One of the major consequences of the WFD is the acquisition of large amounts of biological information on the status of European surface waters, information that may improve our knowledge of the structure of the communities inhabiting these ecosystems. Potentially, these data could contribute significantly to other objectives in addition to those of the WFD, e.g. for monitoring the effects of emerging stressors, for improving our knowledge of species distributions and species invasions, for understanding broad scale drivers shaping community assemblages, for Habitats Directive/Natura 2000 species inventories and biodiversity records. However, as with the variability of methods employed for collecting data, the data structure, quality and quantity are quite variable. This applies to the underlying taxonomy and taxonomic identification codes, taxonomic resolution, density of sampling sites, sampling frequency and data storage (Hering et al. 2010).

WISER has spent considerable time and resources on acquiring and harmonising existing datasets from research projects and monitoring programmes for the development of assessment systems using individual Biological Quality Elements (BQEs), for comparing the response of BQEs to stress and stress release in different aquatic ecosystem types. This has largely been successful on the level of individual combinations of BQEs and water types (e.g. for lake phytoplankton or for fish in transitional waters), while the generation of sufficient data sources for comparing different BQEs and water types proved to be much more difficult. Despite all progress with data handling and processing in the context of the WFD there are still different “schools” in storing environmental data, both on the levels of countries and concerning different BQEs or ecosystem types. The data policy and potential future uses of the data collected and assembled by WISER will be outlined in the contribution (compare also http://www.wiser.eu/programme/data-and-guidelines/data-services/).

Central storage and quality assurance of data on aquatic ecosystems is presently only performed on a generic level, the overall ecological status and selected metrics, rather than original data (e.g. taxa lists), which limits their potential for large-scale analyses and for purposes beyond the WFD. Initiatives to improve this situation include new actions of the EEA in the framework of the WISE system, the EU-funded biodiversity project BioFresh (www.freshwaterbiodiversity.eu) as well as the INSPIRE Directive and the SEIS (Shared Environment Information System) initiative.

For aquatic ecologists it would be desirable to step by step establish an “open access culture” for research projects, i.e. agreeing on a way to make data collected in research projects publicly available, once the projects are finished and the results are published. For monitoring data collected for the WFD it is desirable to establish a network of surveillance monitoring sites, on which the original data are centrally stored.

Assessment

In cooperation with experts from ECOSTAT and the Geographical Intercalibration Groups (GIGs) WISER has produced an overview of all assessment methods presently applied to implement the WFD in the European member states (Birk et al. 2012, http://www.wiser.eu/results/method-database/).

Overall, more than 300 methods are presently applied, which outlines the magnitude of the intercalibration task, which has been performed in recent years.

Countries of Central and Western Europe have developed almost all methods required for the WFD implementation. Two main sampling strategies were discernable: Small-scale sampling of the taxonomically diverse groups of benthic invertebrates and phytobenthos that require elaborate processing, versus large-scale sampling of vast, species-poor plant stands or the mobile fish fauna. About three-quarters of 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 prevailed. Fish-based methods showed the highest number of metrics. Most methods focussed on the detection of eutrophication and organic pollution. Habitat loss was mainly assessed by methods applied to rivers and transitional waters. The pressure-impact relationship of about one-third of methods was not tested empirically with methods for transitional waters being the least validated. Status boundaries were mostly defined using statistical approaches (compare Birk et al. 2012).

WISER has significantly contributed to developing new methods for BQE-water type-combinations, which have mainly been neglected over the last years. Details will be given in other presentations on the final conference and are further summarized on standardized metric fact sheets (http://www.wiser.eu/results/common-metrics/).

Summarizing the experiences with the analysis of the WISER data and the methods reported by Birk et al. (2012) it is obvious that the main stressors affecting Europe's surface waters can be sufficiently indicated by individual BQEs or combinations (Figure 1). The figure outlines also the strength of the WFD approach to use

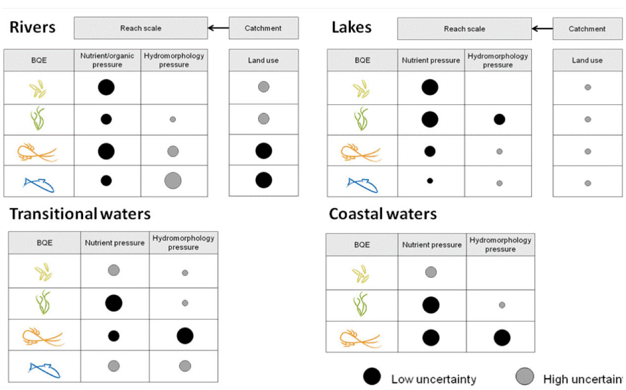

Figure 1: Course classification of the indication potential of BQEs in European water categories, based on WISER results and established assessment methods.

different BQEs for status assessment, as the response to stress might differ.

The variability of status assessment methods has been mitigated by the now almost finalized intercalibration process. The remaining challenges include the combination rules of BQEs for an integrated status assessment ("one out-all out" principle) and the assessment of Heavily Modified Water Bodies (HMWBs) (Hering et al. 2010).

Restoration

As the majority of surface waters in the EU fail achieving good ecological status / good ecological potential a huge number of restoration and mitigation measure will be required; these have been outlined in the first River Basin Management Plans, while the implementation of the majority of measures lies still ahead. WISER has generated an overview of restoration effects, separately for rivers, lakes and coastal/transitional waters, and has specifically targeted the effects of individual restoration programmes. Although the effects are variable and strongly depending on the local situation some general experiences can be formulated:

Rivers

Local restoration ecologically often ineffective

- Successful restoration requires changes in land use along rivers ("buffer strips")
- Successful restoration requires time spans of 10 years+
- Local measures to be embedded into larger scale concepts

Lakes

- Climate change effects aggravate eutrophication effects through food-web interactions and increased pressures
- 10-20 years required for recovery of lakes from nutrient pressures

Coastal and Transitional Waters

- Shifting baselines: Historical references are not achievable any more
- Strong temporal and spatial variability due to dynamic conditions
- 10-20 years required for recovery

Details will be given in other presentations on the final conference.

Research needs

Future research needs for the WFD implementation are particularly related to the development of aquatic ecosystems and ecological status following restoration and emerging stressors. While the general response to restoration, time frames required, the impact of restoration on ecosystem services and long-term monitoring schemes are equally relevant for rivers, lakes and marine ecosystems, there are some specific items particularly relevant for individual water types:

Rivers

– Length, width and architecture of ecologically effective buffer strips

– Practical instruments to establish buffer strips

Lakes

– Long-term monitoring data for addressing climate change and restoration effects

Coastal and Transitional Waters

– How to measure resistance and resilience?

– Considering connectivity for restoration measures (transitional waters)

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Increasing first year growth of perch in Swedish forest lakes?

Kerstin Holmgren

Swedish University of Agricultural Sciences, Department of Aquatic Resources, Institute of Freshwater Research, Stångholmsvägen 2, SE-178 93 Drottningholm, Sweden

Email address of corresponding author: kerstin.holmgren@slu.se

Key words: time series, perch, first year growth, water temperature, organic carbon

Introduction

Perch (*Perca fluviatilis*) is the most widespread and abundant fish species in Swedish forest lakes. The optimum temperature for growth is 23 °C (e.g. Fiogbe & Kestemont 2003), which presently may be reached and exceeded in surface water for short summer periods. Water temperature was not a main factor explaining between-lake differences in growth efficiency of perch (Holmgren & Appelberg 2001). Competitive and predatory interactions were suggested to mask effects of abiotic factors like water temperature and organic carbon. Reduced light condition mediated by increased organic carbon was recently suggested to limit fish production in nutrient-poor lake ecosystems (Karlsson et al. 2009). Changes in light conditions might influence growth potential of a visual feeder like perch, and reduce its ability in competition with species like roach and ruffe (Diehl 1988, Bergman 1988). Within lakes, variation in temperature might still influence first year growth and year-class strength (Holmgren 2001). Lake water temperature might increase due to global climate change, while light conditions might decrease due to increased runoff of organic carbon from lake catchments. This study focussed on forest lakes with more or less annual recruitment of both perch and roach (*Rutilus rutilus*). Times series of first year growth of perch were evaluated in relation to surface water temperature, and trends of total organic carbon and some possible confounding abiotic factors were also explored.

Material and Methods

Eleven lakes with annual mean pH > 6 and viable populations of perch and roach, based on annual monitoring since 1994. Fish communities were sampled in July or August, using multi-mesh gillnets (Holmgren 1999, CEN 2005). Four to eight fish species occurred in each lake, and 14 species were observed at least once in some lake. All lakes were inhabited by pike (*Esox lucius*). In addition to perch and roach, the following species always occurred; ruffe (*Gymnocephalus cernuus*, 6 lakes), rudd (*Scardinius etythroptalmus*, 3 lakes), vendace (*Coregonus albula*, 3 lakes), whitefish (*Coregonus lavaretus*, 3 lakes), smelt (*Osmerus eperlanus*, 2 lakes), bleak (*Alburnus alburnus*, 2 lakes) and bream (*Abramis brama*, 1 lake). The lakes were situated at low to mid altitude (35-268 m). They were rather small (18-489 ha), with maximum depth of 9-42 m, and mean depth of 4-14 m. Mean values of total phosphorous (Total P) ranged from 5-12 µg/L, and total organic carbon (TOC) was 4-11 mg/L (Tab. 1).

Perch were aged using operculum bones and sagittal otoliths. Length after the first year (L at 0+, mm) was back-calculated according to Holmgren & Appelberg (2001), and used as a measure of first year growth. Mean values of first year growth were calculated for each cohort hatched in 1993-2009. Since 1998-2000, one or two temperature loggers per lake recorded surface water temperature (at 1-1.5 m depth) four or six times per day. Daily mean values were averaged for May-September, T(May-Sep). For years with missing data, T(May-Sep) was estimated using a linear regression with mean temperature at 0.5 m depth, recorded at monthly sampling for water chemistry in May-September. Other abiotic factors were expressed as annual mean of 7-8 surface samples per lake and year, taken at 0.5 m depth at a mid-lake station. Concentrations of Total P (µg/L), TOC (mg/L) and sulphate (SO_4, meq/L) were analysed at the Department of Aquatic Sciences and Assessment, following international (ISO) or European standards (EN).

Non-parametric correlation (Kendall's tau) with calendar year was used to test for occurrence of monotonic trends in L at 0+ and each of the abiotic factors, within each of the eleven lakes. Linear regression was used to explore the overall relationship between L at 0+ and T(May-Sep). The residuals were saved to test for between-lakes differences, using Oneway ANOVA.

Table 1: Mean of annual values of L at 0+, T(May-Sep), Total P, TOC and SO_4, and monotonic trends tested by Kendall's tau. Significantly increasing or decreasing trends are marked with bold or italic fonts. Lakes are sorted from south to north.

Lake (N years)	L at 0+ (mm)			T (May-Sep, °C)			Total P (microg/L)			TOC (mg/L)			SO_4 (meq/L)		
	Mean	tau	P	Mean	tau	P	Mean	tau	P	Mean	tau	P	Mean	K's tau	P
628606 (17)	73	**0,353**	**0,048**	17,6	**0,574**	**0,001**	8,7	-0,250	0,161	4,6	0,309	0,084	0,18	-0,853	0,000
633025 (14)	73	**0,495**	**0,014**	16,7	0,385	0,055	12,3	-0,034	0,869	6,7	0,187	0,352	0,17	-0,956	0,000
638317 (17)	69	0,191	0,284	15,8	**0,426**	**0,017**	8,5	-0,465	0,009	10,8	**0,544**	**0,002**	0,10	-0,824	0,000
640364 (17)	69	0,015	0,934	16,5	0,265	0,138	5,6	-0,558	0,002	5,1	**0,397**	**0,026**	0,13	-0,985	0,000
642489 (17)	66	**0,676**	**0,000**	17,7	0,162	0,365	7,2	-0,415	0,021	7,3	0,052	0,773	0,18	-0,971	0,000
645289 (16)	66	0,133	0,471	16,6	**0,567**	**0,002**	9,5	-0,403	0,030	9,1	**0,517**	**0,005**	0,14	-0,933	0,000
652412 (17)	65	0,265	0,138	16,8	**0,353**	**0,048**	9,8	-0,313	0,082	12,8	**0,515**	**0,004**	0,12	-0,779	0,000
655587 (15)	71	**0,429**	**0,026**	17,1	**0,410**	**0,033**	8,8	-0,371	0,054	9,4	**0,448**	**0,020**	0,14	-0,886	0,000
656419 (17)	64	0,221	0,217	17,1	**0,353**	**0,048**	7,5	-0,320	0,076	8,7	0,059	0,742	0,13	-0,441	0,013
683673 (17)	63	0,279	0,118	15,0	0,185	0,303	7,3	-0,468	0,009	6,7	0,221	0,217	0,04	-0,716	0,000
708619 (16)	61	0,000	1,000	13,4	0,226	0,224	10,2	-0,350	0,059	9,3	0,159	0,392	0,03	-0,650	0,000

Results and discussion

L at 0+ either increased or showed no significant trend (Tab. 1), and T(May-Sep) showed a similar pattern. Total P tended to decrease. TOC tended to increase in accordance with a more general trend of increasing dissolved organic carbon since 1990 across eastern North America and northern and central Europe (Monteith et al 2007). The most consistent trend was, however, a significant decrease in SO_4 in each of the lake, as previously reported from Swedish lakes (Fölster et al. 2005) and rivers (Erlandsson et al. 2008), reflecting a long term decrease in airborne acidifying deposition.

Time series of T(May-Sep) were more or less synchronised, with relatively warmer growth seasons in 1994, 1997, 1999, 2002 and 2006 (Fig. 1). Considerably lower values were usually found in the two northernmost lakes. L at 0+ was similarly synchronised between lakes, although to a somewhat lesser extent. The most synchronised peaks occurred in 1997, 2002 and 2006. The mean first year growth was lowest in the two northernmost lakes, but relatively high considering their lower temperature. Far north a longer day length during summer might to some extent compensate for a shorter growth season. T(May-Sep) explained 36 % of the overall variation in L at 0+, and positive correlations were indicated within lakes as well. Residual L at 0+ differed significantly between lakes (Oneway ANOVA, $P < 0.001$). The largest negative residuals (i.e. lower than expected growth) were found in two of the smallest, but rather deep lakes, with pelagic populations of vendace. The largest positive residuals (higher than expected growth), were found in one of the largest, but shallowest lakes, with no or weak thermal stratification in late summer.

In summary, first year growth of perch tended to increase, and a positive effect of temperature was revealed both by the overall relationship using data from all lakes, and in the time series within lakes. There was no evidence that a negative effect of increasing TOC overrules the positive effect of increasing temperature. A further increase in water temperature might therefore be beneficial for juvenile perch in the studied lakes. Better conditions for growth of young fish might be just one possible reason for increasing dominance of small and young fish in a warmer climate (e.g. Jeppesen et al. 2010).

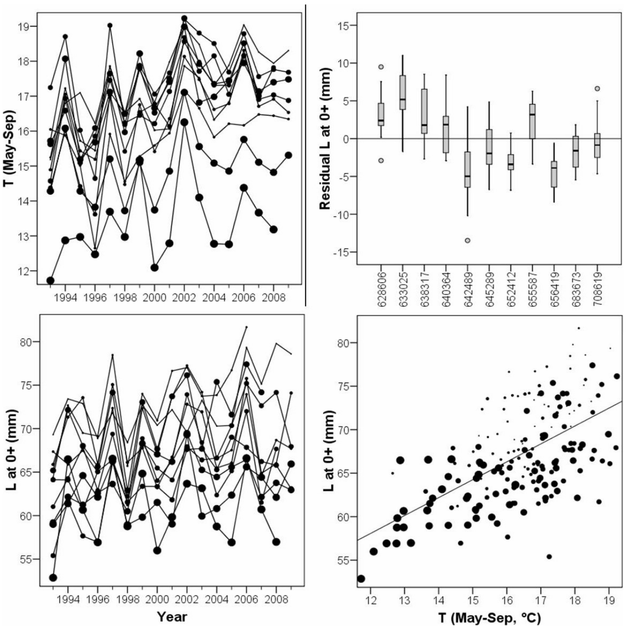

Figure 1: Time series of surface water temperature (upper left) and first year growth of perch (lower left). Lake symbols have increasing size from south to north. The overall linear relationship is shown in the lower right panel. The lake-specific residual variation is shown in the upper right panel.

Acknowledgements

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The classification of the Biological Quality Element phytoplankton for the Water Framework Directive in the transitional waters of the Rio Mondego, Portugal

John Icely[1], Cátia Luis[1], Peter Henriksen[2], Karstern Dromph[2], Bruno Fragoso[1]

[1] IMAR, Institute of Marine Research, FCT, University of Coimbra,l3004-517 Coimbra, PORTUGAL
[2] NERI, National Environmental Research Institute , Aarhus University, DK-4000 Roskilde, Denmark

Email address of corresponding author: john.icely@gmail.com

Introduction

This study the EU funded large-scale integrating project WISER (Water Bodies in Europe: Integrative System to assess Ecological Status and Recovery), considers the implementation of the Water Framework Directive (WFD) for coastal and transitional waters in Portugal with specific reference to the biological quality element (BQE) phytoplankton in the transitional waters (TWs) of the Rio Mondego. TWs are defined by the WFD as "bodies of surface water in the vicinity of of river mouths which are partly saline in character as a result of their proximity to coastal waters but which are substantially influenced by freshwater flows" (CEC, 2000). The WFD requires Member States to report on the state of their water bodies (WBs) according to a five level classification system (High, Good, Moderate, Poor and Bad). The ecological status is expressed as the Ecological Quality Ratio (EQR) between a reference value and a value measured at a WB. This classification is based on metrics for each BQE which, in the case of phytoplankon, are biomass and abundance expressed as concentration of chlorophyll a and total cell counts of microphytoplankton (>20µm). Composition is also considered in the form of *Phaeocytis* spp blooms, but this metric is not applicable to Portugal (Newton et al., 2008).

However, the values for these metrics in defining class boundaries have initially only been considered for coastal waters and do not include TWs (Carletti & Heiskanen, 2009). Brito et al (2011) have recently redressed this situation by including some modifications to the metrics by combining phytoplankton biomass (90th percentile of chlorophyll a) with the occurrence of phytoplankton booms (number of cells per litre), adjusted to three salinity classes (< 5, >5 to <25, >25). The Rio Mondego is part of the Northern typology for the TWs in Portugal, and has been labelled A1which is equivalent to international type North East Atlantic 11 for purposes of intercalibration (Bettencourt, et al. 2003). Table 1 shows the reference conditions , boundary values and the EQRs for this typology and Table 2 shows that the Mondego has poor to high ecological status for biomass depending on the salinity class. However, much of the Mondego is defined as "highly modified" where the ecological potential should be assessed instead of the ecological status and this would probably be have to be done against different thresholds from those shown in Table 1.

This abstract focuses on the estuary of the Rio Mondego, which is one of only two transitional WBs in the WISER project. In the case of the BQE phytoplankton

Table 1: (modified from Brito et al 2011). Phytoplankton biomass (Chl a µg l-1_ upper table) and bloom frequency (%_ lower table) showing reference conditions, boundary values and EQRs for the salinity classes of the Rio Mondego.

Biomass (Chl a µg l^{-1})	< 5	>5 to < 25	> 25	EQR
REF	*6.67*	*6.67*	*6*	
H/G	*10*	*10*	*9*	*0.67*
G/M	*15*	*15*	*13.5*	*0.44*
M/P	*22*	*22*	*20*	*0.3*
P/B	*33.5*	*33.5*	*30*	*0.2*
Phyto. bloom frequency				
REF	*16.67%*	*16.67%*	*16.67%*	*1.0*
H/G	*33.33%*	*33.33%*	*33.33%*	*0.5*
G/M	*41.67%*	*41.67%*	*41.67%*	*0.40*
M/P	*50.0%*	*50.0%*	*50.0%*	*0.33*
P/B	*58.33%*	*58.33%*	*58.33%*	*0.29*

Table 2: (modified from Brito et al. 2011). Historical data set for chlorophyll a concentration by salinity class for the Rio Mondego (Northern typology).

Salinity classes	Sample nº	Mean Chla (µg l^{-1})	90 percent. Chla (µg l^{-1})
< 5	*37*	*17.8*	*31.4*
>5 to <25	*47*	*13.0*	*24.9*
>25	*52*	*4.2*	*6.8*

(WP 4.1), the objectives are:

1. develop and validate multi-species or assemblage phytoplankton metrics;
2. evaluate the potential use of pigment data in phytoplankton assemblage metrics;
3. evaluate uncertainty or determinations of phytoplankton biomass and community compositions due to temporal and spatial heterogeneity.

There is not sufficient historical phytoplankton data for the Mondego to contribute to Objective 1, but there is sufficient data from field work in September 2009, as well as historical data, to contribute to Objectives 2 and 3.

Materials and Methods

Location

The Rio Mondego drains a hydrological basin of 6670 km^2 on the North -west coast of Portugal, but this study is concerned with the estuarine area that occupies 860 hectares (Fig. 1), with a tidal range of 0.35-3.3 m and a mean flow of 79 m^{3}s^{-1}. The WISER field experiment for uncertainty was carried out in September 2009 at three different sites defined by the salinity classes oligo- (< 5), meso-(>5 to<25), and polyhaline (>25). A more detailed description of the area is provided by Marques et al. (2006).

Sampling

Sampling of parameters for characterisation of the phytoplankton communities, pigments and enumeration of cells, was completed in seven European water-bodies. In the case of the Mondego Estuary, sampling was carried out at the three sites shown in Figure 1, where a further three stations were selected approximately 200 metres apart to identify the contribution of large scale patchiness to uncertainty. At each of these stations, two water samples were taken from just below the surface. In one sample, two aliquots of one litre each were filtered through 47 mm GF/F filters for analysis by spectrophotometric analysis, and a further two for pigment analyses by high-pressure liquid chromatography (see details below_HPLC). A sample for plankton enumeration was separated into three sub-samples (see details below_Lugol), two for counting by one expert and one for counting by another expert. In the case of the other water sample, the same processing was carried out but

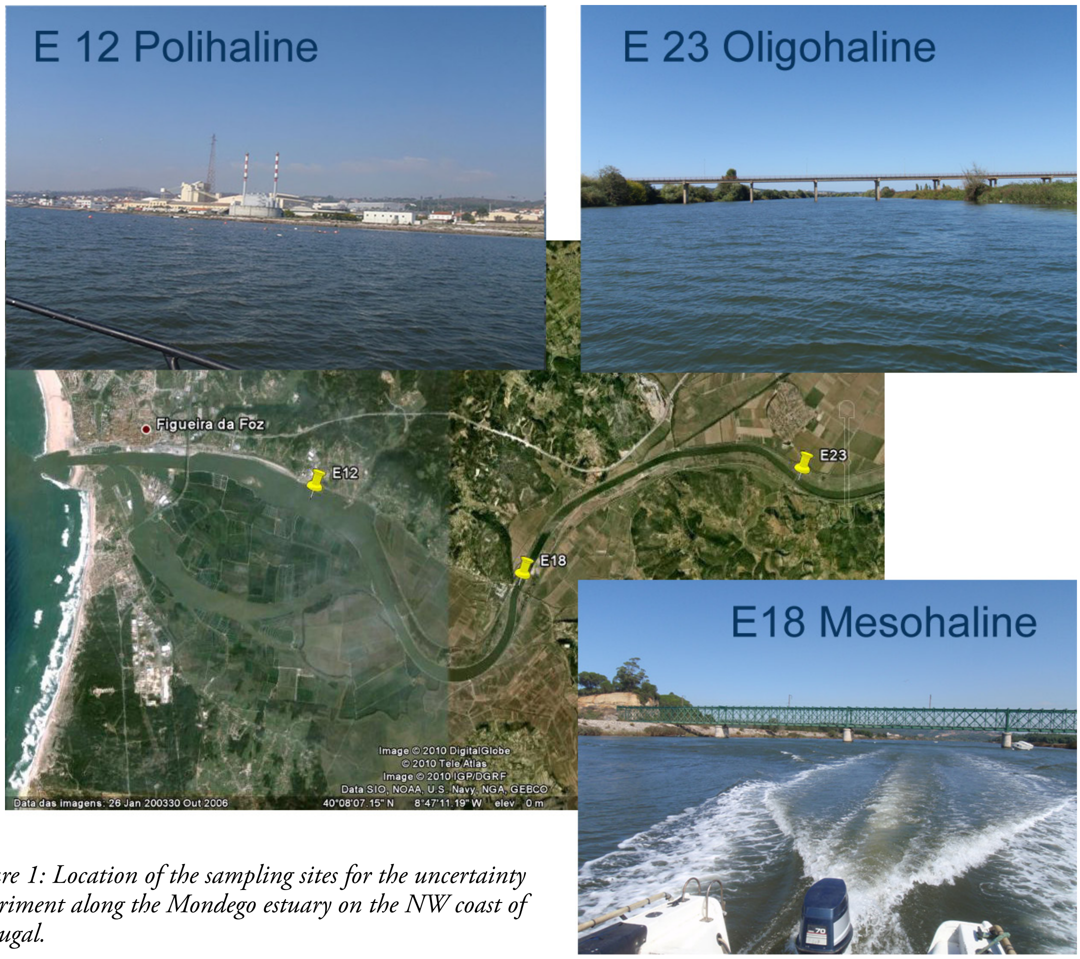

Figure 1: Location of the sampling sites for the uncertainty experiment along the Mondego estuary on the NW coast of Portugal.

with no replicates i.e. only one aliquot for spectrophometry, HPLC, and plankton enumeration by only one expert. Finally, at each station, two additional water samples were taken within approximately 20 metres of the station that were processed only for pigment analysis to assess the contribution of small-scale patchiness to uncertainty.

Pigment analyses

Water samples (150 ml) were filtered onto 25 mm Whatman GF/F glass fibre filters that were immediately frozen under liquid nitrogen before shipping to Denmark on dry ice for analysis. Prior to analysis in Denmark the filters were stored at -80 ºC. Pigments were extracted from 25 mm filters in 5 ml of methanol placed in the dark at -20 ºC. After 24 h, the filters in methanol were sonicated for 15 s on ice and 1 ml of 0.2 µm filtered extract was diluted with 250 µl water in an HPLC vial. Analyses were carried out on a Shimazu LC 10A system with a Supelcosil C18 column (250 x 4.6 mm, 5 µm) using the Wright et al. (1991) method for HPLC. Pigments were identified by retention times and absorption spectra identical to those of authentic standards, and quantified against standards purchased from DHI, Hørsholm, Denmark.

Enumeration of phytoplankton

Sub-samples from replicate water samples for phytoplankton analyses were fixed immediately after sampling with neutral Lugol's solution at5 ml l^{-1} (final concentration ca. 0.5 %). The fixed samples were stored at 4°C in the dark. After storage, 25cm^3 sub- samples were concentrated down in a sedimentation chamber for counts of phytoplankton using the Utermöhl (1958) method with an inverted microscope. A minimum of 300 cells were counted from each sample; rare and large species were checked by scanning the entire counting chamber.

Statistical analysis

The proportion of variation explained by the variance components was estimated by fitting a hierarchical mixed effect model that contained only one fixed effect in the form of an intercept. The variance components: water-body, station, sample, and sub-sample for counting, were estimated as random effects. The hierarchical structure was applied in order to take into account that stations were nested within water-bodies, samples were within stations, and sub-samples were within samples (Singer, 1998). The models were parameterized using the MIXED procedure in the statistical software package SAS/STAT 9.2 (SAS Institute Inc., 2009).

Results and Discussion

With reference to Objective 2, Henriksen et al. (2011) have found that the total concentration of chlorophyll a is significantly correlated with total nitrogen (TN) across the geographically different sampling localities for the WISER sites, but the distribution patterns of pigment samples and communities show a major correlation with salinity and temperature and only minor correlation with TN as a measure of eutrophication . A prerequisite for the use of pigment based community composition as a WFD indicator is the establishment of reference conditions. The major influence from salinity and temperature on the distribution pattern of the WISER samples has hindered the use of any of these sampling stations as reference sites for a pigment based phytoplankton indicator. Furthermore, although the concentration of individual pigments increased with increasing TN, these could not be readily related to any specific plankton species or group of species (Table 3).

Table 3: Percentage variation explained by selected variance components for pigments from Mondego Estuary.

Obs	Pigment	Station	Sample	Sub Sample	Residual
1	*Chl_a*	*51.07*	*46.56*	*0.11*	*2.26*
2	*Chl_b*	*39.85*	*39.65*	*0.00*	*20.50*
3	*Chl_c1_C2*	*55.01*	*43.43*	*0.12*	*1.44*
4	*Diadinoxanthin*	*89.83*	*5.50*	*0.00*	*4.67*
5	*Diatoxanthin*	*51.92*	*29.99*	*0.00*	*18.09*
6	*Fucoxanthin*	*56.37*	*40.50*	*0.00*	*3.13*
7	*Lutein*	*62.70*	*32.28*	*0.00*	*5.02*
8	*Neoxanthin*	*0.00*	*99.70*	*0.00*	*0.30*
9	*Peridinin*	*5.16*	*93.84*	*0.00*	*1.01*
10	*Zeaxanthin*	*61.82*	*25.35*	*0.00*	*12.83*
11	*alfa_carotene*	*60.17*	*26.60*	*0.00*	*13.23*
12	*beta_carotene*	*47.04*	*36.31*	*0.00*	*16.65*

The observations from WISER sites for the relationship between chlorophyll and nitrogen where confirmed by Brito et al (2011) for the Mondego where they demonstrated a significant relationship between these parameters (Fig 2).

With reference to Objective 3, Dromph et al. (2011) have assessed the variation in pigment concentrations and population densities attributed to water-body, station and sample levels at the seven European water-bodies from the WISER project. The pigment data (Tables 3, 4) shows that the main proportion of the variation between measurements is explained by the variation between stations (10-68% of variation) followed by the variation between water-bodies (6-52% of variation). While for measurements of population density, the main proportion of the variation between

Table 4: (from Dromph et al., 2011). Estimated percentages of total variance for the number of taxa and total cell counts in a hierarchical mixed effect model of selected variance components.

Variable	Water Body	Station	Sample	Subsample	Taxonomy	Residual
Number of taxa	*83.07*	*0.00*	*0.00*	*0.00*	*1.55*	*15.38*
Total density	*10.17*	*0.46*	*0.00*	*0.00*	*34.70*	*54.67*

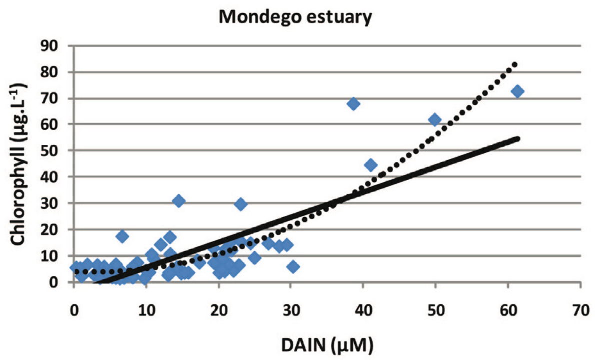

Figure 2 (from Brito et al, 2011). Linear and 2nd order polynomial regressions between chlorophyll and Dissolved Available Inorganic Nitrogen in the Mondego.

cell densities is explained by the variation between the taxonomists counting the samples (55%). In order to improve the precision of estimates for pigment concentrations for a specific water-body, the number of stations should be increased, whilst continuous training and inter-calibration of the staff would improve the assessment of phytoplankton densities by cell counts .

This study demonstrates that extending the phytoplankton metric to pigment based community composition is problematic. It also highlights where errors could contribute to WFD misclassification and, thereby, help to avoid the subsequent legal and economic costs.

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Lake management, restoration and the impact of global climate change

Erik Jeppesen, Martin Søndergaard, Lone Liboriussen & Torben L. Lauridsen

Department of Bioscience, Aarhus University, Vejlsøvej 25, PO Box 314, DK-8600 Silkeborg, Denmark

Email address of corresponding author: ej@dmu.dk

Key words: eutrophication, recovery, fish assemblage change, cascading effects, cyanobacteria , adaptation

Eutrophication and recovery from eutrophication

Freshwater lakes provide water for consumption and irrigation, constitute valuable food sources, are used for recreational activities, and add to biodiversity on Earth. For 50-100 years eutrophication has posed the most serious threat to lakes worldwide. High nutrient loading has resulted in turbid water, excessive blooms of nuisance cyanobacteria, dominance of coarse fish and loss of biodiversity. In recent years major efforts have taken in the rest of the world to combat eutrophication. The measures applied include treatment or diversion of sewage and numerous actions to reduce diffuse loading. Lakes often respond slowly to reductions of external nutrient loading, which may be due to release of phosphorus stored in the sediment during the eutrophication period. The typical 10-15 years recovery period is to be expected although it may last much longer in some cases.

To re-enforce recovery several physico-chemical and biological methods have been used. These include sediment removal, aluminium treatment of sediment and biomanipulation (fish removal, restoring plant communities). Such measures have provided short-term and often considerable improvements in ecological state, but the long-term perspectives are less clear. So far, sediment removal seems to have the highest probability of producing a long-term effect, while other methods need further refinement. In Europe removal of planktivorous fish (mainly roach, *Rutilus rutilus*, and bream, *Abramis brama*) has commonly been used as a method to improve the ecological quality of lakes for the past 10-15 years. We analyze the general and long-term effects obtained after the removal of 40-1,360 kg fish ha^{-1} in 36 mainly shallow and eutrophic Danish lakes. When less than 200 kg fish ha^{-1} were removed within a 3-year period only minor effects were observed, but at higher removal marked effects could be traced on both chemical and biological variables. The concentrations of chlorophyll, total phosphorus, total nitrogen and suspended solids decreased to 50-70% of the level prior to the removal. The most significant and long-lasting effects were found for suspended solids and Secchi depth, while the lowest and most short-lived effects were seen for chlorophyll a, probably reflecting an efficient and persistent reduction of the bream stock and, with it, reduced resuspension, while the biomass of roach sooner returned to former levels. Total algal biomass also declined after fish removal, particularly that of cyanobacteria, whereas the biomass of cryptophytes increased, indicating enhanced grazing pressure by zooplankton. The abundance and species numbers of submerged macrophytes increased in a majority of the lakes. For most variables the effects of the fish removal were significant for 6-10 years, after which many lakes tended to return to pre-restoration conditions, probably mainly because of consistently high external and internal phosphorus loading. Promising results on the short-term at least have been obtained with various chemical treatment of sediment, including various Aluminium treatment methods and the somewhat more expensive phoslock method. However, the long term effects are also here somewhat disappointing. It is therefore likely that both biomanipulation and chemical treatment need to be repeated several times during a prolonged period and maybe combined methods turns out to be most robust and cost-effective (e.g. combing chemical treatment of the sediment with biomanipulation).

Effects of global change and global warming

Global change and global warming put a pressure on the ecological state of lakes. A major increase in the world´s population in this century will increase the need for agricultural production and water use. This will in lead to higher nutrient loading in agricultivated areas and thus eutrophication. Moreover, global warming show enhanced symptoms of eutrophication. Changes in precipitation affect nutrient loading to streams and lakes. Nitrogen and phosphorus loading is expected to increase in north temperate coastal streams, not least during winter and reduced in warm temperate and arid streams. Despite reduced loading in arid systems nitrogen and phosphorus concentrations may increase due to lower water table in both lakes and streams and higher removal of the lower oxygen pool in the water. Climate change may have profound effect on phosphorus processes in lakes. Higher temperature enhances phosphorus release from sediment due to enhanced oxygen consumption and consequently reduced redox conditions, which in turn result in release of iron bound phosphate. Major shift in fish assemblages is also to be expected with cascading negative effects on the ecological state of lake ecosystem. A recent analysis of 24 European long-term fish data series has shown profound changes in fish assemblage composition, size and age structure 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 in several of the lakes. Vendace, whitefish and smelt has also been affected but to various degree depending on lake depth and latitude. Perch was apparently stimulated in the north, with stronger year classes in warm years, but its abundance has declined in the south. Where introduced, roach now seems to take advantage of the higher temperature after years of low populations. Eurythermal species such as bream, pikeperch and shad are on the increase. Moreover mean size of fish has decreased in several lakes, again contrasting what is expected when nutrient loading is reduced. The shift in fish assemblages in direction of small and abundant plankti-benthivorous also enhances predator control of zooplankton, which in turn increase phytoplankton production and sedimentation, and thus risk of P release from the sediment. Multiple regression analysis of late summer data from 800 Danish lakes indicates that warming will lead to increased chlorophyll a and phytoplankton biomass, higher dominance of dinophytes and cyanobacteria, most notably of nitrogen fixing forms, but lower abundance of diatoms and chrysophytes, reduced size of copepods and cladocerans, a tendency to reduced zooplankton biomass and zooplankton:phytoplankton biomass ratio. Consequently, these changes result in increased turbidity, which among other effects will impoverish the growing conditions of submerged macrophytes. Studies of lakes along a latitude gradient also provide evidence for the shift in trophic structure and increasing dominance of cyanobacteria when lakes get warmer.

Adaptation

With increasing warming and the expected global development it will therefore become more difficult to meet the present-day targets set for the ecological state of Northern European lakes according to the European Water Framework Directive. Additional efforts must be initiated to reduce the external nutrient loading to levels lower than present-day recommendations. This calls for adaptation measures , which in the north temperate zone, should include less intensive land use in catchments with sensitive freshwaters to reduce diffuse nutrient inputs, re-establishment of riparian vegetation to buffer nutrient transfers to streams and rivers; improved land management to reduce sediment and nutrient export from catchments; improved design of sewage works to cope with the consequences of flood events and low flows in receiving waters and, where appropriate, re-establishment of lost wetlands, riparian buffer zones and re-meandering of channelized streams to increase retention of organic matter and nutrients.

Performance of profundal macroinvertebrate assessment in boreal lakes depends on lake depth

Jussi Jyväsjärvi[1], Jukka Aroviita[2]* and Heikki Hämäläinen[1]

[1] University of Jyväskylä, Department of Biological and Environmental Science, P.O. Box 35, FIN-40014 Jyväskylä University, Finland
[2] Finnish Environment Institute, Freshwater Centre, Monitoring and Assessment, P.O. Box 413, FI-90014 Oulu University, Finland

* Email address of corresponding author: jukka.aroviita@ymparisto.fi

Key words: bioassessment, ecological status, reference condition approach, shallow lakes, Water Framework Directive

Introduction

Profundal macroinvertebrates (PMI) are widely used in ecological assessment of lakes. However, research have primarily based on data from large and deep stratified lakes with distinctive pelagic and profundal zones, whereas small and shallow lakes have received much less attention despite their abundance in the boreal region. There is thus an obvious need to evaluate procedures for their PMI assessments, particularly given the demands set by modern water legislatives like the Water Framework Directive (WFD).

We investigated the effect of lake depth on biological metrics indicative of the quality status of the PMI fauna. We compared the degree of natural variability of PMI assemblages and their sensitivity to detect human impact among groups of shallow, intermediate, deep and very deep boreal lakes. We used multivariate analyses to examine the compositional differences between groups of minimally disturbed reference lakes and eutrophication impacted lakes, and measured the status of PMIs with three community metrics corresponding to WFD criteria.

Material and methods

We used previously published data on macroinvertebrate communities and environmental variables from 255 Finnish lake basins. The data are from multiple sources, including the HERTTA database of SYKE (Finnish Environment Institute). Macroinvertebrates were collected once from the deepest point of each lake basin in September–October between 1989 and 2008 using standard methods. Environmental data of geographic, water chemistry and morphometric variables for each macroinvertebrate sampling site and occasion were mainly compiled from the HERTTA database (Table 1).

Using data on anthropogenic pressures, the lake basins were categorised as reference (hereafter REF, N=114) sites with minimal human influence. The remaining

Table 1: Mean values of geographic, morphometric and water chemistry variables for minimally disturbed reference sites (REF) and sites impacted by human activity (IMP) in each depth category.

	Shallow		Intermediate		Deep		Very deep	
	REF (N=25)	IMP (N=58)	REF (N=30)	IMP (N=30)	REF (N=32)	IMP (N=28)	REF (N=27)	IMP (N=25)
Altitude (m.a.s.l.)	*142.2*	*104.5*	*114.0*	*104.5*	*109.6*	*83.6*	*87.3*	*79.7*
Surface area (km²)	*3.1*	*9.9*	*9.2*	*15.9*	*50.7*	*175.7*	*437.7*	*493.9*
Latitude	*62.8*	*62.4*	*62.1*	*62.2*	*62.3*	*62.5*	*62.1*	*62.1*
Longitude	*27.2*	*27.1*	*26.7*	*26.8*	*27.5*	*26.8*	*27.7*	*26.7*
Mean depth (m)	*3.2*	*2.9*	*5.0*	*5.0*	*7.3*	*8.2*	*12.6*	*13.3*
Max. depth (m)	*8.7*	*7.7*	*15.7*	*15.5*	*28.0*	*31.7*	*50.6*	*42.5*
Colour (mg Pt L^{-1})	*78.7*	*108.4*	*51.1*	*95.3*	*37.4*	*52.3*	*28.9*	*51.3*
Total P (µg L^{-1})	*16.5*	*43.4*	*10.5*	*27.9*	*7.8*	*21.1*	*6.5*	*15.6*
Total N (µg L^{-1})	*418.7*	*734.9*	*355.5*	*608.6*	*348.0*	*659.1*	*398.2*	*557.0*
Conductivity (µS L^{-1})	*3.0*	*6.7*	*4.0*	*5.6*	*4.8*	*6.1*	*4.8*	*6.1*
Chl. a (µg L^{-1})	*9.9*	*18.8*	*5.6*	*13.5*	*3.9*	*7.5*	*3.6*	*7.2*
Hypol. temp. (°C)	*13.4*	*14.6*	*9.0*	*11.3*	*7.9*	*9.4*	*7.1*	*8.2*
Diss. oxygen (mg L^{-1})	*3.2*	*4.9*	*4.0*	*3.2*	*5.3*	*4.7*	*7.9*	*6.9*

sites were assigned to impacted (IMP, N=141) sites' group subject to a range of anthropogenic disturbances, mainly nutrient enrichment. To address our study question, we constructed a multivariate regression tree (MRT, Death 2002, Jyväsjärvi et al. 2011) using lake mean depth (the primary natural environmental driver of PMI fauna; Jyväsjärvi et al. 2009), as the sole predictor to divide the REF sites into four biologically meaningful depth categories with balanced number of sites (Figure 1).

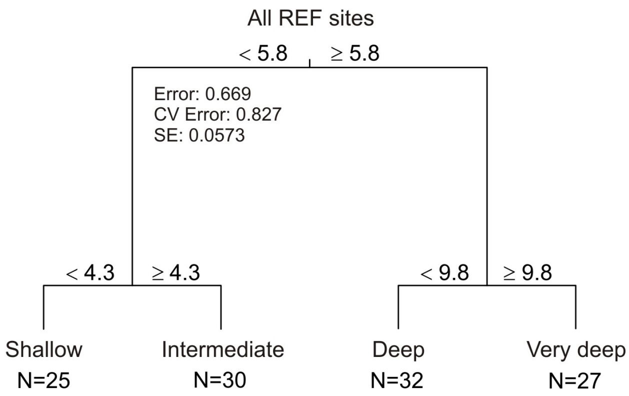

Figure 1: Multivariate regression tree (MRT) showing mean depth threshold value for each node and the numbers of REF sites in each terminal leaf.

The WFD requires that the deviation of the 'ratio of disturbance-sensitive taxa to insensitive taxa', the 'level of diversity' and the 'taxonomic composition and abundance' from undisturbed type-specific reference conditions is used in assessment of lake benthic invertebrates. We selected three metrics that each principally corresponds to one of the structural features, to assess the status of PMI fauna. To measure the ratio of disturbance sensitive taxa to insensitive taxa, we used Benthic Quality Index (BQI) (Wiederholm 1980), that has been commonly used in Europe to indicate nutrient enrichment. To measure level of diversity, we used the Shannon index (H´). To measure the taxonomic composition and abundance, we used Percent Model Affinity (PMA; Novak & Bode 1992), which compares the observed taxonomic composition (relative abundances of taxa) in a site to the taxonomic composition of a reference (model) assemblage.

For each metric and site, we estimated the deviation from reference conditions as the ratio of observed (O) value to the expected (E) value (O/E, i.e. the Ecological Quality Ratio). For BQI, site-specific E values were derived with a regression model of Jyväsjärvi et al. (2010). For Shannon H´ and PMA, E values were derived as depth category specific mean values among REF sites.

To illustrate the compositional differences between REF and IMP sites, we performed separate Detrended Correspondence Analyses for each depth category, and tested the differences with Analysis of Similarities (ANOSIM). Further, coefficients of variation (CV %) were used to assess the normalized amount of variation in the three assessment metrics among the REF sites, and signal to noise ratio (S/N) was to parameterize the power of each metric to detect human-induced impairment of PMI fauna.

Results

The lake mean depth MRT-categories partitioned PMI assemblages into four compositionally dissimilar groups (ANOSIM $R = 0.214$, $p = 0.001$ for all sites; $R = 0.348$, $p = 0.001$ for REF sites). Within each category, the DCA ordinations indicated a marked overlapping of REF and IMP sites in the shallow and intermediate lakes, whereas the separation of REF and IMP sites in ordination space was more distinct in deep and very deep sites (Figure 2). Accordingly, the communities of shallow REF and IMP sites did not differ from the REF communities ($R = -0.003$, $p = 0.512$). IMP communities in intermediate lakes were weakly dissimilar compared to the REF communities ($R = 0.07$, $p = 0.027$), whereas assemblages of deep and very deep IMP sites differed significantly from the REF communities ($R = 0.178$, $p = 0.001$ and $R = 0.102$, $p = 0.001$, respectively).

Two (O/E_{BQI} and $O/E_{SHANNON}$) of the three metrics had large variation among shallow REF lakes, and IMP sites

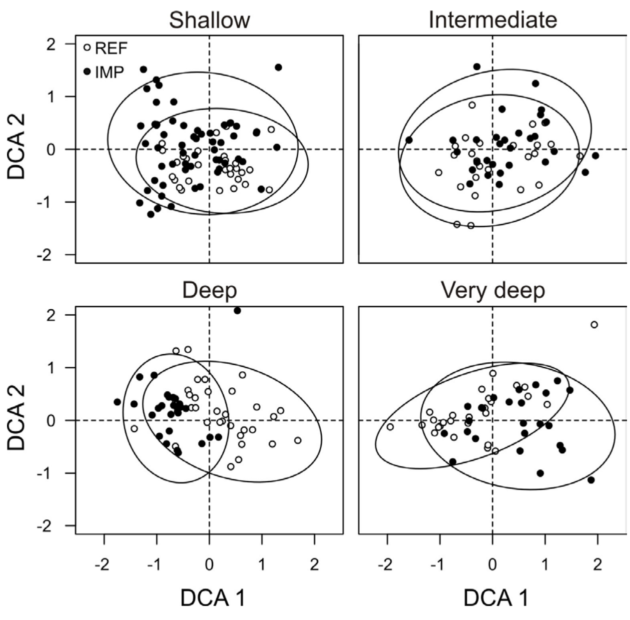

Figure 2: DCA ordinations within the four mean depth categories. Circles and black dots denote REF and IMP sites, respectively. Ellipses indicate 95 % confidence intervals of status groups.

differed from REF sites only by O/E_{PMA} (Figure 3). The REF variation of O/E_{BQI} and $O/E_{SHANNON}$ was smaller in intermediate lakes and smallest in deep and very deep lakes. For deep and very deep sites, all metrics also showed significant difference between REF and IMP sites.

Discussion

The results suggest a poor performance of profundal macroinvertebrate assemblages in ecological assessment of shallow boreal lakes. In comparison to deeper lakes, the assemblages and the assessment metrics among shallow REF lakes were more variable and sites impacted mainly by anthropogenic nutrient enrichment were poorly separable from the reference conditions. Furthermore, both community variation and assessment metrics showed weak or no response to nutrient status among shallow lakes. A partial explanation may be a strong association of profundal macroinvertebrate community structure with lake depth among minimally disturbed boreal lakes (Jyväsjärvi et al. 2009). Profundal taxa tolerant of eutrophy (e.g. *Chironomus plumosus* and *Chaoborus flavicans*) are typical inhabitants of shallow near-pristine boreal lakes. Therefore, it is unlikely that the profundal assemblages that are naturally dominated by these taxa are altered by even considerable degree of eutrophication.

In small and shallow boreal lakes macroinvertebrate sampling stations in main basin are usually in the vicinity of shoreline and may not fully represent pelagic and profundal conditions. Indeed, the offshore macroinvertebrate assemblages in shallow lakes are often a heterogeneous compilation of littoral, sublittoral and profundal species (e.g. Hämäläinen et al. 2003, Jyväsjärvi et al. 2009). Moreover, the generally small surface area and elevated concentration of humic compounds make these lakes particularly prone to strengthened thermal stratification (Jones 1992) and consequent hypolimnetic oxygen depletion (Fulthorpe & Paloheimo 1985). On the other hand, shallow lakes with less sheltered

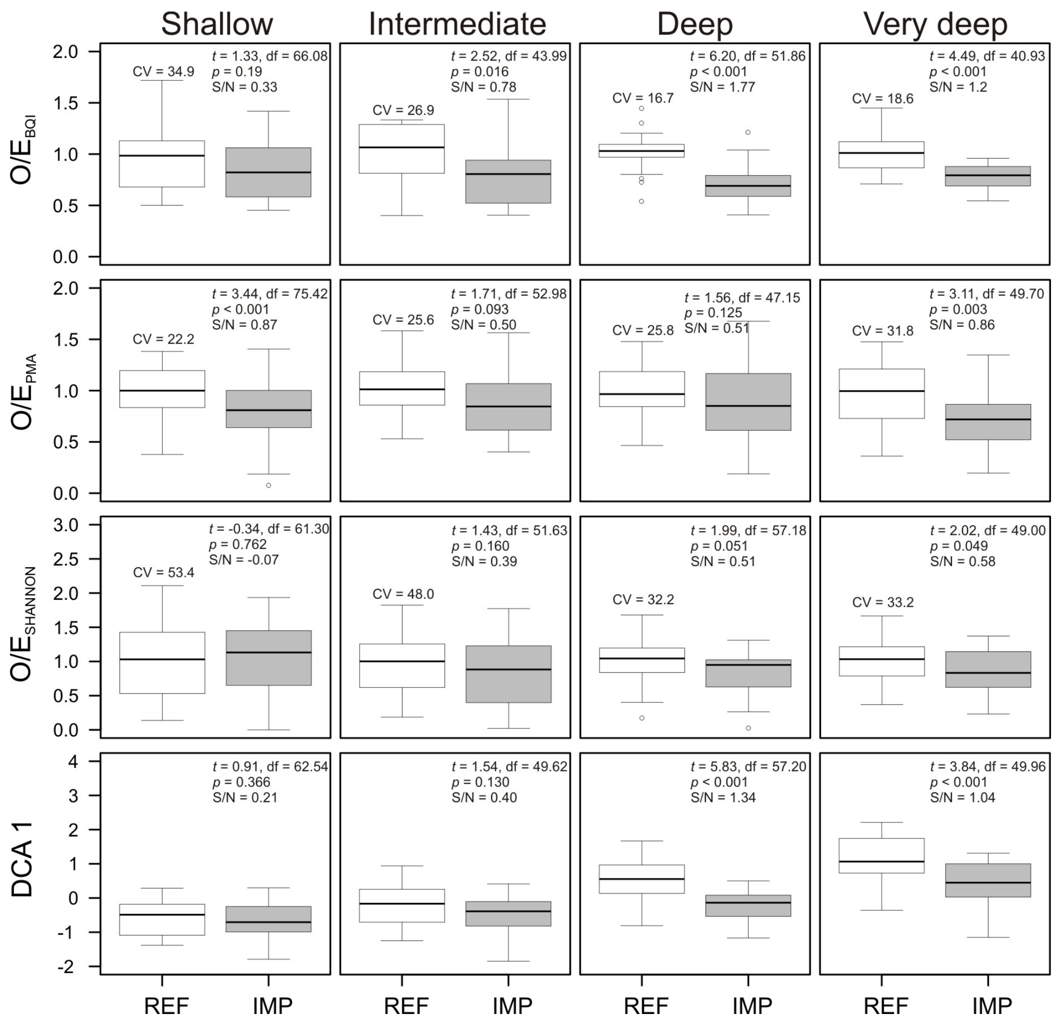

Figure 3. Variation of the three assessment metrics and first DCA axis scores among REF and IMP sites in each depth category. The boxes show 25 and 75 percentiles, thick lines median, whiskers range and open circles outliers. In each panel, coefficient of variation (CV) for the REF sites is shown above the boxes and t-test statistics and signal/noise -ratios.

location are more susceptible to wind-induced temporal mixing of water column followed by the lack of or weakened thermal stratification (Gorham & Boyce 1989), fluctuating nutrient status (Søndergaard et al. 1992) and increased resuspension of lake sediments (Evans 1994). Consequently, the factors which drive the natural variation of true profundal fauna (Jyväsjärvi et al. 2009) and also cause faunal impairment due to anthropogenic disturbance (i.e. increase in organic sedimentation and dissolved oxygen consumption) tend to vary stochastically among shallow lakes. This might lead to the observed high natural faunal variation and thereby hinder human impacts or their recognition.

Conclusions

The results of this study indicate obvious difficulties in using profundal macroinvertebrate assemblages in the ecological assessment of shallow boreal lakes. The assemblages of deeper boreal lakes responded predictably to anthropogenic nutrient enrichment, whereas assemblages of shallow lakes are either 'naturally eutrophic' and thus resistant to change or unpredictably variable making it difficult to detect any impacts of anthropogenic nutrient enrichment on communities. Future studies are thus needed to evaluate the performance of littoral macroinvertebrate assemblages (e.g. White & Irvine 2003, Aroviita & Hämäläinen 2008) for shallow lake bioassessment.

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Factors influencing macrophyte metrics in Estonian coastal lakes in the light of ecological status assessment

Katrit Karus, Tõnu Feldmann

Centre for Limnology, Estonian University of Life Sciences, 61117 Rannu, Tartu, Estonia

Email address of corresponding author: katrit.karus@emu.ee

Key words: aquatic vegetation, coastal lakes, ecological status, macrophyte-based assessment, environmental factors

Summary

Aquatic macrophytes and factors affecting their distribution were studied in 19 shallow coastal lakes of Estonia in the years of 2009-2011. Morphological, physico-chemical and catchment area characteristics of the studied lakes varied to a large extent. The analysis revealed that the numbers of several macrophyte species were strongly correlated with morphometric and catchment area variables while the abundance of the emergent and submerged species correlated with physico-chemical variables such as conductivity, sulphates and chlorides. Factor analysis of the abiotic variables divided them between three factors – 1) parameters of catchment area, 2) morphometric variables, and 3) water characteristic. In the ecological status assessment of coastal lakes all three aforesaid factors must be considered.

Introduction

Most of the Estonian coastal lakes are located in the western part of the Baltic Sea coast of Estonia constituting 5.2% of the total number of the local lakes (Tamre et al., 2008). These lakes are the former gulfs still partially connected to the Baltic Sea or totally isolated from it as a result of the postglacial land uplift of up to 2.8 mm y-1 as estimated for western Estonian mainland and islands (Vallner et al., 1988). The age of the present coastal lakes is mostly between 5000-10 000 years (Tamre et al., 2008) although the formation of coastal lakes is still active due to land uplift, which leads to the separation juvenile coastal lakes from the sea. The macrophytes of Estonian coastal lakes and the factors influencing their distribution have still been poorly investigated. Nevertheless several macrophyte metrics such as the abundance of *Chara aspera* Willd, *Chara tomentosa* L., *Utricularia vulgaris* L. and *Cladium mariscus* L. have been developed to assess the ecological status of the lakes. Unfortunately some of these quality indicators are often missing in the coastal lakes.

We assume that the physico-chemical, morphological and catchment area variables have a strong effect on the distribution and composition of macrophytes in coastal lakes because abiotic variables change over a large scale depending on the development stage of lakes determined by the connection with and the distance from the sea.

Material and methods

Description of the study areas

Surveys of 19 coastal lakes were conducted from July to August in three successive years 2009, 2010 and 2011. Among the lakes 7 were located in the mainland of West-Estonia, 1 on the Vormsi Island and 11 on the Saaremaa Island. The surface area of the studied lakes ranged from 0.07 to 5.31 km^2. The lakes were very shallow with maximum depths between 0.5 and 2.1 m. In general, coastal lakes are characterized by high pH (>9) and high contents of sodium, chloride, sulphate and halogen ions. Based on this study, some of the physico-chemical variables ranged widely, for instance conductivity from 285 to 7586 μS/cm, chlorides (Cl^-), from 5.2 to 2200 mg/l, and sulphates (SO_4^{2-}), from 12 to 425 mg/l. A more detailed overview of the morphological and physico-chemical characteristics of the investigated lakes is given in Table 1.

Sampling design and collection of data

Submerged macrophytes were investigated from July to August in the years 2009-2011 using a transect method in combination with phytolittoral mapping. The more developed the shore stretch, the more profiles were investigated. In each profile, which started from the water line and reached to the maximum depth of macrophytes occurrence, we registered the taxonomic composition, abundance of emergent, floating, floating-leaved and submerged plants and measured their colonization depth (Table 1). The abundance of the species was based on Braun-Blanquet (1964) scale that was modified by condensing it to five points. Species abundances were estimated separately in three groups: emergent, floating and floating leaved plants, and submerged plants.

Among catchment area characteristics (Table 1) we determined the proportion of fields and forests, the number of stock-raising buildings (SRB) around the lakes and the shortest distance to the sea.

Statistical analysis

Data were analysed using Spearman's correlation analysis and factor analysis offered by STATISTICA 8.0 (StatSoft, Inc. 2007). Spearman's correlation analysis was used to determine the relationships between the macrophyte indices (number of the species among emergent, floating-leaved, floating and submerged plants; abundances of the main species; maximum colonization depth) and the abiotic variables (morphological,

Table 1: Characteristic features of the coastal lakes. Abbreviations: SLD – shore line development, Cond – conductivity, SRB – number of stock-raising buildings, Dist – shortest distance to the sea.

Lake	Year	Surf. area (km^2)	Mean depth (m)	Max depth (m)	SLD	TP (mg/l)	TN (mg/l)	SO_4^{2-} (mg/l)	Cl^- (mg/l)	Cond. (µS/cm)	HCO_3^- (mg-ekv/l)	Catch. (km^2)	Forest (%)	Field (%)	SRB	Dist.
Aenga	*10*	*0.17*	*0.9*	*1.5*	*1.73*	*0.062*	*1.90*	*300*	*1725*	*6560*	*1.5*	*0.80*	*27.50*	*-*	*3*	*95*
Laidevahe	*10*	*1.97*	*1.0*	*1.6*	*5.57*	*0.055*	*1.26*	*210*	*1335*	*4810*	*2.85*	*7.34*	*74.31*	*12.40*	*6*	*27*
Linnulaht	*10*	*0.69*	*0.5*	*2.0*	*2.75*	*0.061*	*2.10*	*35*	*29*	*342*	*1.80*	*4.80*	*47.92*	*-*	*9*	*864*
Mullutu	*10*	*4.13*	*0.9*	*1.7*	*2.71*	*0.003*	*1.20*	*28*	*36*	*358*	*1.85*	*237.85*	*54.10*	*17.00*	*12*	*2191*
Oessaare	*10*	*1.02*	*1.0*	*1.6*	*2.41*	*0.005*	*0.98*	*48*	*13*	*375*	*2.50*	*217.21*	*51.13*	*30.39*	*17*	*2266*
Poka	*10*	*0.19*	*0.4*	*0.6*	*2.06*	*0.036*	*1.60*	*31*	*40*	*458*	*2.30*	*217.21*	*51.13*	*30.39*	*17*	*2063*
Põldealune	*10*	*0.31*	*0.8*	*1.5*	*1.65*	*0.035*	*1.50*	*100*	*1000*	*3980*	*2.00*	*1.10*	*49.09*	*26.36*	*3*	*596*
Suurlaht	*10*	*5.31*	*1.2*	*2.1*	*1.98*	*0.020*	*1.00*	*31*	*39*	*349*	*1.65*	*11.47*	*57.63*	*7.76*	*9*	*2096*
Vägara	*10*	*0.84*	*0.6*	*1.1*	*2.46*	*0.022*	*1.50*	*12*	*30*	*306*	*1.95*	*64.61*	*58.44*	*9.61*	*8*	*2201*
Mõisalaht	*09*	*0.76*	*0.8*	*1.9*	*3.14*	*0.047*	*0.71*	*220*	*2390*	*6925*	*2.68*	*30.20*	*28.44*	*41.99*	*7*	*17*
Undu	*09*	*2.27*	*1.0*	*2.0*	*3.13*	*0.028*	*0.74*	*-*	*-*	*2120*	*2.78*	*7.74*	*43.80*	*18.09*	*14*	*14*
Kahvatu	*11*	*0.07*	*0.3*	*0.5*	*1.65*	*0.049*	*1.88*	*25*	*163*	*285*	*2.50*	*0.38*	*44.74*	*26.32*	*7*	*239*
Kiissa	*11*	*0.27*	*0.4*	*0.5*	*2.37*	*0.034*	*1.35*	*135*	*406*	*1772*	*1.35*	*16.20*	*43.64*	*37.78*	*1*	*321*
Kudani	*11*	*0.12*	*0.3*	*0.8*	*3.25*	*0.020*	*1.67*	*25*	*26*	*362*	*2.05*	*1.10*	*44.55*	*-*	*0*	*1206*
Käomardi	*11*	*0.15*	*0.3*	*0.5*	*1.95*	*0.045*	*1.70*	*145*	*315*	*1475*	*1.65*	*17.10*	*49.12*	*43.86*	*1*	*328*
Kasse	*11*	*0.73*	*0.3*	*0.5*	*2.06*	*0.036*	*1.70*	*85*	*20*	*419*	*2.40*	*31.94*	*5.85*	*43.49*	*3*	*320*
Prästvike	*11*	*0.38*	*0.3*	*0.5*	*1.76*	*0.014*	*1.03*	*30*	*5.2*	*332*	*3.00*	*11.40*	*67.63*	*13.60*	*5*	*661*
Vööla	*11*	*0.68*	*0.4*	*1.0*	*2.02*	*0.046*	*1.68*	*425*	*2200*	*7586*	*2.00*	*10.32*	*42.25*	*7.75*	*1*	*379*
Kooru	*11*	*0.84*	*0.3*	*1.2*	*3.57*	*0.010*	*0.73*	*32*	*7.3*	*282*	*2.00*	*36.55*	*72.48*	*6.35*	*2*	*584*

Table 2: Spearman correlations between the abiotic and biotic variables ($p<0.05$). Abbreviations: number of emergent species (A), floating species (B) and submerged species (C); maximum abundance of Cladium mariscus (L.) Pohl (Cla. mar), Potamogeton pectinatus L. (Pot. pec), Najas marina L. subsp. intermedia (Wolfg. ex Gorski) Casper) (Naj. int), Bolboschoenus maritimus (L.) Palla (Bolb. mar), Schoenoplectus tabernaemontanii (C. C. Gmel.) Palla (Scho. tab) and Utricularia spp. (Utr. spp); nearer straight distance from the sea (Dist) and maximum colonization depth (Max col. depth).

Abiotic/biotic variables	A	B	C	Clad. mar	Pot. pec	Naj. int	Bolb. mar	Scho.tab	Utr .spp	Max col. depth
Surface area	*0.55*	*-0.50*								*0.71*
Mean depth	*0.49*	*-0.63*							*-0.56*	*0.73*
Max depth		*-0.80*								*0.90*
SLD					*-0.52*					
Transparency		*-0.75*				*0.52*				*0.87*
TN					*0.46*					
TP				*-0.59*						
HCO_3^-										
SO_4^{2-}				*-0.51*			*0.71*		*-0.53*	
Cl^-							*0.53*		*-0.54*	
Cond.							*0.66*	*0.47*	*-0.57*	
Dist.				*0.49*			*-0.50*	*-0.56*		
SRB	*0.54*		*-0.47*							
Forest (%)	*0.51*								*0.48*	
Field (%)				*-0.54*						*-0.50*

physico-chemical and catchment area variables of the lakes). Factor analysis was performed using the rotation method 'Varimax normalized' for describing the variability of the macrophyte indices.

Results

The coastal lakes studied in the years 2009-2011 were characterized by well-developed reed beds and dense submerged vegetation. Spearman correlation analysis revealed that the number of species and the maximum colonization depth of macrophytes were mainly correlated with the geo-morphological and catchment area variables, whereas the abundance of different plant species was strongly correlated with the physico-chemical variables, such as sulphates, chlorides, conductivity and less strongly with total phosphorus and nitrogen (Table 2).

Factor analysis divided the abiotic variables into three factors (Table 3), which explained 76.1% of the total variability.

Table 3: 'Varimax normalized' rotated factor loadings of the variables (abs>0.7) and the direction of their influence (+ or -).

Variable	Factor 1	Factor 2	Factor 3
Surface area		+	
Mean depth		+	
Max depth		+	
Transparency		+	
HCO_3^-			
SO_4^{2-}			+
Cl^-			+
Conductivity			+
BSR	+		
Forest	+		
Field	+		
Catchment area	+		
Precent of variability	*28.4*	*24.9*	*22.8*

The variables were divided between these three factors so that the catchment area variables assembled to Factor 1, the morphometric variables of the lakes to Factor 2 and the water characteristics to Factor 3.

In conclusion, the distribution of macrophytes in the coastal lakes is affected by three large groups of variables characterising the catchment area, the morphology of the lake basin, and the water chemistry. Assessing the ecological status of coastal lakes we must consider all the three aforesaid factors. It is especially crucial, given that some of these factors restrict significantly the distribution of some species that are used in the ecological state assessment.

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Assessing the state of UK upland lakes and streams: comparison of recovery trajectories and threat

M. Kernan, G. Simpson. & C. Curtis

UCL, University College London, Gower Street, London, England

Email address of corresponding author: m.kernan@ucl.ac.uk

Biological recovery in acid and acidified lakes and streams in the UK: 1988-2009

Recent analyses of 20 years of chemical and biological monitoring data from 22 lake and stream sites from the UK Acid Waters Monitoring Network has demonstrated that there have been significant improvements in water quality in response to the major reductions in emissions and deposition of sulphur and nitrogen across the UK. All previously acidified sites have shown increases in Acid Neutralising Capacity and the concentration of non-marine sulphate has fallen substantially across the network. pH has increased at most acidified sites and toxic labile aluminium concentrations have decreased at the most severely acidified sites. This chemical recovery has elicited a biological response. There has been a clear change in diatom species composition over the last 20 years at all acidified sites, consistent with the increases in pH. New acid-sensitive aquatic plant species have appeared in seven of the lake sites and four of the stream sites. Benthic invertebrates exhibit significant temporal trends at about half of the sites and the shifts in assemblage composition that have taken place are those expected as a result of reduced acidity. There are new populations of brown trout at three of the most acidified sites in the network. However, the biological recovery is still limited and the trends are not consistent across the UK uplands. Here we examine the chemical and biological trends on a site by site basis to examine specifically how the biology is responding to trends in chemistry over the 20 year monitoring period for each site. We examine whether the different organism groups are responding in a similar way at each site and whether there are geographical patterns in the response and differences between stream and lake sites. We then use co-correspondence analysis to examine whether there is a coherent biological response across all organism groups. The results will provide some insight into whether biological recovery at specific sites is being confounded by other factors such as climate change, land-use change, atmospheric nutrient enrichment or variations in catchment characteristics.

Evaluating taxonomic composition macrophyte metrics for assessment of eutrophication in Europe– searching for the best responding common metric

Agnieszka Kolada[1], Bernard Dudley[2], Nigel Willby[3], Peeter Nõges[4], Martin Søndergaard[5], Seppo Hellsten[6], Marit Mjelde[7], Ellis Penning[8], Gerben van Geest[8], Vincent Bertrin[9], Frauke Ecke[10], Helle Mäemets[4], Katrit Karus[4]

[1] Institute of Environmental Protection (IEP), Kolektorska 4, 01-692 Warsaw, Poland
[2] Natural Environment Research Council Centre for Ecology & Hydrology (NERC)
[3] University of Stirling (UoS)
[4] Estonian University of Life Sciences (EMU)
[5] Aarhus University – Bioscience (AU)
[6] Finnish Environment Institute (SYKE)
[7] Norwegian Institute for Water Research (NIVA)
[8] Deltares (DELFT)
[9] French Research Institute for Agricultural and Environmental Engineering (CEMAGREF)
[10] Swedish University of Agricultural Sciences (SLU)

Email address of corresponding author: akolada@ios.edu.pl

Introduction

The Water Framework Directive (WFD) requires that taxonomic composition of macrophytes, supplementary to abundance, should be included in assessment of the ecological status of lakes (Annex V, EC/2000/60). In existing methodologies different taxonomic composition metrics are used, from the relatively simple ones, such as diversity indices, or proportion of functional groups, to more sophisticated ones based on trophic scores of taxa along a pressure gradient (Birk 2010 and extensive literature cited there).

In many European countries aquatic plants have been aggregated into categories according to their responses to trophic conditions. The lists of taxa being tolerant and sensitive to eutrophication are usually elaborated on the national level (country specific lists) and cannot be applied universally (Schneider 2007; Birk & Willby 2010). Therefore, a more universal approach is needed to develop common metrics that can be used EU wide. The main goal of this study was to indicate a macrophyte taxonomic composition metric responding sufficiently to eutrophication and being applicable in different Geographic Intercalibration Groups in the EU (GIGs, Heiskanen et al. 2004), countries and lake types.

Material and methods

In the common WISER database both, biological and chemical data for over 1500 lake-years from 12 countries were available. For testing the response of macrophyte metrics to eutrophication total phosphorus concentration (TP) was used as a pressure proxy. All the analysed lakes belong to three GIGs (Central-Baltic, Nordic, and Eastern Continental), however the EC GIG was represented by 17 lakes only. The use of presence/absence data in case of all the metrics was decided as the most applicable and universal. Three groups of metrics on taxonomic composition were tested: (i) indices based on trophic scores, (ii) indices based on species richness; (iii) indices based on proportion of functional groups.

Intercalibration Common Metric for lake macrophytes (ICM_LM)

The metric was elaborated by an IC expert (Nigel Willby, UoS) for the purpose of the pan-European intercalibration exercise. For macrophyte taxa a lake trophic rank (LTR) has been derived which grades taxa by their response to pressure, mainly nutrient enrichment. In the WISER macrophyte dataset the LTR for 135 taxa of hydrophytes (including charids, isoetids, elodeids, nymphaeids, bryids and filamentous algae) was indicated ranged from -2.2 for *Tolypella canadensis* to 11.4 for *Lemna minuta*. For all the lakes an Intercalibration Common Metric (ICM_LM) was calculated as an average value of LTRs.

Ellenberger Index (EI)

Since the ICM_LM includes only hydrophytes, and the role of emergent vegetation in lake assessment has been discussed for long, to investigate an impact of including or excluding helophytes in metric response to pressure, the Ellenberg Index (EI) was explored. The index was elaborated based on the trophic score system for vascular plants of central Europe elaborated by Ellenberg (1988, after Hill et al. 1999). The Ellenberg values for nitrogen (N-score) for 241 macrophyte taxa in the common database both hydrophytes and helophytes were available. For 37 aquatic taxa with no Ellenberg value (mainly species from genera: *Callitriche*, *Chara*, *Nitella*, *Tolypella*, *Potamogeton* and *Sparganium*) and with LTR value elaborated, the missing N-scores have been estimated from the LTR-Ellenberg regression equation. For all the lakes in the database the Ellenberg Index was calculated as an average N-score value, both using total number of taxa including helophytes (EI_TT) and only submerged taxa (EI_ST).

Number of taxa (N)

The species richness was expressed as a number of all taxa identified within a lake (N_TT) and the number of taxa submerged only (N_ST). Additionally, the number of sensitive taxa: characeans (N_char) and isoetides (N_iso) was tested.

Proportion of characeans (%_char) and isoetids (%_iso) in total number of submerged taxa

The proportion of taxa from taxonomic groups: characeans (%_char) and large isoetids (%_iso) in a number of taxa submerged was calculated. To determine a growth form for taxa, the common taxa list (available at: http://www.aqplants.ceh.ac.uk) was used as a reference.

The macrophyte metrics were plotted against TP gradient per GIG, country and IC common lake type. The values of determination coefficient $R^2 \geq 0.30$ and Pearson's correlation coefficient $R \geq 0.55$ (for linear relationships), and Spearman correlation coefficient $R_{Sp} > 0{,}60$ (for non-linear relationships) were assumed as criterion for good performance of a metric.

Results

The ICM_LM gave a strong ($R^2=0.52$, $p<0.0001$) and almost linear overall relationship with the pressure variable (logTP) over all lakes analysed. The strength of the ICM_LM: TP relationship differed between countries (Table 1); the metric performed best in Nordic lakes and slightly worse in Central-Baltic lakes (Table 2). No significant relationship between ICM_LM and TP was found for the lakes in the Eastern-Continental GIG. The pressure-response curve (TP vs. ICM_LM) exhibited systematic differences by alkalinity (Alk) indicating at equal TP levels worse status in lakes with higher alkalinity (Fig. 1). This may have implications for the applicability of the metric for lake types with broad alkalinity ranges (e.g. in Central-Baltic GIG). The ICM_LM:TP relationship was stronger (R^2 close to 0.4) in lakes with moderate and high alkalinity, and weaker in less buffered lakes ($R^2=0.26$).

Due to a good performance of the ICM_LM in most

*Table 1. The relationships between macrophyte metrics and TP concentration (μg/L) in lakes in different European countries; Pearson's correlation coefficient R>0.55 in linear regression model (for ICM_LM and EI), and Spearman's correlation coefficient values R_{Sp}>0.60 in non-linear model (for N_TT and N_ST) marked in bold; ns – non-significant at p>0.05; *p<0.05, **p<0.01, ***p<0.001.*

Country	n	ICM (R)	EI (R)	N_TT (RSp)	N_ST (RSp)
UK	54	**0.82*****	**0.69*****	-0.40***	-0.37**
NO	230	0.81***	0.70***	-0.19**	**-0.22****
IE	126	0.67***	0.69***	ns	**ns**
FI	403	**0.58*****	**0.53*****	0.34***	**0.23*****
SE	250	0.56***	0.53***	0.32***	**0.22*****
LV	150	0.50***	0.31***	ns	-0.17*
NL	54	0.45***	ns	-0.52***	-0.54***
PL	175	0.38***	0.48***	-0.32***	-0.48***
EE	35	0.34*	0.34*	ns	ns
BE	7	ns	0.90**	ns	ns
LT	7	ns	ns	ns	ns
RO	17	ns	ns	ns	ns

*Table 2. The relationships between macrophyte metrics and TP concentration (μg/L) in different IC lake types; Pearson's correlation coefficient R>0.55 in linear regression model (for ICM_LM and EI), and Spearman's correlation coefficient values R_{Sp}>0.60 in non-linear model (for N_TT and N_ST) marked in bold; ns – non-significant at p>0.05; *p<0.05, **p<0.01, ***p<0.001.*

IC-GIG/ type	n	ICM (R)	EI (R)	N_TT (RSp)	N_ST (RSp)
ALL	1501	0.72***	0.68***	0.05*	ns
N-GIG	894	0.66***	0.53***	0.28***	0.17***
N1	49	0.71***	0.61***	0.29*	ns
N8	92	0.64***	0.57***	0.27*	ns
N3	155	0.52***	0.53***	0.31***	0.21**
N6	33	0.50**	0.55**	ns	ns
N5	14	ns	ns	0.57*	0.61*
N2a	67	ns	ns	ns	ns
N2b	11	ns	ns	ns	ns
CB-GIG	448	0.58***	0.40***	-0.16**	-0.24***
CB1	207	0.55***	0.39***	ns	-0.18**
CB2	186	0.55***	0.38***	-0.30***	-0.35***
CB3	50	0.56***	0.45**	ns	-0.30*
EC-GIG	17	ns	ns	ns	ns

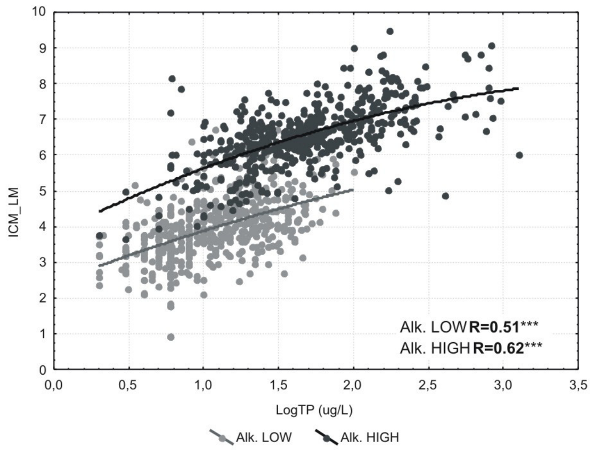

Figure 1. The relationship between ICM_LM and TP concentration in lakes of different alkalinity level (LOW <0.1meq/L, HIGH >2.0 meq/L).

countries and many lake types, it was suggested as a common metric for intercalibration.

The results on testing the Ellenberg Index (EI) calculated on total number of taxa (EI_TT) and only submerged taxa (EI_ST) indicated that including helophytes improved the strength of the general relationship with TP. Therefore, only the EI_TT was explored further. The EI_TT had a similarly high predictive power for TP (R^2=0.47) as the ICM_LM showing it as a well performing metric. Similarly to ICM, the best performance of EI_TT in detecting lake eutrophication was found in NO, UK and IE (Table 1). In most of the remaining countries the diagnostic value of the index was considerably lower or the relationship non-significant. The TP vs. EI_TT relationship was slightly modified by lake depth and alkalinity classes. Strongest relationships were found for shallow and deep lakes (R^2=0.52 and 0.46, respectively) and slightly weaker in very shallow lakes. The Ellenberg Index performed best in Nordic and slightly weaker in Central-Baltic common lake types (Table 2).

Among variables characterizing species richness, the total number of taxa (N_TT) and the number of submerged taxa (N_ST) had a unimodal distribution relative to TP in a pool of all the lakes (Spearman's non-parametric test used). In different countries the correlations with TP were positive, negative or non-significant. In eutrophic lakes the increase in the number of helophyte taxa with increasing TP levels compensated the decrease in the number of submerged taxa. This diminishes the metric diagnostic value. Due to the poor metric response in general (R_{Sp}<0.60 or non-significant at p>0.05 in almost all countries and lake types; Table 1 and 2), the potential use of these metrics for IC purposes is very limited. Since the overall relationships of TP with the number of characeans (N_char) and large isoetids (N_iso), as well as with the proportion of characeans (%_char) and large isoetids (%_iso) over all the lakes were weak (although statistically significant) or non-significant in most of the countries and lake types, these metrics also cannot be considered as promising for the IC purposes as well.

Conclusions

- The best performing metrics were those based on trophic scores – Intercalibration Common Metric (ICM_LM) and Ellenberg Index (EI). They can be recommended in many countries (e.g. UK, NO, IE, FI, SE, LV) and lakes types (mainly lowland, shallow, moderate and high alkalinity) as common metrics for IC purposes.
- The ICM_LM performed better in moderate- and high alkalinity lakes and its use in ecosystems of the alkalinity <0,2 meq/L may be limited.
- The Ellenberg Index was a relatively well performing metric, however its usefulness for detecting eutrophication in different countries and lake types appeared to be lower compared to the ICM_LM. In countries where macrophyte-based assessment methods have not been developed yet or were no trophic scores for local flora are available, the well-known and widely applicable Ellenberg Index can be considered as taxonomic composition component at first.
- The metrics based on taxa richness or proportions of functional groups responded weaker than those based on trophic scores and they cannot be recommended as useful indicators for assessment of eutrophication process.

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The Water Framework Directive and state of Europe's water

Peter Kristensen, Anne Lyche Solheim [2], Kari Austnes [2]

[1] European Environment Agency (EEA), Copenhagen, Denmark
[2] Norwegian Institute for Water Research (NIVA), Gaustadalléen 21, 0349 Oslo, Norway

Email address of corresponding author: peter.kristensen@eea.europa.eu

Key words: Water Framework Directive, WFD, state of water, ecological status, pressures

Europe's waters are affected by several pressures including water pollution, water scarcity, floods; and by major modifications affecting morphology and water flow. The continuing presence of a range of pollutants in a number of Europe's freshwaters threatens aquatic ecosystems and raises concerns for public health. Driven by the EU Urban Wastewater Treatment Directive (UWWTD), improvements in the collection and treatment of wastewater in some regions of Europe have led to a reduction in the discharge of some pollutants to fresh and coastal waters. Challenges remain, however, because UWWTD implementation remains incomplete and other significant sources of water pollution exist, especially agriculture and urban storm flows. Despite improvements in some regions, pollution from agriculture remains a major pressure on Europe's freshwater, causing widespread problems of nutrient enrichment in lakes and rivers. Structures such as dams for hydropower or supplying water for irrigation have resulted in significant hydromorphological modifications – physical changes – to many of Europe's waters. Navigation activities and navigation infrastructure such as cross profile construction – dams, weirs, locks, and impoundments; canalisation; straightening; bank reinforcement and deepening are typically associated with a range of hydromorphological changes with potential adverse ecological consequences.

To maintain and improve the essential functions of our water ecosystems, we need to manage them well. This can only succeed if we adopt the integrated approach introduced in the Water Framework Directive (WFD) and other water policies. Europe has via the WFD and other water policies strong water legislation in place and the challenge now is to see how it works in practice. In March 2010, EU Member States had to deliver their River Basin Management Plans (RBMPs). Each of the 170 RBMPs contains much information and results on the ecological and chemical status of water bodies and the pressures affecting them.

2012 will be the European year of water in which the EU Commission will publish its "Blue-print to safeguard European waters" comprising reviews of the RBMPs, water scarcity and drought and vulnerability and adaptation policies. To accompany and inform these events and policy processes the European Environment Agency (EEA) plans for a 2012 report "State of Europe's water". The report will be based on information reported in 2010 via RBMPs and supplemented with assessments of information from other sources.

Ultimo August 2011 data from 141 river basin districts (RBDS) have been uploaded to the EEA Country Data Repository (CDR) and incorporated into the WFD-WISE database. There are still missing reporting from some countries and RBDs. The European Environment Agency (EEA) and its Topic Centre on Water are currently analyzing the detailed information and data reported in the RBMPs. The analysis focuses on analyzing data and information on status, pressures and impacts of European surface waters. The current presentation will provide overviews of preliminary results on the ecological status or potential of Europe's waters and the pressures affecting them.

Results on status of European surface waters

In the following preliminary results from the analysis of river basin management plants are presented. The results are based on more than 140 RBMPs reported from 23 EU Member States.

Category	*Member States*	*RBDs*	*Number of water bodies*	*Length or area*
Rivers	*22*	*141*	*82811*	*912 000 km*
Lakes	*20*	*126*	*17477*	*80 200 km^2*
Transitional	*15*	*77*	*952*	*13 200 km^2*
Coastal waters	*18*	*97*	*2774*	*267 600 km^2*
Total	*23*	*141*	*104014*	

Table 1: Number of countries, RBDs, water bodies, and length or area, per water category.
Note: Prelimilary results from analysis of 141 RBD reported by 23 EU Member States to the WISE-WFD database

Throughout the EU, more than 100 000 surface water bodies (WBs) have been reported (Table 1); approximately 80 % are river water bodies; 17 % lake water bodies and the remaining 3 % coastal and transitional water bodies, respectively. All Member States (MS) except Malta have reported river WBs, 20 MS have reported lake WBs, and 15 and 18 MS have reported transitional and coastal WBs, respectively. In total, more than 900 000 km of river length, 80 000 km^2 lakes and 280 000 km^2 transitional and coastal waters have been reported.

Overall, more than half (55 %) of the total number of reported and classified water bodies in Europe are in less than good ecological status/potential (figure 1). For rivers, there are 43 000 water bodies (57% of the total number of WBs), in less than good ecological status or potential. For lakes, the overall status is somewhat better than in rivers, but there are still almost 6000 lake water bodies (43% of total number) in less than good ecological status or potential. The worst water category is transitional waters, where 68% of the water bodies are in less than good ecological status or potential. In coastal waters, the situation is somewhat better with 49% of water bodies in less than good ecological status or potential.

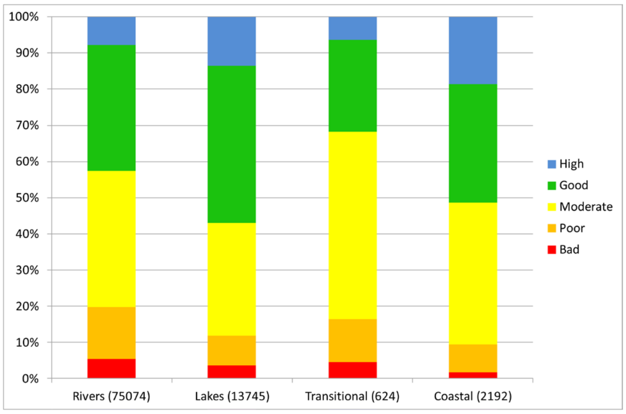

Figure 1: Distribution of ecological status/potential of classified EU rivers, lakes, transitional and coastal waters.
Note: Preliminary results from analysis of 141 RBD reported by 23 EU Member States to the WISE-WFD database. Number of water bodies is given in parenthesis.

The reason why lakes are better than rivers are due to two thirds of the reported lake water bodies being from Sweden and Finland where the population density and agricultural pressures are relatively low, while the rivers are more evenly distributed throughout Europe with a larger proportion of rivers in densely populated and cultivated areas in Central Europe.

The reason why transitional waters are so much worse than coastal waters are due to their proximity to pollution sources from land based sources including loading from the upstream river basins. Moreover, transitional waters are exposed to extensive hydromorphological pressures caused by land reclamation, flood protection, as well as large harbours causing altered habitats in these water bodies.

Pressures affecting Europe's surface waters

The proportion of water bodies without pressures and the main significant pressures affecting the different water categories are illustrated on figure 2. For 57 % of the lake water bodies no significant pressures were identified, corresponding to 57 % of lake water bodies classified as having good or high ecological status or potential. Only one third of the river water bodies, as well as the transitional water bodies have no pressures and around 40 % of the coastal water bodies have been reported to have no pressures.

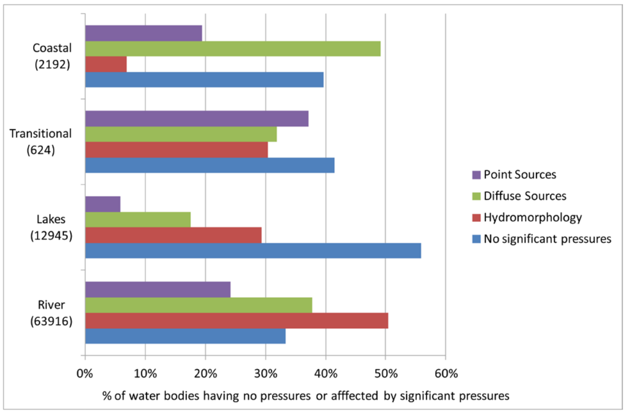

Figure 2: Proportion of classified EU rivers, lakes, transitional and coastal waters without no significant pressures and affected by main pressures.
Note: Preliminary results from analysis of 141 RBD reported by 23 EU Member States to the WISE-WFD database. Number of water bodies is given in parenthesis.

Both for rivers and lakes hydromorphological pressures have been identified as affecting the highest proportion of water bodies. For rivers more than half of the water bodies are affected by hydromorphological pressures, while for lakes and transitional water bodies around 30 % of the water bodies are affected by hydromorphological pressures. Only 7 % of the coastal water bodies are affected by hydromorphological pressures.

Generally 30 to 50 % of the water bodies are affected by pollution pressures with emissions from diffuse sources being the most important pollutant pressure.

Nearly 40 % of the river water bodies are affected by diffuse sources and a quarter of the river water bodies have point sources as a significant pressure. The proportion of lake water bodies being affected by pollution sources are lower than the other water categories reflecting that the majority of lake water bodies have been reported from Sweden and Finland.

For coastal waters almost half of the water bodies are affected by pressures from diffuse sources and 19% of the water bodies are affected by point sources. For transitional waters point sources and diffuse sources are affecting more than a third of the water bodies.

Ecological status and pressures of rivers

The ecological status or potential of rivers varied from some Member States having more than half of the river water bodies in at least good ecological status/potential (Estonia to Ireland in Figure 3a), while other Member States (Czech Republic to Belgium (Flanders)) have less than 20 % in good ecological status.

– The Central-European Member States, with high population density and intensive agriculture, generally have a high proportion of river water bodies in less than good ecological status, while

– The highest proportion of river water bodies with good ecological status or potential is mainly found in more sparsely populated Member States with less arable land, e.g. Northern Europe, and other parts of Europe (Estonia, Latvia, Spain, Romania, Slovakia, Ireland and Italy).

– Although the general picture is valid, the detailed results for each country are however uncertain due to weaknesses in the monitoring and assessment systems applied for this first cycle of RBMPs.

The proportion of river water bodies with no significant pressure generally followed the ranking of Member States based on at least good ecological status (Figure 3b), i.e. Member States having a more than half of the river water bodies in good ecological status generally also had the a high proportion of river water bodies without significant pressures. Opposite the Member States with a large proportion of water bodies in less than good ecological status generally had the majority of river water bodies with significant pressures.

There is good agreement with the ranking of MS by at least good ecological status and the proportion of river water bodies per Member State being affected by diffuse pollution and hydromorphology pressures (Figure 3c).

Future work

The above presented results illustrate the value of the information reported in the river basin management plans concerning the status and pressures affecting Europe's waters; more results will be presented at the WISER end -user conference. EEA and its Topic Centre are for the moment finalizing reports illustrating the status and pressures affecting Europe's waters. The draft reports will be put out for country and stakeholder consultation during February and March and published later in 2012. It is EEAs hope that the draft reports will be read and commented by the WISER community.

Figure 3 a-c): Ecological status or potential of classified river water bodies in different Member States sorted by proportion of good or better ecological status/potential (lower panel), river water bodies with and without pressures reported (left panel on next page), and proportion of river water bodies affected by diffuse pollution and hydromorphology pressures (right panel on next page).

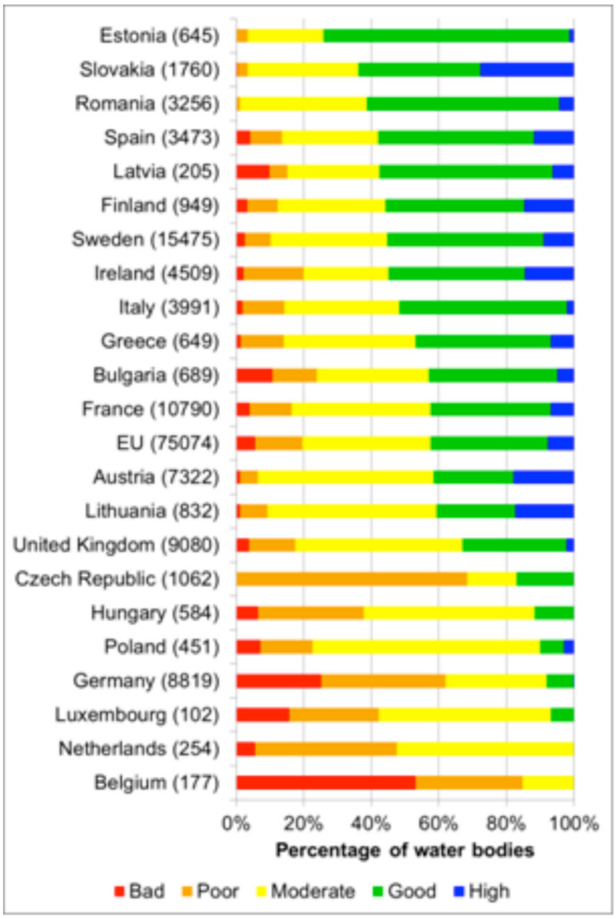

Figure 3 a): Rivers Ecological status or potential

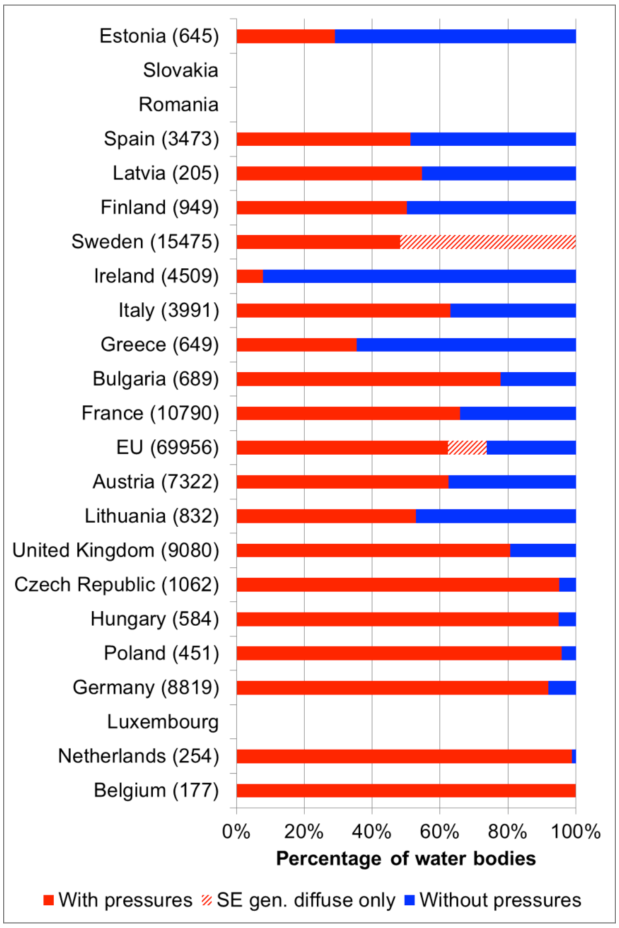

Figure 3 b): River WBS with pressures/no pressures (details see previous page)

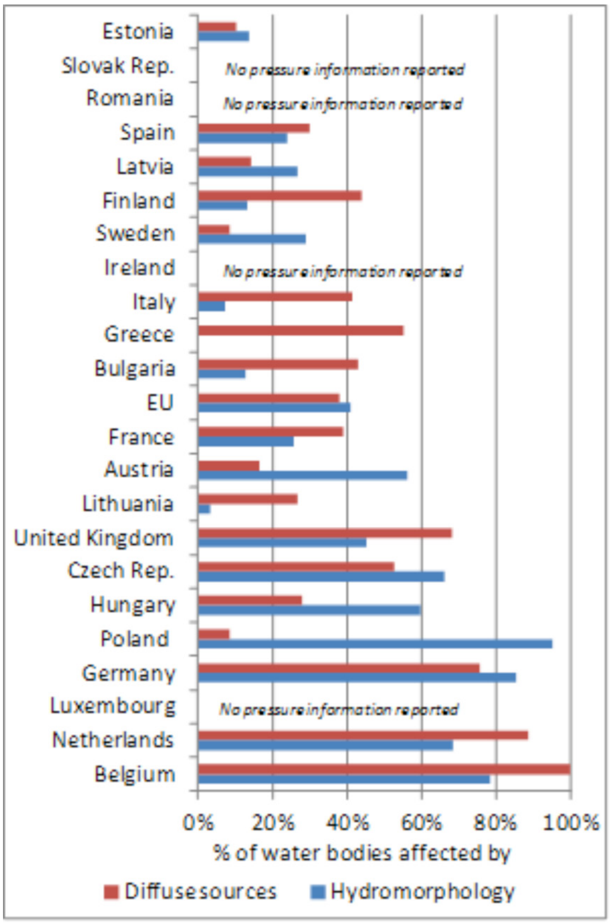

Figure 3 c): Rivers affected by hydromorphology and diffuse pollution pressures (details see previous page)

Ecological potential and fish communities of Czech artificial lakes

J. Kubečka[1, 2], D. Boukal[3], J. Duras[4], M. Kratochvíl[2], J. Peterka[2] and M. Prchalová[2]

[1] Biology Centre of the AS CR, v.v.i.
[2] Institute of Hydrobiology and
[3] Entomology, Na Sádkách 7, 370 05 České Budějovice, Czech Republic
[4] Vltava River Authority, Water Quality Laboratories, Denisovo nábř. 14, Plzen 30420, Czech Republic

Email address of corresponding author: kubecka@hbu.cas.cz

Key words: fish, lake, reservoir, succession, biotic integrity, nutrient loading, management, conservation

Abstract

Czech Republic lacks natural lakes larger than 20 hectares. On the other hand, hundreds of artificial water bodies were built during the past six centuries. These are:

1. Drainable fish ponds of mostly medieval origin, used primarily for fish (mainly carp) production and usually drained every second or third year. Although they are frequently highly eutrophic and hyper-productive, simple management options are available to improve their water quality because they typically have small catchments and their management involves at most few stakeholders.
2. Reservoirs built for retention of water (drinking water, flood protection, irrigation), power generation, leisure activities (swimming, recreational fishing, boating and other water sports) and navigation, built mostly during the 20th century. They are drained only in exceptional cases for emergency repairs or removal of bottom sediments. The state of these reservoirs varies between oligotrophic to eutrophic; any management aimed at improving water quality in these reservoirs is typically complicated by large catchments and multitudes of different stakeholders.
3. Lakes created after strip mining of sand, gravel or brown coal, built mainly in the past 10 years. They are permanent and will be used mainly as recreational sites. Target water quality and management will be subject to discussions and decisions in the future.

Reference states and desired ecological states of these diverse artificial systems are difficult to define due to their different usage and sometimes conflicting or undefined stakeholder interests. However, it is clear that the present ecological state of many artificial water bodies in the Czech Republic is far from satisfactory. The main ecological pressures involve eutrophication, acidification, excessive angling pressure on predatory fish, water level fluctuation driven by flood protection and hydropower generation, and spread of invasive fish species. Therefore, efforts to improve the ecological status involve extensive work with multiple stakeholders groups.

In the absence of natural lakes as local analogues, management aimed at improving the conditions of the various lakes, reservoirs and ponds can draw from expert judgment based on long-term studies of fish communities, limnological parameters and water quality in selected reservoirs (Římov, Nýrsko, Klíčava, Lipno and Slapy Reservoirs and Chabařovice Open Cast Lake) and from reference examples from natural lakes in neighboring countries (Austria, Germany, Poland, Slovakia).

Fish are a key component of animal communities in these systems. We present a summary of long-term case studies of fish stock monitoring and management in Czech lakes and reservoirs as a background for future effort of classification of ecological potential and its enhancement. We use state-of-the-art sampling methodology (Kubečka et al. 2010) that can match the most advanced approaches in EU. In 2008, pilot monitoring program was executed and regular monitoring is expected to start in the coming years. At the moment, sound monitoring information is available of some 15 reservoirs of the above category 2 and 3 "coal lakes". The process of defining of lake typology, ecological potential and its classification officially has not started, but the philosophy of it is being elaborated as more information is collected about Czech and neighboring systems. Czech Republic at present does not directly develop the intercalibration of ecological status of lakes but functions as an observer in the initiative to define fish ecological classification for the Water Framework Directive.

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Enhancing and Improving Monitoring and Assessment in Light of Climate Change Impacts in the United States

Sarah Lehmann, Ellen Tarquinio

U.S. EPA, 1301 Constitution Ave, NW, 4503T, Washington, D.C. 20460 USA

Email address of corresponding author: Lehmann.sarah@epa.gov

Key words: water quality, monitoring, indicators, reference condition, probability, climate change

Introduction

Scientific research and monitoring are critical aspects of understanding water quality condition, changes and the causes of those changes. This applies to traditional water quality problems as well as to climate change related impacts. The United States Environmental Protection Agency (EPA) and its partners are researching and testing a variety of indicators and monitoring designs to enhance our ability to detect climate related changes as well as uncover potential vulnerabilities that climate change may pose to our traditional monitoring programs. In the United States (U.S), a variety of monitoring programs collect data, however they were not designed specifically to address climate change issues. This presentation will discuss some of the on-going work in the United States and how one of these monitoring programs, the National Aquatic Resource Surveys, can be leveraged to support such needs.

Water Quality Monitoring Background

In the U.S., local, regional, state and federal agencies conduct a wide variety of water quality monitoring. For example, state/tribal water monitoring programs have the primary authority under the Clean Water Act to conduct monitoring in support of water programs. Since 1991, the U.S. Geological Survey has operated the National Water-Quality Assessment program to develop long-term information on targeted streams, rivers, and ground water sites. The National Estuary Programs, designed to protect and restore the water quality and ecological integrity of estuaries of national significance, conduct monitoring to assess conditions and track progress implementing their management plans.

While these efforts provide valuable information and help address site-specific issues, differing monitoring objectives, designs, and methods for collecting and assessing data mean that it is not possible to integrate information to make national and regional assessments of the condition and changes over time of all U.S. waters. To address these issues, EPA initiated the National Aquatic Resource Surveys (NARS).

National Aquatic Resource Surveys

NARS is a partnership between EPA and States and Tribes. The surveys are designed to provide the public and resource managers with scientifically valid reports on the quality of the nation's waters. On a five year rotating basis, individual surveys are implemented for rivers/streams, lakes/reservoirs, coastal waters (including the marine and Great Lake coastal waters), and wetlands (Tab. 1). Specific objectives of each survey include assessing the biological and recreational (e.g., public health) condition and key stressors of all waters across the U.S., and ranking stressors based on the relative associations between indicators of condition and indicators of stress. Additionally, the national surveys are helping build stronger monitoring programs across the country by fostering collaboration on new methods, indicators and water quality research.

For each survey, more than 1000 sites are sampled during the field year. The national surveys use randomized sampling designs, core indicators, and consistent monitoring methods and lab protocols. The specific indicators differ by waterbody type, but always includes chemical, physical and biological data. Additional targeted

Table 1: National Aquatic Resource Surveys Schedule.

	2010	2011	2012	2013	2014	2015
Rivers/ Streams	Lab/data analysis	Lab/Data Analysis; Research	Report; Design	Field	Field	Lab/Data Analysis
Coastal	Field	Lab/Data Analysis	Report	Research	Design	Field
Wetlands	Design	Field	Lab/Data Analysis	Report	Research	Design
Lakes	Research	Design	Field	Lab/Data Analysis	Report	Research

reference (least disturbed) sites are also sampled. More than 700 lake and stream reference sites have already been identified through NARS with more coming from the rivers and wetlands surveys.

Climate Change Monitoring Research and Leveraging the NARS

The same inconsistencies in monitoring programs that led to the development of NARS can limit the utility of data for evaluating the impacts of climate change on our water resources. To more effectively understand and track the impacts of climate change, water resource scientists and managers need comparable long-term data from waters (streams, rivers, lakes, wetlands and coastal waters) that are minimally disturbed by human activities as well as across the range of human disturbance. Recognizing this issue, the EPA and others have been researching and evaluating improved indicators and network designs to address climate issues.

Indicators

Work is ongoing in the U.S. to identify indicators and metrics that are sensitivity to climate change in terms of condition and vulnerability. EPA, states and tribes are exploring how biological assessments can be used in concert with physical, chemical, and land use data to help identify baseline biological conditions against which the effects of global climate change on aquatic life can be studied and compared. Such information could enable a water quality management program to calibrate biological assessment endpoints and criteria to adjust for long-term climate change conditions.

For example, EPA and Utah are partnering to evaluate the potential impact of global climatic trends on the aquatic biota. As part of this project, EPA analyzed biological data from four reference streams, in two ecoregions with long term stream invertebrate data to determine whether past climate trends could be detected and to characterize vulnerabilities of the biological assessment program to future climate conditions. Results indicated that long-term declines in richness or abundance of cold-preference taxa were detecTab. at the two longest-term sites. From those results, EPA and Utah estimated that a 25 - 40 percent loss of EPT (Ephemeroptera, Plecoptera, and Trichopera) taxa could occur with current scenarios of temperature increases by 2050. Should such losses occur due to climate change, it would confound measures of ecological condition and decisions regarding attainment of aquatic life uses.

Additionally, work is going on to apply the Biological Condition Gradient (BCG), a model describing biological response to increasing levels of stressors, in support of climate change-related efforts. The model describes how ten attributes of aquatic ecosystems change in response to increasing levels of stressors. Some initial work by EPA and others has suggested that multi-metric indices may be more vulnerable to climate change than predictive models. Related research suggests that a temperature-modified EPT richness metric shows promise as a way to track climate-change related impacts. EPA and others are also examining whether improved assessments can be realized by focusing more specifically on species traits and functional roles which could be guided by the BCG model. The BCG framework can also be used to demonstrate how climate change impacts can alter biological communities in such a manner as to be confused with traditional stressors.

Although NARS was not established with climate change issues in mind, the core elements allow the surveys to be adapted for this purpose. For example, an important component of NARS is that the surveys include biotic and abiotic data collection at all sites. This provides a valuable dataset for examining the relationships between biological condition and stressors, including land use changes. Climate researchers are using this unique, nationally consistent data set to support their efforts. Additionally, working with climate change researchers and our state/tribal partners, EPA has the opportunity to consider and add indicators – such as improved biological metrics -- that will support climate change assessments not in one area of the country or for one waterbody type but across the U.S. and for all of our rivers, streams, lakes, wetlands and coastal waters. The continuing nature of the surveys will also allow scientists to reanalyze data from previous time periods as new insights are gained.

Building Monitoring Networks and Reference Condition

There have been many calls in the U.S. for greater coordination of existing monitoring programs. One recent example comes from the U.S. National Oceans Policy (NOP). The NOP provides a comprehensive national policy for the stewardship of the ocean, our coasts, and the Great Lakes. Among other recommendations, it calls for strengthening monitoring networks to provide information about how coastal conditions and resources are changing over time, including impacts of climate change. Agencies working on action plans for the NOP have called for leveraging NARS to serve as a critical component of the monitoring network.

Additionally, EPA recognizes the need for a robust reference condition network not only for our conventional water programs but to support climate-change information needs. Several efforts are ongoing in the United States to design and begin implementation of such a network although questions remain to be answered pertaining to the design requirements. The EPA Global Change Research Program is designing a pilot climate change network in the New England region. Initial steps for this pilot include determining the most appropriate target population, determining what design (targeted, probabilistic, or a mix) will be used, how to identify 'vulnerable' sites, what indicators will be tracked, and how frequently. The National Water Quality Monitoring Council is also proposing development of a long-term collaborative reference network for streams.

Because NARS uses least disturbed sites across the U.S. and across waterbody types, the surveys can serve as critical component of establishing a reference network to support climate change. In the examples identified above, NARS can and is playing a critical role in framing and implementing the networks. The NARS design could be modified to collect climate-change related parameters at all sites, at reference quality-sites or at sites deemed especially vulnerable. Given additional resources, partners could increase the frequency of sampling to more effectively meet climate change related needs.

Impact of climate change on freshwater fish species at the European scale and associated uncertainty

Maxime Logez[1], Andreas Melcher[2] and Didier Pont[1]

1IRSTEA (former Cemagref), UR HBAN, 1 rue Pierre-Gilles de Gennes – CS 10030, Antony, France; phone: +33 (0)4 42 66 69 86
[2] University of Natural Resources and Life Sciences (BOKU), Institute of Hydrobiology and Aquatic Ecosystem Management, Max-Emanuel-Straße 17, 1180 Wien, Austria

Email address of corresponding author: maxime.logez@irstea.fr

Key words: global change, species distribution models, logisitic regression, confidence interval, case study, temperature, precipitation, Traun River

Background

Environmental conditions are one of the main drivers of fish species distributions along river networks. In order to anticipate and assess the effect of climate change on the occurrences of fish species, and especially of temperature rising, numerous Species Distribution Models (SDMs) were developed to project the species distribution depending of the gas emission scenario. The idea underlying these models is to relate the occurrence of a given species in stream sections (sites) to the environmental conditions observed in these sites. This assumption is valuable for all statistical methods that were used to define SDMs. Occurrence-environment relationships are then used to predict species presence-absence under different climatic scenarios.

Even if numerous SDMs were used to predict the shift in species distributions and assemblage composition, very often these models were computed over a spatial extent that would reflect only a small fraction of the realized niche of these species. These models should thus over- or underestimate the effect of climate changes because they would extrapolate species occurrences for environmental conditions not taking into account in the calibration data sets. Moreover, very often these models only integrate temperatures and do not consider precipitation. Logez et al. (accepted), developed SDMs for 23 widespread riverine fish species in Europe with the data base of the EFI+ project (14 European countries). These models: consider a great diversity of environmental conditions, were computed on sites not or slightly impacted and integrate both temperature and precipitations through the stream power. These models would be useful to assess the climate change effect on the future distribution of these 23 species.

In parallel to the SDMs, some long term studies provide real observations of the climate change effect on river functioning and their consequences on fish populations. The case study of the Traun River conducted over more than 30 years highlights the response of a coldwater species to water warming.

Objectives and approach

The objectives of these studies were to assess the climate change, both in term of temperature and precipitations, on the future distribution of fish species, but also the uncertainty on the predicted patterns. This is a major concern for water managers to have an idea of the vulnerability of populations to climate change to program restoration measures or to assess the reliability of taken these measures.

Four emission gas scenarios, averaged from three global circulation models, were used to predict species distributions at the European scale by 2020–2030 and 2050–2060. For each scenario, the probabilities of presence of each species, in absence of pressure, were computed from the logistic regressions developed by Logez et al. (accepted). These probabilities were compared to a threshold probability (different for each species) and derived into absence-presence. The confidence intervals associated with each expected probability were computed using the Wald's approach. These intervals estimate the uncertainty around each prediction. Larger this confidence interval is, more uncertain is the species response to global change.

In parallel, the case study of the Traun River, central Austria, was used to assess the potential effect of temperature rising on historical grayling populations. This study focus on the grayling populations of section Traun lake outflow downstream.

Results

Species present various patterns of response to climate change

On average, the mean air temperature in July is expected to increase in our sampling sites by 1.7 °C in 2020–2030 and by 3 °C in 2050–2060, whereas the warming between these two periods is less pronounced for mean air temperature in January (Table 1). Precipitations are expected to decrease by 16 mm in 2020-2030 and by 8 mm in 2050-2060.

Table 1: Monthly mean air temperatures for current and projected climatic conditions.

	2020-2030		2050-2060		Current	
Scenario	Jan.	July	Jan.	July	Jan.	July
A1F1	*2.946*	*19.850*	*3.586*	*21.783*		
A2	*2.924*	*19.787*	*3.155*	*21.151*		
B1	*3.081*	*19.825*	*2.993*	*20.674*		
B2	*3.100*	*19.842*	*3.056*	*20.733*		
Average	3.013	19.826	3.197	21.085	1.054	18.113

The patterns of responses to climatic shifts are highly variable between species, but the greatest changes will occur by 2050–2060. For cold- and coolwater species such as brown trout and its associated species, grayling and Atlantic salmon, the number of locations will suitable habitat would be greatly reduced. Other species such as nase, dace and soufie will face both local extinctions and new colonisations suggesting a shift of their distribution by 2050–2060, while climatic conditions will become highly suitable for bleak (Table 2, Figure 1).

Table 2: Number of locations where habitat will remain unsuitable/suitable by 2050–2050 and the number of locations that will present future unsuitable/suitable habitat conditions (averaged from the four scenarios).

Species	Stable unsuitable habitat	Stable suitable habitat	Lost habitat	New habitat
Alburnus alburnus	*1806*	*1532*	*4*	*774*
Barbus barbus	*1454*	*847*	*242*	*420*
Chondrostoma nasus	*715*	*431*	*317*	*285*
Cottus gobio	*1510*	*535*	*1514*	*143*
Gobio gobio	*1989*	*1220*	*416*	*426*
Leuciscus cephalus	*1097*	*1342*	*443*	*827*
Leuciscus leuciscus	*1780*	*773*	*344*	*317*
Rhodeus amarus	*504*	*1139*	*103*	*800*
Salmo salar	*1735*	*388*	*917*	*0*
Salmo trutta	*2069*	*1161*	*1274*	*4*
Telestes souffia	*672*	*104*	*189*	*435*
Thymallus thymallus	*2264*	*299*	*687*	*36*

Uncertainty could blur the pattern of responses

For some species the uncertainty around the probabilities of presence was so high that depending of the confidence interval limit opposite patterns of response to climate change could be predicted (Table 3). This is the case for instance of nase, for which local extinctions are mainly predicted when using the lower limits of the confidence intervals or an important expansion of its distribution area when considering the upper limits.

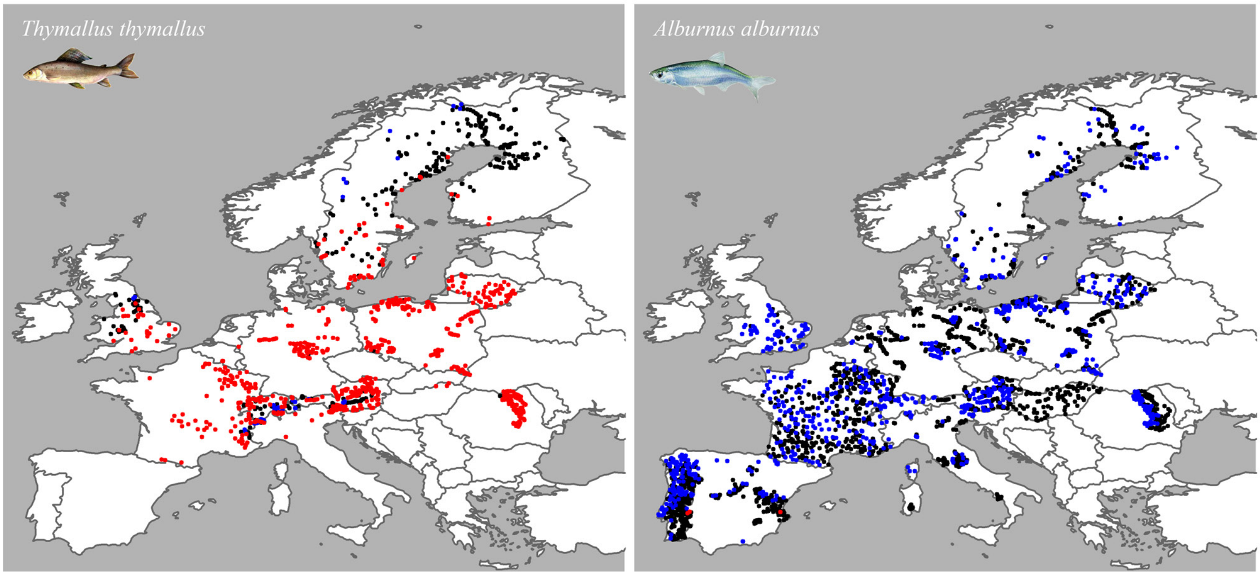

Figure 1: Projected distributions of grayling (left plot) and bleak (right plot) in the period 2050–2060. Black dots represent unchanged suitable conditions (compared to current climatic conditions), blue dots represent location with climatic conditions becoming suitable, and red dots location with climatic conditions becoming unsuitable.

Table 3: Number of location with habitat becoming unsuitable/suitable by 2050–2060, estimated with the lower limit of the confidence intervals or with the upper limit of the confidence intervals associated with predicted probabilities.

	Lost habitat		New habitat	
Species	**Lower limit**	**Upper limit**	**Lower limit**	**Upper limit**
Alburnus alburnus	*27*	*0*	*364*	*1237*
Barbus barbus	*326*	*187*	*268*	*599*
Chondrostoma nasus	*528*	*208*	*84*	*487*
Cottus gobio	*1695*	*1337*	*78*	*216*
Gobio gobio	*485*	*355*	*299*	*570*
Leuciscus cephalus	*502*	*393*	*660*	*985*
Leuciscus leuciscus	*432*	*258*	*150*	*507*
Rhodeus amarus	*217*	*37*	*478*	*1006*
Salmo salar	*988*	*833*	*0*	*0*
Salmo trutta	*1511*	*1025*	*0*	*11*
Telestes souffia	*232*	*147*	*176*	*627*
Thymallus thymallus	*768*	*574*	*18*	*78*

Low uncertainties are associated with species with a marked response to climate change such as bleak and Atlantic salmon.

Grayling populations will suffer from climate change

Over the period 1976-2008, the water temperature in August of the section Traun lake outflow downstream, increase by 2.2 °C on average. In parallel with this warming the abundance of the grayling sharply decline, replaced by species such as barbel (Figure 2). For this river, the global change was followed by a shift of assemblage composition toward species with warmer thermal ranges. At a measuring point temperatures of 24 °C and more are reached constituting a substantial source of stress for the grayling.

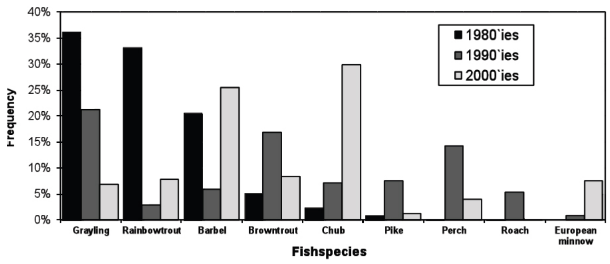

Figure 2: Abundances of fish species 1980-2000

Implications

Although restoring rivers from a societal and ecological point of view a good thing, water managers must take into account the climate change in their decision process to restore rivers. The restoration programs could be inefficient in regards of the prior objectives, not because of inappropriate measures but because climatic conditions will overwhelm the limiting effect of human pressures and limit the occurrence of the target species. The case study of the Traun river clearly show that a return to the historical fish assemblages is impossible and can´t be reached for instance by fish stocking at the lake outflow section

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A detailed study of climate change effect on fish distribution, assemblage composition and functional structure, illustrated by case studies will be available through WISER deliverable 5.1-3.

Changes of phytoplankton diversity in geographical, nutrient and lake morphometry gradients

Kairi Maileht[1], Tiina Nõges[1], Peeter Nõges[1], Ute Mischke[2], Laurence Carvalho[3], Bernard Dudley[3] and Ingmar Ott[1]

[1] Estonian University of Life Sciences, Institute of Agricultural and Envriornmental Sciences, Centre for Limnology, 61117 Rannu Parish, Tartu County, Estonia.
[2] Leibniz-Institute of Freshwater Ecology and Inland Fisheries Department of Shallow Lakes and Lowland Rivers. Justus-von-Liebig-Str. 7. D-12489 Berlin, Germany.
[3] Centre for Ecology & Hydrology (CEH), Bush Estate, Penicuik, Midlothian EH26 0QB, UK

Email address of corresponding author: kairi.maileht@emu.ee

Key words: phytoplankton, lake, diversity, species richness, lake morphometry, nutrient, geographical distribution

Abstract

The biogeographical distribution of freshwater phytoplankton diversity and its driving factors are still largely unknown. Large-scale gradients are driven by local environmental factors that vary along latitudinal, longitudinal and altitudinal gradients and differ in lakes of different morphometry. Our aim was to reveal large-scale distribution patterns of environmental factors, such as temperature, nutrients, and lake morphometric characteristics, in the latitudinal, longitudinal and altitudinal gradients; to assess the impact of these factors on phytoplankton amount, composition and diversity, and to reveal the potential implications for the assessment of the ecological status of lakes. The analysis tries to account for differences in taxonomic resolution of phytoplankton analyses and applies besides taxa richness also the numbers of higher taxonomic groups (genera and classes).

Twenty countries have provided data on 1683 lakes (BE-11, CY-7, DE-223, DK-108, EE-66, ES-147, FI-162, FR-9, GR-1, HU-28, IE-54, IT-18, LT-41, LV-65, NL-50, NO-516, PL-50, RO-10, SE-96, UK-21) in frames of the WISER project. Summer months July and August were selected for the analysis. Data has been gathered during a long period (1972-2009), but the bulk (56%) originated from the last ten years. The database included environmental parameters (latitude, longitude, altitude, alkalinity, catchment area, humic type, maximum depth, mean depth, surface area, colour, P-PO_4, Secchi disc visibility, total N, total P, temperature) and phytoplankton data (species richness, chlorophyll a, total phytoplankton biovolume, separate biovolumes of ten dominating species, biovolumes of taxonomical classes) from surface samples. Altogether the database consisted of 6700 rows.

To reveal the general patterns in data, we used Factor Analysis (FA) and Principal Component Analysis (PCA) offered by Statistica 8.0 (StatSoft, Inc., 2007). For more detailed analysis, the most dominating species from each sample with a biomass > 5 g/m^3 (altogether one hundred) were selected for the Canonical Correspondence Analyses (CCA) using the multivariate statistical package (MVSP; KCS, 2007).

The FA and PCA analyses revealed lake morphometry, location and nutrient availability as the three major factors affecting taxa richness and the amount of phytoplankton (Fig. 1).

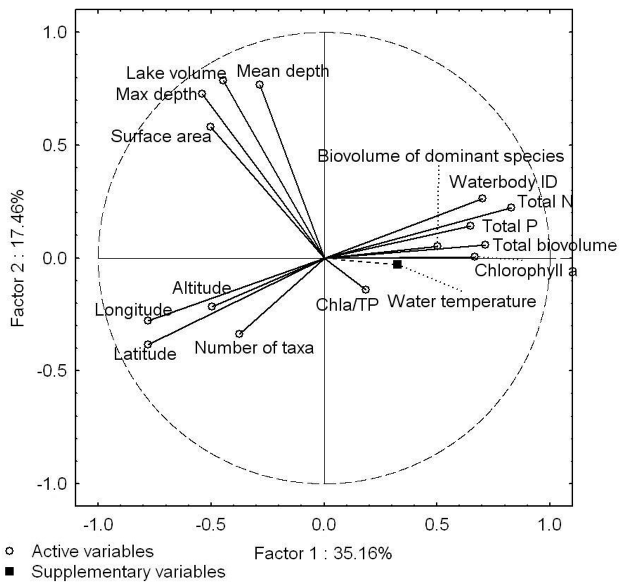

Figure 1: Multivariate statistical factor analyses of environmental and phytoplankton parameters in Europe.

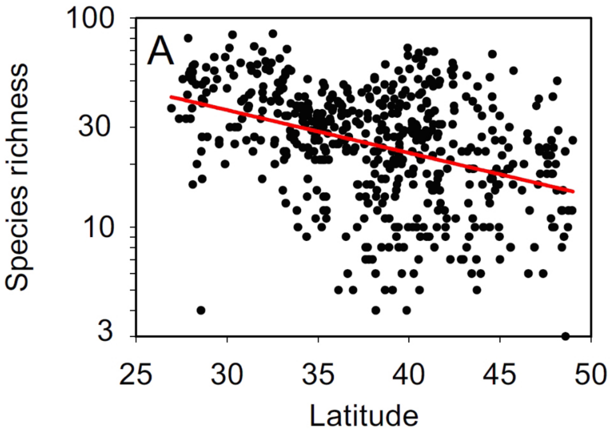

Figure 2: Phytoplankton taxa richness in the gradient of latitude in USA (Stomp et al., 2011).

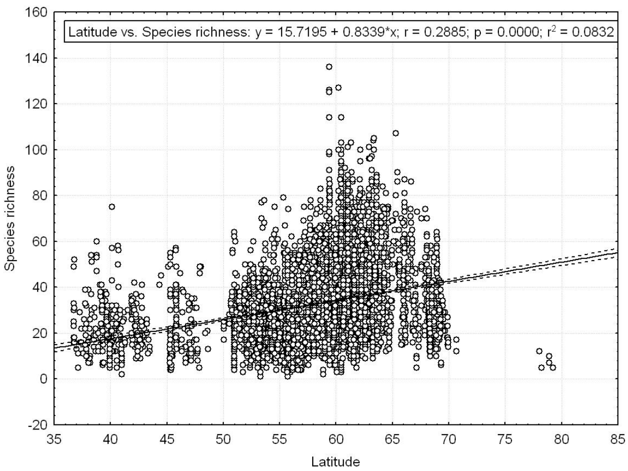

Figure 3: Phytoplankton taxa richness in the gradient of latitude in Europe.

The correlation between total phosphorus (TP) and the amount of phytoplankton, probably one of the best-known connections in lakes, was obvious also in our data. Besides that, the amount and taxa richness of phytoplankton might be related with the location. Nevertheless these connections are often hard to reveal because of regionally different taxonomic resolution and traditions in taxonomic work. As one successful attempt, Stomp et al. (2011) found strong latitudinal, longitudinal, and altitudinal gradients in phytoplankton biodiversity in 540 lakes and reservoirs distributed across the continental United States (Fig. 2).

Differently from his findings, the phytoplankton taxa richness seems to increase with latitude in Europe (Fig. 3). This seeming anomaly can probably be attributed to the northward increasing lake richness in Europe as four countries with largest percentage of lakes in the total land cover - Sweden, Finland, Norway, and Estonia - are located in North-Europe. For comparison, northern lakes in the USA are located at higher altitudes and they are smaller (Stomp et al., 2011; Nõges, 2009), whereas northern lakes in Europe are larger and shallower with smaller catchment areas, lower alkalinity, pH and conductivity. Within Europe, northern lakes have less nutrients and more dissolved organic compounds than southern lakes. Probably the variety of habitats in northern lakes is bigger giving growing opportunities to a larger number of species.

The CCA showed, similarly to FA and PCA, the importance of morfometry, location and nutrients on phytoplankton variables, but the groups were not so well distinguished (Fig. 4).

The factors encompassing the effects of morphometry and altitude, included in countdown order the species *Anabaena planctonica*, *Staurastrum sp.*, *Planktothrix rubescens*, *Planctonema lauterbornii*, *Mougeotia sp.*, *Synura sp.*, *Fragilaria crotonensis*, and *Peridinium bipes*. In accordance with our expert knowledge, most of these species are characteristic of lakes with larger volume and surface area. This opinion is supported as well by the list of phytoplankton indicator genera created by G. Phillips with co-authors (2010) within the WISER project .

Among nutrients, TP was the strongest factor showing a positive effect on *Cyclotella meneghiniana*, *Microcystis sp.*, *Aphanizomenon sp.*, *Pediastrum boryanum*, *Woronichinia naegeliana* and a negative effect on *Ceratium furcoides*. The list of taxa positively affected by TP included some taxa, such as *Mallomonas sp.*, which occurrence, by expert knowledge, has been commonly associated with moderate content of nutrients.

The third factor consisted of location parameters, but included also water colour. Well understandable is the positive relationship of the raphidophyte *Gonyostomum sp.* with this factor, since this is a well-known nuisance alga in Scandinavian and Baltic soft water lakes. Other taxa associated with this factor were in countdown order *Uroglena sp.*, *Peridiniopsis cunningtonii*, *Aulacoseira islandica*, *Ulnaria acus* and *Staurastrum crenulatum*.

Our analyses showed a detectable effect of geographical parameters on the distribution of phytoplankton species in European lakes that needs to be taken into account especially if applying taxonomy based phytoplankton indices for status assessment in areas outside the one for which it has been elaborated.

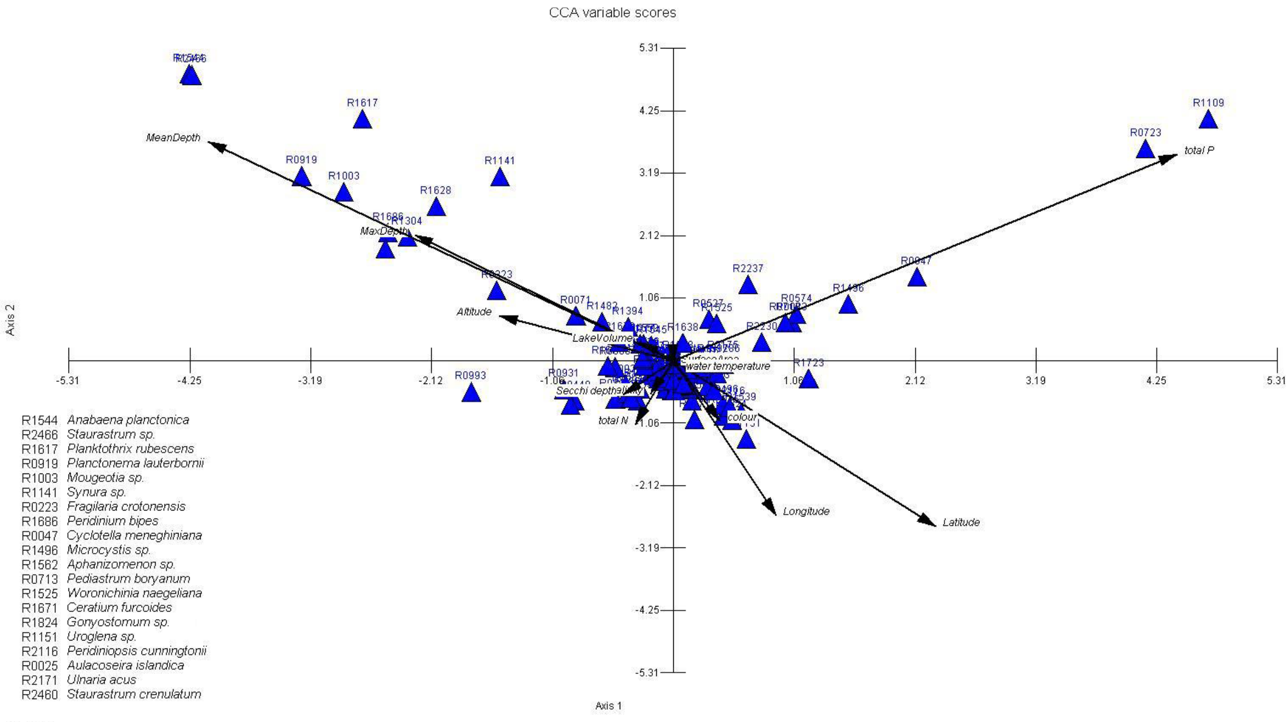

Figure 4: Factors affecting distribution of dominant phytoplankton taxa by canonical correspondence analyses.

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Diversity of European seagrass indicators. Patterns within and across regions

Núria Marbà[1], Dorte Krause-Jensen[2], Teresa Alcoverro[3], Sebastian Birk[4], Are Pedersen[5], Joao M Neto[6], Sotiris Orfanidis[7], Joxe M Garmendia[8], Iñigo Muxika[8], Angel Borja[8], Kristina Dencheva[9]

[1] IMEDEA (UIB-CSIC), Spain
[2] NERI (AU), Denmark
[3] CEAB (CSIC), Spain
[4] University of Duisburg-Essen, Germany
[5] NIVA, Norway
[6] IMAR, Portugal
[7] National Agricultural Research Fundation, Greece
[8] AZTI, Spain
[9] IO-BAS, Bulgaria

Email address of corresponding author: nmarba@imedea.uib-csic.es

Abstract

Seagrasses are key components of coastal marine ecosystems and many monitoring programs worldwide assess seagrass health and apply seagrasses as indicators of environmental status. This study aims at identifying the diversity and characteristics of seagrass indicators in use within and across European ecoregions in order to provide an overview of seagrass monitoring effort in Europe. Through a compilation we identified 49 seagrass indicators representing 42 monitoring programs and including a total of 51 seagrass metrics used either alone or in various combinations of up to 14 metrics per indicator. The seagrass metrics represented 6 broad categories covering different seagrass organizational levels and spatial scales. The large diversity is particularly striking considering that the pan-European Water Framework Directive sets common demands for the presence and abundance of seagrasses and related disturbance-sensitive species across Europe, and the diversity of indicators reduces the possibility to provide pan-European overviews of the status of seagrass ecosystems. The diversity can be partially justified by differences in species and associated time scales of responses as well as by differences in habitat conditions and associated community types but also seems to be determined by tradition. We encourage an evaluation of seagrass indicators on the basis of their responses to pressures in space and time and their associated uncertainty in order to identify the most suiTab. indicators for specific European regions.

The relative influence of watershed, riparian zone and local anthropogenic pressures on fish and macroinvertebrate communities in French rivers

Anahita Marzin[1], Piet Verdonschot[2] and Didier Pont[1]

[1] Cemagref, UR HBAN, 1 rue Pierre-Gilles de Gennes, CS 10030, 92761 ANTONY cedex, France
[2] Alterra, Wageningen UR, Postbus 47 / Droevendaalsesteeg 3 (gebouw Lumen) 6700 AA Wageningen, Netherlands

Email address of corresponding author: anahita.marzin@cemagref.fr

Key words: redundancy analysis, land uses, local anthropogenic stressors, river, fish, macroinvertebrate

Background

The idea that rivers should be managed at the catchment scale has become widespread. Managers are more and more prone to use money saving and easy to acquire proxies of river ecological status such as land uses/covers instead of fastidious direct measures of the local pressures and samplings of the biological communities. Indeed, it is commonly accepted that river reach scale communities are structured by local abiotic factors (e.g. water physical and chemical parameters) that are in turn constrained at larger scales such as buffer or catchment factors (land uses and covers). Impacts of anthropogenic factors on river communities have been largely documented at the local scale and are now well documented at large scales too (segment and catchment). Nevertheless, to our knowledge, only few studies have attended to compare the ability to explain the variability in biological assemblages at these different spatial scales and results are not always consistent among studies. In addition, most of the studies did not distinguished environmental factors defining the system conditions and that are quasi-independent of human activity (here named "natural" environment factors) from those directly influenced by human activity (commonly named "human pressure factors"). As other authors, we advocate that for river management purposes, the latter are of prime interest as they represent meaningful triggers for stakeholders to restore or maintain ecological quality of water bodies.

Objectives and approach

The objective of this study was to compare the relative influence of anthropogenic pressures on river biological assemblages at different scales (watershed, riparian zone, site) while differentiating influence of "natural" environmental factors and anthropogenic stressors. Three questions were addressed: (i) What are the links among watershed and riparian zone land uses and reach scale pressures? (ii) What are the links among anthropogenic pressures variables and river biological community composition in French rivers? (iii) What is the part of the variation in French freshwater communities (fish and macroinvertebrates) explained by system condition variability, human-induced pressures at the reach scale, riparian land uses and catchment land uses? Based on the results of previous studies, we expected to observe strong links between land uses and local pressure variables, links between pressure variables and biological community compositions and that biological composition variability would be more affected by natural environmental factors and reach scale pressures than larger scale stressors. Finally, we suspected that complex interaction effects exist among these spatially different pressures.

Our predictions were examined through French national data on 301 river sites. First, in order to describe land uses relationship with local habitat modifications, correlations were calculated among land cover types at the two scales (buffer, catchment) and local stressors. Second, partial redundancy analyses were conducted at three spatial scales (local, buffer, catchment) for each biological group to define the relationship among anthropogenic pressures and river communities removing beforehand the effect of the "natural" environment. Finally, partition of the variation of the biological communities were analysed in order to compare unique and shared influences of natural environment and of the 3-scales anthropogenic variables.

Results

Land uses as proxies of local anthropogenic pressure variables

Water quality parameters were generally better correlated to land covers than hydro-morphological parameters implying that when considering land uses as proxies for river local degradations, water quality problems will be better represented than local habitat and hydro-morphological problems. Upstream catchment land covers were better correlated to water quality reach scale parameters and buffer land covers to hydro-morphological degradations. These results are in accordance to those of previous studies (e.g. Moerke and Lamberti, 2006) suggesting that catchment land covers are possible proxies of local water quality parameters and buffer land covers predictors of local habitat and hydro-morphological parameters.

Linked among anthropogenic pressures and biological community composition

In this part, we have focused on the influence of human-induced pressure variables at different scales after having removed the variability related to “natural” environment factors. The part of the total inertia of communities’ compositions explained by the analyses was lower for macroinvertebrate than for fish. Although, abiotic factors explained a significant part of the variability in biological communities, the chosen abiotic factors might be more relevant to explain fish community variation than macroinvertebrate community.

However, common patterns were observed for the response of fish and macroinvertebrate communities to pressures at the different scales and biological community distributions along the pressure gradients were coherent with bio-ecological knowledge on fish and macroinvertebrate taxa. The presence of an impoundment emerged as the main human pressure factor shaping the communities at the local scale, followed by water quality and morphological pressure gradients. At broader scales (buffer and catchment), fish and macroinvertebrate communities appears to be greatly influenced by a common gradient from forested covers to agricultural land uses. Increase in buffer artificial and wetland covers appears to be another important gradients influencing macroinvertebrate assemblage composition. These findings are consistent with numerous previous studies demonstrating the important role played by human-induced pressures on the species composition of riverine assemblages.

Ability of catchment, riparian zone or reach scale variables to explain biological assemblage variation

As expected, variables not directly influenced by human activities, as geology or altitude, account for a large part of the among-sites explained differences in community composition (about 30%). These findings strengthen the idea that “natural” variability in environment is a key parameter explaining river community composition diversity and should be always considered and taken into account beforehand when looking at the effect of human-induced pressures on river ecological quality in order to attempt to distinguish the two effects.

A large part of the explained variability in community composition was related to factor shared effects (around 40% of the explained variability). Such complex effects illustrate why it is so delicate to establish simple pressure-impact relationship for fish and macroinvertebrates in river as pressure effects are generally difficult to separate. Consequently, in the common case of multi-impacted sites, it will very hard to answer the water managers about the main pressure disturbing the river ecological status.

In addition, results concerning relative influences of anthropogenic pressures were different for macroinvertebrate and fish communities. Land use variables seem more important for macroinvertebrate community composition while fish community composition appears to be more sensitive to local anthropogenic pressures. These results are not surprising given previous finding supporting that land use variables mainly reflect water quality degradations of reach and upstream area. Indeed, previous works have already shown that macroinvertebrate communities are generally more sensitive to water quality degradation than fish communities.

Conclusions

Given these results, it appears likely that land uses and local pressures both significantly explain river fish and macroinvertebrate community compositions. Although land uses appear to be useful approximations of the global water quality degradation of the upstream river, they should be combined with information on local scale pressures and the “natural” environmental factors be considered beforehand to describe effect of human activities. Finally, this study supports the idea that pressure effects on river communities are usually complex and that it is often hard to determine the main pressure affecting a river.

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Exploring the robustness and reliability of several macrophyte-based classification methods to assess the ecological status of coastal and transitional ecosystems

O. Mascaró[1], T. Alcoverro[1], K. Dencheva[2], D. Krause-Jensen[3], N. Marbà[4], J.M. Neto[5], V. Nikolić[6], S. Orfanidis[7], A. Pedersen[8]

[1] Centre d'Estudis Avançats de Blanes (CSIC). C/ d'Accés a la Cala St. Francesc 14, 17300 – Blanes (Girona). Spain
[2] Institute of Oceanology - BAS, "Parvi may str. N 40", PoBox 152, 9000 Varna, Bulgaria
[3] Department of Bioscience, Aarhus University. Vejlsøvej 25 DK-8600, Silkeborg. Denmark
[4] Department of Global Change Research. IMEDEA (CSIC-UIB), Institut Mediterrani d'Estudis Avançats. Miquel Marquès 21, 07190 Esporles (Illes Balears). Spain
[5] Institute of Marine Research (IMAR-CMA). Dep. of Life Sciences, University of Coimbra. Largo Marques Pombal, 3004-517 Coimbra, Portugal
[6] Institute of Oceanography and Fisheries. I. Meštrovića 63, HR-21000 Split. Croatia
[7] National Agricultural Research Foundation. Fisheries Research Institute, 640 07 Nea Peramos, Kavala. Greece
[8] Center for Coastal and Marine Research, Norwegian Institute for Water Research - NIVA, Gaustadalléen 21, NO-0349 Oslo. Norway

Email address of corresponding author: omascaro@ceab.csic.es

Key words: uncertainty analysis; Water Framework Directive; biological quality elements; risk of misclassification

Abstract

One of the main aims of the WISER Project (Water bodies in Europe: Integrative Systems to assess Ecological status and Recovery) is to evaluate the robustness and reliability of the different indices developed by the EU members, addressing all water categories, organism groups and environmental stressor types. This is to be done mainly through the use of uncertainty analysis, one of the most powerful tools to assess the main weaknesses of biotic indices that allow the identification of the factors contributing to the potential misclassification of the ecological status class of water bodies (Clarke and Hering 2006). The estimation of uncertainty is a central element in WFD-compliant assessment methods, since they are based on biological communities that show both spatial and temporal heterogeneity, and because errors will be introduced during sampling and analytical stages (Kelly et al. 2009). If the major sources of variability are known, they can potentially be minimised through the re-design of sampling schemes (additional sampling sites or frequency), through improved training by operating procedures, CEN (European Committee for Standardization) guidance, taxonomic training or through the use of model-based assessment methods. For this reason, ecological status classification results should always be given in terms of probabilities depending upon the variability associated with these communities over time and space (Hering et al. 2010). However, only a small proportion of classification methods have put this into practice and the uncertainty analyses available in the literature are scarce at the moment (Staniszewski et al. 2006, Kelly et al. 2009, Bennett et al. 2011).

The objective of this contribution is to analyse the uncertainty associated to several WFD-compliant classification methods based on macrophytes (both macroalgae and seagrasses) that have been developed by different EU Member States (Table 1). Specifically, we attempt to determine which sources of variability (factors) associated with the sampling design of the different indices most greatly influence the ecological status classification of water bodies. In addition, we also observed how the boundary values between status classes can affect the general pattern of uncertainty displayed by the different factors in each index.

The analyses are based on both official and non-official EQR datasets from the different indices that include some of the key sources of variability associated with the design and implementation of a regional scale biomonitoring program (e.g. spatial scales of sampling, the temporal scale of sampling, the human-associated source of error). However, the number and nature of factors examined that potentially contribute to the uncertainty of the EQR estimations of coastal water bodies differ among the indices, especially due to differences in

Table 1. Main characteristics of some of the indices included in this study.

Index	Country of application	Target species	Metric/s used	References
MSMDI *Multi Species Maximum Depth Index*	*Norway*	*Saccharina latissima* *Chondrus crispus* *Rhodomela confervoides* *Coccotylus truncata* *Phyllophora pseudoceranoides* *Halidrys siliquosa* *Delesseria sanguinea* *Phycodrys rubens* *Furcellaria lumbricalis*	*Lower depth limit*	–
EDL *Eelgrass Depth Limit*	*Denmark*	*Zostera marina*	*Lower depth limit*	*Krause-Jensen et al., 2005*
POMI *Posidonia oceanica Multivariate Index*	*Spain, Croatia*	*Posidonia oceanica*	*Physiological, morphological, population (density) and community, integrated onto a single scale using Principal Component Analysis*	*Romero et al., 2007*
EEI-c *Ecological Evaluation Index*	*Italia*	*Cymodocea nodosa-ESG IA* *Ruppia cirrhosa-ESG IA* *Cystoseira barbata-ESG IB* *Gracilaria bursa-pastoris-ESG IIA* *Cladophora spp.-ESG IIB* *Ulva spp.-ESG IIB*	*Coverage (%) of 5 different Ecological Status Groups clustered hierarchically into two ESG's*	*Orfanidis et al., 2011*
EI *Ecological Index*	*Bulgaria*	*Cystoseira barbata-ESGI* *Cystoseira crinite- ESGI* *Corallina spp.- ESGI* *Gelidium latifolium- ESGI* *Zostera noltii- ESGI* *Zostera marina- ESGI* *Potamogeton pectinatus- ESGII* *Ulva spp.- ESGII* *Cladophora spp.- ESGII* *Ceramium spp.- ESGII* *Chaetomorpha spp.- ESGII* *Polysiphonia spp.- ESGII*	*Biomass proportion (%) of different macrophyte species classified in 2 different Ecological Status Groups: sensitive (ESGI) and tolerant (ESGII)*	*Dencheva in press*
SQI *Seagrass Quality Index*	*Portugal*	*Zostera noltii*	*- Taxonomic Composition (TC)* *- Bed Extent (BE)* *- Shoot Density (SD)*	–

both the metrics used and their sampling designs. First of all, the total variance and variance components associated to each factor were estimated for all indices using a linear mixed effects model in the lme4 package of R (Version 2.10.1, R_Development_Core_Team 2009). It is important to note that variability among water bodies, whilst important in the analysis of variance components, is not discussed in this study because by definition they should differ in their ecological status. Posteriorly, the uncertainty in ecological status classification was estimated using WISERBUGS (WISER Bioassessment Uncertainty Guidance Software®, Clarke 2010). WISERBUGS helps determine whether an observed ecological status classification is indeed the most probable classification for a particular site, given the inherent sources of variability. Because the current study was interested in the uncertainty in classification generated by a particular factor (rather than the probability of misclassifying individual sites), the probability of misclassification for each factor was determined along the full range of possible observed EQR values (0 - 1).

Generally for all factors, the probability of misclassification peaks when a site's observed EQR score is very close to the boundary between two status classes, usually around 50%. In contrast, when the observed EQR falls in the middle of a status class the probability of misclassification declines to the minimum. Probabilities of misclassification >50% may indicate that the associated variability is actually higher than the EQR range of the status class. The magnitude of these maximum and minimum uncertainty levels differ greatly among factors and indices as a result of the differences in the variance extracted.

Our results show that spatial scales of variability (above and below the water body scale) have different influence in the ecological classification status of water bodies depending on the index. For example, the uncertainty associated to the factor region was high in EDL and POMI indices (Figs. 1a and 1b), which may indicate that it was separating groups of water bodies of similar quality status. Below the water body spatial scale, variability among sites showed also a high uncertainty associated in EDL, MSMDI and EI indices (Figs. 1a, 2a and 3b) compared to POMI, SQI and EEI-c (Figs. 1b, 2b and 3a), indicating that the spatial heterogeneity displayed by these biological communities was not properly captured in their corresponding sampling designs. In order to absorb part of this spatial variability and minimize the risk of misclassification, the sampling effort must be increased to include a greater number of sites within water bodies and, in each one, to collect several sub-samples and average metric values. In contrast, the temporal scale of sampling did not promote important levels of uncertainty in the ecological status classification of water bodies in any of the indices that included this factor (EDL, POMI, MSMDI, EI and SQI; Figs. 1a, 1b, 2a and 3b respectively). This indicates that the EQR scores of water bodies are fairly consistent throughout the years, for which the frequency of sampling could be decreased without greatly reducing the precision of ecological status estimates. Surprisingly, low levels of uncertainty were also attributed to differences among surveyors (Figs. 1b and 2a). This may be attributed to the fact that these macrophyte-based indices do not require complicated taxonomic identifications, which can greatly affect the precision of the EQR estimations in the case of other classification methods based on diatoms (Kelly et al. 2009) or freshwater macrophyte communities (Staniszewski et al. 2006). Finally, we observed that the risk of misclassifying the quality status of water bodies is also affected by the width of the status class in which the EQR score falls, as reported in Kelly et al. (2009), with narrower classes leading to greater probabilities of misclassification. Thus, indices in which the EQR range is not equally split into the 5 official classes (EDL, POMI and EEI-c) present, for a certain variance associated to a factor, changing uncertainty levels depending on the status class (Figs. 1a, 1b and 3a). This fact may have drastic implications for bio-monitoring programs, because a greater sampling effort may need to be assigned 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.

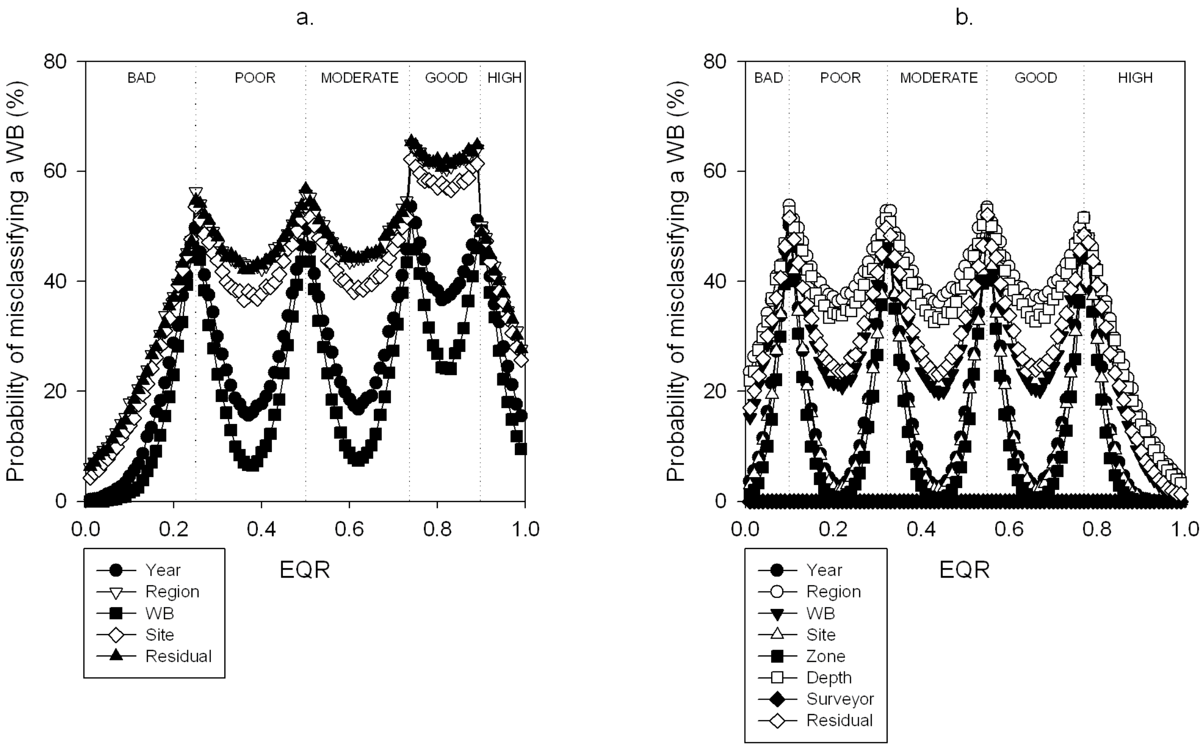

Figure 1. Probability of misclassifying the ecological status associated to the different factors analysed for the EDL (a) and POMI (b) indices. Vertical dashed lines represent the boundaries of each status class. For EDL: Bad = 0 – 0.249; Poor = 0.25 – 0.499; Moderate = 0.5 – 0.739; Good = 0.74 – 0.899; High = 0.9 – 1. For POMI: Bad = 0 – 0.099; Poor = 0.1 – 0.324; Moderate = 0.325 – 0.54; Good = 0.55-0.774 and High = 0.775 – 1.

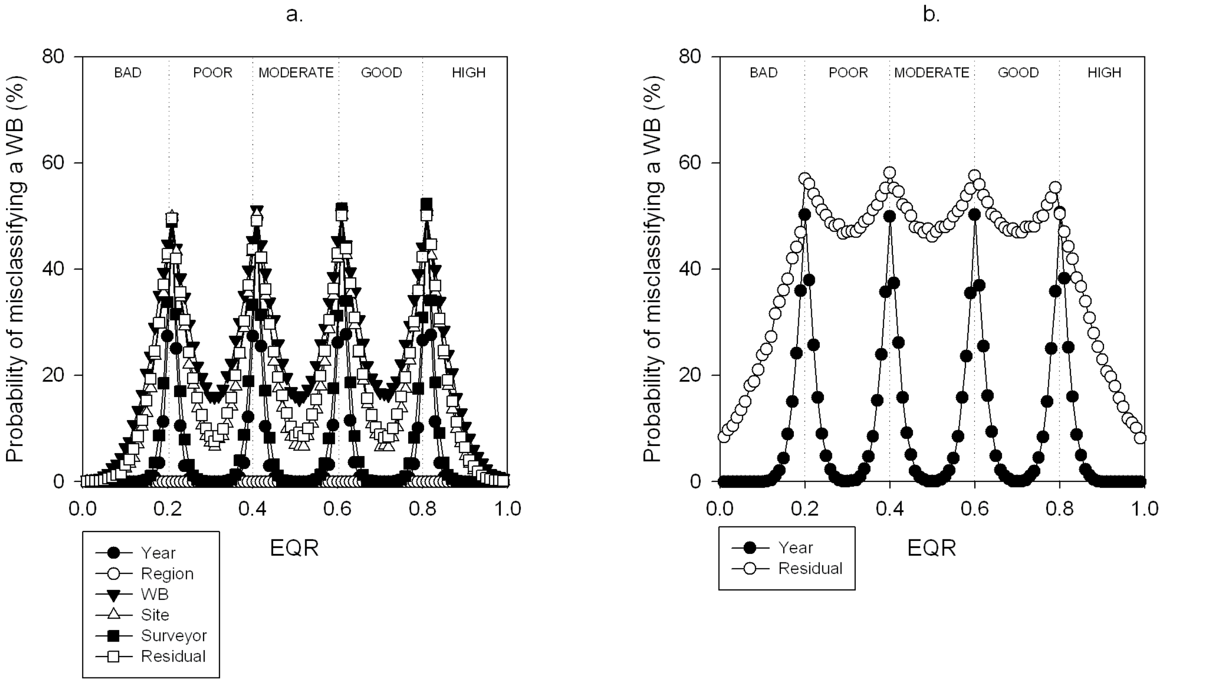

Figure 2. Probability of misclassifying the ecological status associated to the different factors analysed for the MSMDI (a) and SQI (b) indices. Vertical dashed lines represent the boundaries of each status class; for both indices: Bad = 0 – 0.2; Poor = 0.21 – 0.4; Moderate = 0.41 – 0.6; Good = 0.61-0.8 and High = 0.81 – 1.

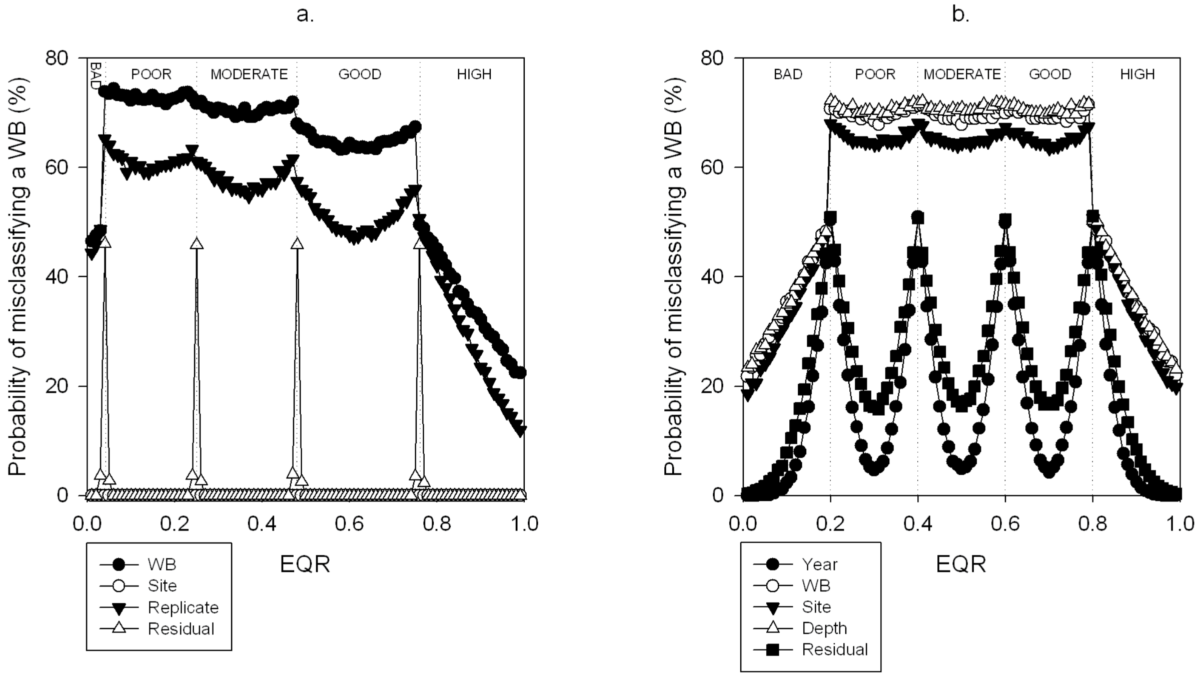

Figure 3. Probability of misclassifying the ecological status associated to the different factors analysed for the EEI-c (a) and EI (b) indices. Vertical dashed lines represent the boundaries of each status class. For EEI-c: Bad = 0 – 0.04; Poor = 0.041 – 0.25; Moderate = 0.26 – 0.48; Good = 0.49-0.76 and High = 0.77 – 1. For EI: Bad = 0 – 0.2; Poor = 0.21 – 0.4; Moderate = 0.41 – 0.6; Good = 0.61-0.8 and High = 0.81 – 1.

The current study is in line with one of the main objectives of the WISER Project, helping to gain insight into the robustness and reliability of some of the ecological status classification methods proposed for European waters under the WFD. Applying uncertainty analysis to extensive bio-monitoring datasets, we have been able to detect the main weaknesses of these indices and provide robust foundation for improving their monitoring programmes, as well as guide decisions in future management plans. Besides, this study highlights the importance of extensive data series, essential to improve the methodologies proposed to assess the ecological status of coastal and transitional ecosystems under the WFD.

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Effects of climate change on fish assemblages in terms of lakes and their outlets in Alpine areas – explained by the case study Traunsee

A.H. Melcher, H. Kremser, F. Pletterbauer, S. Schmutz

University of Natural Resources and Life Sciences (BOKU), Institute of Hydrobiology and Aquatic Ecosystem Management, Max-Emanuel-Straße 17, 1180 Wien, Austria

Email address of corresponding author: andreas.melcher@boku.ac.at

Key words: Climate change, Water temperature, Fish fauna, Lake, River, European Alps

Introduction

Riverine ecosystems are affected by different anthropogenic pressures. Besides water pollution, hydro-morphological alterations, connectivity disruptions and direct interferences with the fish community (e.g. fishing, stocking), climate change jeopardizes the integrity of river ecosystems (Schmutz & Mielach 2011). The rising of air-temperature is the best known phenomenon for global climate change (Matulla at al 2007; Kromp-Kolb 2003). Since water temperature is mainly determined by heat exchange with the atmosphere, higher air temperatures lead to higher water temperatures. For rivers, there are strong correlations between water and air temperature (Solheim et al. 2010; Hari et al. 2005). In water bodies water-temperature is a determining factor and plays a major role in the distribution of fish species.

The global climate change and local anthropogenic impacts, like factory heating emissions, can result in warming water bodies and consequently permanently modify their fish biocenoses. Most aquatic organisms (e.g. salmonids) have a specific range of temperatures they can tolerate, which determines their spatial distribution along a river or on a regional scale. Climate change could lead to the extinction of some aquatic species or at least modify their distribution in a river system or move their distribution northwards. Several indications of climate impact on the functioning and biodiversity of freshwater ecosystems have already been observed, such as northward movement, phenology changes and invasive alien species (Solheim et al. 2010). Temperature has to be considered as an environmental resource and should have increased importance as a structuring factor of river fish assemblages, especially in regulated and degraded river systems, directly related to the loss of fluvial habitats therein (Wolter 2007).

Relevance and effects of temperature changes on fish fauna

Water-temperature is one of the most significant factors for the survival of aquatic biota (flora and fauna) in freshwater ecosystems (Armour 1991; Hutchinson 1976; Fry 1971). The temperature regime influences all life stages in the fish population, including their migratory behaviour, egg evolution, spawning process, fertilization and growth rate as well as their metabolism, respiration and tolerance towards parasites. Minor modifications often restrict species in their occurrence and respective distribution (Schmutz et al. 2000; Jungwirth & Winkler 1984). For example a small increase in water-temperature can alter the entire fish species community, Low temperatures cause lethargy to species (reduced digestion, low reaction time), elevated temperatures increase metabolism (e.g. digestion) to the extent where fish cannot find enough food to compensate and their fat reserves are exhausted. Eurythermic species prefer significantly higher temperature during summer, while temperature conditions in rivers are very similar during winter for steno- and mesothermic species (Jungwirth et al. 2003, Schmutz & Mielach 2011).

Interventions in the temperature regime of a water body can lead to advantages for one species and disadvantages for others. Fish species prefer different temperature regimes along a river continuum that correspond with typical distributions of fish species assemblages. Traditionally they are assigned into different fish zones and their indices (Huet 1949, Matulla et al. 2007, Schmutz & Mielach 2011). The Austrian fish-zonation-index (FiZi) generally ranges from 3.8 (trout zone) to 5 (grayling zone) to 6 (barbel zone) and 6.8 (bream zone).

Temperature and climate change

Selected Austrian rivers and lakes (Fig. 1) were analysed in terms of changing water-temperature from 1976

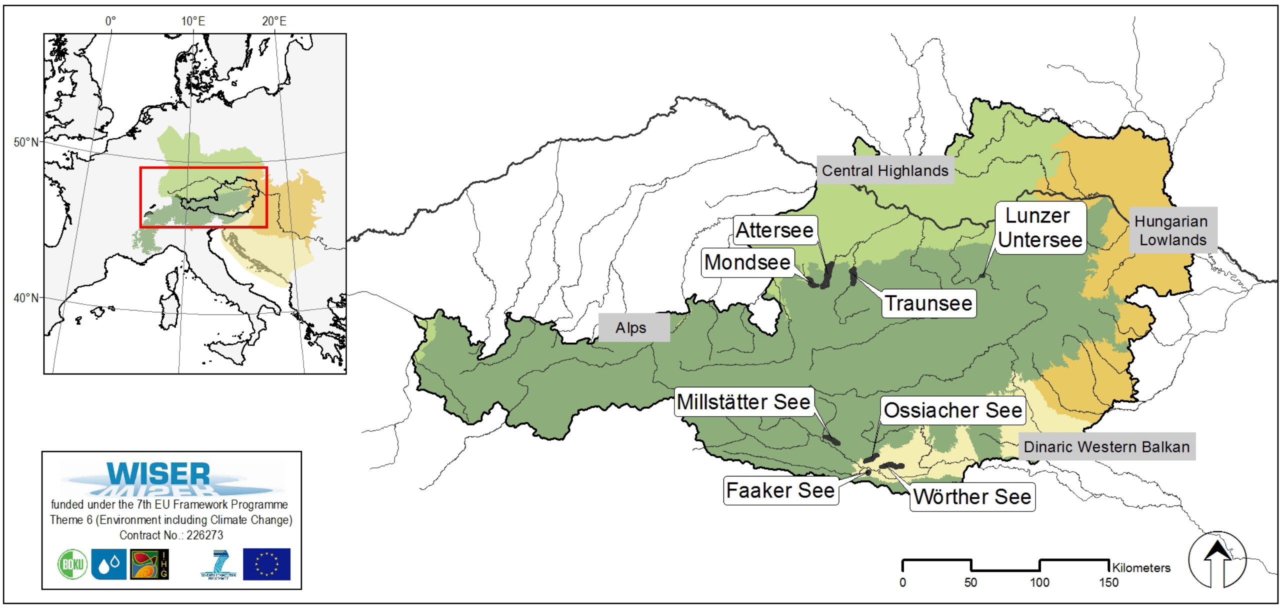

Figure 1: Study area – Case study Lake Traun (Traunsee) and seven other Alpine lakes and their rivers (grey lines).

until 2006 and represented for increasing temperatures trends.

Analyses of approximately 90 Austrian Water Gauging Stations (HZB stations; not impacted in terms of water-temperature) revealed an increase of temperature of approximately 2.5°C in and 3°C in lakes in the last 30 years. The results match the study of Webb and Nobilis (2007) that showed a temperature increase of nearly 2°C in Austrian streams during the 20th century. The rising of water-temperature is explicit for lakes. It is evident that summer 2001 (June, July, and August) could be selected as an average one. The time series analyses revealed for August 2003 especially warm, and August 2005 rather cold.

Lake outflows (Tab. 1) are influenced by a lake situated upstream. It is obvious that these lakes are more strongly affected by climate change than water bodies devoid of influence from a lake. if climate warming trends continue the lake outflows could be a surrogate prognosis for the for all rivers in the future.

Table 1: Alpine lake's outlets (river sections directly below the lakes) and their specific mean water-temperature in August (2001), altitude and their fish-zonation-index (FiZi, Schmutz et al. 2000).

Lake names	Water Temperature [°C]	Altitude [m]	FiZi
Lunzer Untersee	*19.4*	*607*	*4.8*
Traunsee (Lake Traun)	*20.4*	*421*	*5.3*
Attersee	*21.1*	*467*	*5.6*
Mondsee	*22.1*	*479*	*6.1*
Millstätter See	*23.2*	*587*	*6*
Faaker See	*23.9*	*553*	*5.6*
Ossiacher See	*24.2*	*500*	*6.2*
Wörthersee	*24.3*	*439*	*6*

Case study Traun river and Traun lake

The case study area is located in the central part of Upper Austria (Fig. 1). The whole catchment area of the Traun River ranges from almost 3000m (Dachstein) to 250m (mouth into Danube River) above sea level (Fig. 2).

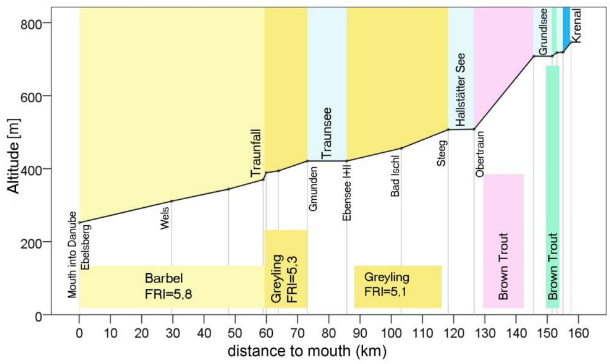

Figure 2: Schematic course of Traun River from the source down to the mouth into the Danube including its three lakes, the historic fish-zones and their FiZi (after Haunschmid et. al 2006).

We focus on the section downstream from Lake Traun, a typical and historically proven grayling zone (Fig. 2) The European grayling (*Thymallus thymallus*) almost disappeared from this section when their biomass drastically decreased over the last 30 years from 70 to 4 kg/ha (Melcher et al. 2009).

Water temperature analyses of Lake Traun followed a general trend towards a mean increase of 2.2°C within

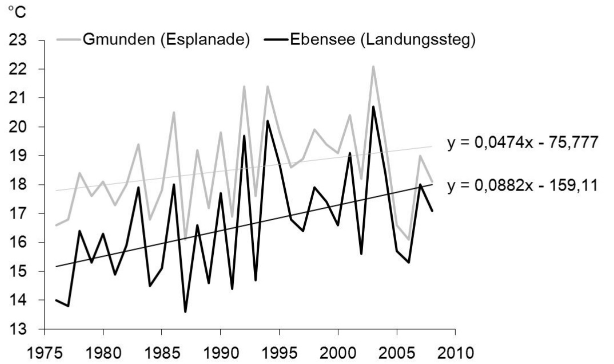

Figure 3: Mean water-temperature trend (August, 1976 - 2008) of Lake Traun at site Gmunden and Ebensee (lower and upper part of the lake).

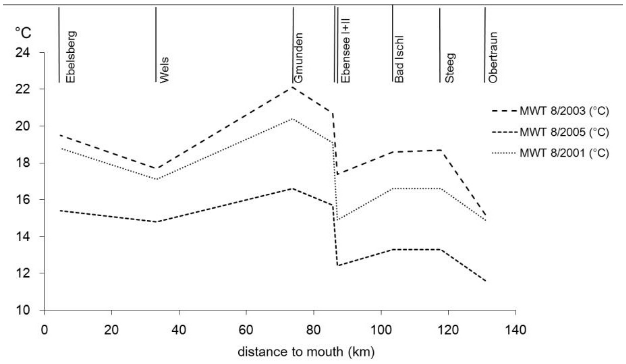

Figure 4: Schematic Traun River course from upstream down to the mouth into the Danube (Ebelsberg) and the Lake Halstatt (Steeg) and Lake Traun (Ebensee and Gmunden) in between. Above Wels the tributaries Ager and Alm cool down the main river. Three years (2001, 2003, and 2005) were selected to demonstrate a typical cold (2005), warm (2003) and an average year (2001).

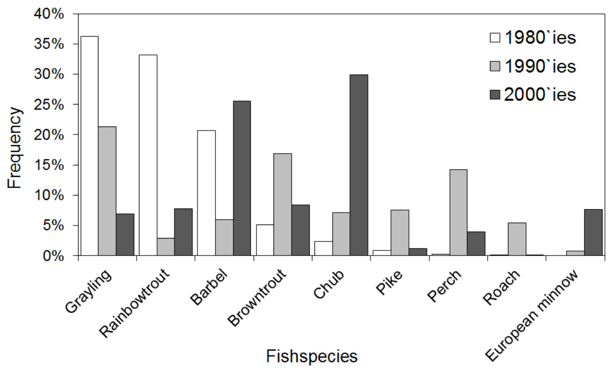

Figure 5: Shift of fish species composition from the 1980s until today.

the last 33 years from 1976 to 2008 (Fig. 3). Furthermore there is an average temperature difference of 2°C between the upper (station Ebensee) and lower part (station Gmunden).

Fig. 4 is demonstrating the increase of River Traun´s temperature because of the influence of the lakes Lake Halstatt (Halstätter See) (km 120) and Lake Traun (km 80) within three selected years (average 2001, warm 2003 and cold 2005). Even in the cold years the water temperature in the river section below the lake (outflow) is too warm for grayling with it is at the upper optimum of 18°C for adults. As stated above, in this outflow section the biomass and abundance of grayling decreased dramatically.

In addition the fish species composition completely changed in the River Traun below Lake Traun over the last 30 years (Fig. 5). In the 1980's grayling was the main fish species, today barbel and chub dominate, which are more tolerant of higher water-temperatures. Another biological indicator for temperature change is biodiversity, where the total number of fish species in this section has nearly doubled from 14 to 27 species (Melcher et al. 2009).

Conclusion

For adult graylings a temperature of 18°C is the upper optimum. At the former grayling zone at Traun River rising temperatures could now reach 24°C due to climate change trends and corresponding warming of the Traun Lake (Fig. 6). For the grayling, these higher temperatures constitute a substantial source of stress (Küttel et al. 2002). Under these circumstances there is no possibility of a return to the historic fish assemblages and excludes the option of mitigation with fish stocking.

The River Traun will fail to achieve good ecological status expected by the WFD. River management plans should take into account rising water temperature trends when assessing cumulative effects and additional pressure on river systems. Furthermore the impact from increased temperature scenarios are compounded by barriers to fish migration (longitudinal and lateral; e.g.

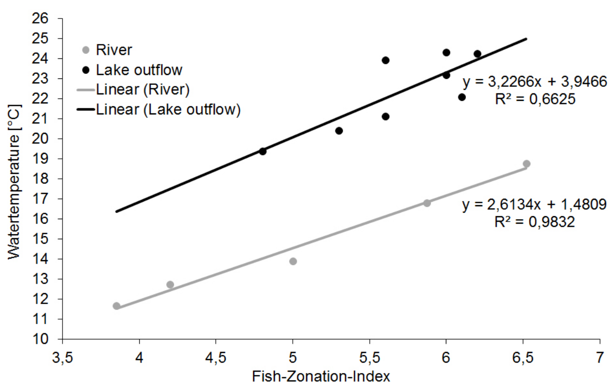

Figure 6: Linkage of fish species composition (FiZi) and water-temperature (August) for rivers and river sections below a lake (lake outflow).

hydro facilities and channelization) restrict the movement of fish into colder regions to avoid hot spots in river sections or other water bodies. For future research and management it is expected that anthropogenic-induced temperature impacts will become increasingly significant in regulated river systems with reduced hydrodynamics (Wolter 2007). This will have repercussions on the survival of endangered coldwater species such as grayling (*Thymallus thymallus*) or Danube salmon (*Hucho hucho*) (Matulla et al. 2007).

Acknowledgments

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A multimetric tool for the ecological assessment of hydromorphological lake shore degradation using benthic littoral invertebrates

Oliver Miler[1], Jukka Aroviita[2], Ken Irvine[3], Tamara Jurca[4], Elaine McGoff[5], Leonard Sandin[6], Francesca Pilotto[1, 7], Gwendolin Porst[1], Angelo Solimini[7] and Martin Pusch[1]

[1] Leibniz Institute of Freshwater Ecology and Inland Fisheries, Berlin, Müggelseedamm 301, 12587 Berlin, Germany
[2] Finnish Environment Institute (SYKE), Freshwater Centre, P.O.Box 413, FI-90014 Oulu, Finland
[3] UNESCO-IHE Institute for Water Education, Westvest 7, PO Box 3015, 2601 DA Delft, The Netherlands
[4] School of Natural Sciences, Trinity College Dublin, Ireland
[5] Swedish University of Agricultural Sciences (SLU), P.O. Box 7050, SE 750 07 Uppsala, Sweden
[6] Department of Bioscience Aarhus University, Vejlsøvej 25, P.O. Box 314, DK-8600 Silkeborg, Denmark
[7] Department of Public Health and Infectious Diseases, La Sapienza University of Rome, Viale Regina Elena 324, 00161 Rome, Italy

Email address of corresponding author: miler@igb-berlin.de

Key words: benthic invertebrates, hydromorphological degradation, lakes, littoral zone, Lake Habitat survey, multimetric index, stressor index

Introduction

The European Union Water Framework Directive (EU WFD) demands from the EU member states to develop ecological assessment tools for lakes, streams and coastal/transitional waters. Assessment results then can be taken to decide about appropriate restoration measures in order to achieve a good ecological status of all natural surface water bodies. The ecological assessment is based on Biological Quality Elements (BQEs), i.e. phytoplankton, macrophytes/phytobenthos, fish and benthic invertebrates. While the trophic status of many lakes has been substantially improved in the last decades, recent studies (Brauns et al. 2007, Strayer & Findlay 2010) have pointed out the importance of littoral zones to lakes, and the ecological effects of hydromorphological degradation. Since macrozoobenthos communities of the eulittoral zone of lakes are sensitive to hydromorphological degradation (Brauns et al. 2007), their functional and taxonomic community composition is a suitable indicator for this environmental stressor. Hence in the WISER WP 3.3 we developed multimetric indices for several European biogeographic regions that may be used to assess hydromorphological lake shore alterations based on benthic macroinvertebrate surveys.

Materials and Methods

Within the WP 3.3, macroinvertebrate communities in the eulittoral zone of lakes in Germany, Ireland, Sweden and Italy were sampled to assess the hydromorphological degradation of the shore. The sampling campaign included lakes from three trophic levels (eutrophic, mesotrophic and oligotrophic), which were sampled at shoreline sections representing three hydromorphological degradation levels (unmodified, moderately modified and highly modified), with a minimum of 9 lakes sampled in each of the four countries. For the representation of degradation levels natural shore sections (unmodified level), recreational beaches ('soft alteration', moderately modified level)) and sites with artificial physical bank protection ('hard alteration, e.g. through walls or rip-rap) were chosen. Each alteration type was replicated three times per lake, resulting in nine samples for each lake. In addition, it was decided to sample cross-BQE lakes where all four BQEs where studied within the WISER project. Even though these lakes did not always contain all three alteration types or where located in other countries or ecoregions, these lakes were added to the above described sampling scheme.

Hence, the eulittoral zones of lakes were sampled in Germany (9 lakes), Denmark (2 lakes), Ireland (9 lakes), United Kingdom (3 lakes), Sweden (9 lakes), Finland (4 lakes) and Italy (15 lakes altogether; 8 lakes in the subalpine and 6 lakes in the Mediterranean region). With respect to the macrozoobenthos sampling methodology, at each sampling site a 1 minute composite sample, integrating the percentage distribution of the habitats occurring at the sampling site, was taken. In conjunction with the macrozoobenthos sampling, properties of the physical structure of the lake shores were recorded using the Lake Habitat Survey (LHS) method (Rowan 2008). These data were then used to construct a stressor index representing the degree hydromorphological degradation, which was necessary to calibrate the final multimetric index and its component metrics.

Results and Discussion

Development of a typology based on faunal assemblages

Since the sampled lakes were located in several biogeographical regions, first the development of a typology based on dissimilarities in macrozoobenthos assemblages was necessary to account for natural differences in benthic macroinvertebrate community composition. A quantitative biocoenotic differentiation between countries/regions was achieved through an Analysis of Similarities (ANOSIM) in macroinvertebrate community compositions together with a Multidimensional Scaling (MDS) plot (Fig. 1). Results showed marked differences among countries (and the regions of northern and central Italy), indicated by high R values in the ANOSIM analysis. R values for the comparisons among Germany/Denmark, Ireland/United Kingdom, Sweden/Finland and central Italy/northern Italy were lower than 0.85. Samples from lakes in Denmark (2 lakes), United Kingdom (3 lakes) and Finland (4 lakes) could not be analysed specifically because of low observation numbers. Based on this typology, a stressor index and a multimetric index were developed for each of the four country (region) pairs Germany/Denmark, Ireland/United Kingdom, Sweden/Finland and central Italy/ northern Italy.

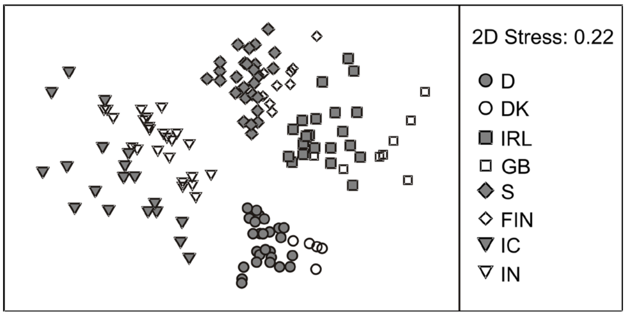

Figure 1: Multidimensional scaling (MDS) plot of the macroinvertebrate community composition in the studied lakes. The plot clearly mirrors the geographical distances among the regions and countries sampled (D = Germany, DK = Denmark, IRL = Ireland, GB = United Kingdom, S = Sweden, FIN = Finland, IC = central Italy, IN = northern Italy).

Development of a stressor index

In order to develop a stressor index that indicates the degree of hydromorphological alteration of lake shores, the LHS results were tested with an Analysis of Variance (ANOVA) testing their differences among the three pre-defined alteration types. As a result, eight environmental variables were identified that differentiated well between the three alteration types, which were hence regarded as potential stressor index components (Table 1).

Table 1: Components of the hydromorphological stressor index developed for the four biogeographical regions (D = Germany / DK = Denmark, IRL = Ireland / GB = United Kingdom, S = Sweden / FIN = Finland, IC = central Italy / IN = northern Italy).

	Biogeographical Region			
Stressor Index Component	*D/DK*	*IRL/ GB*	*S/FIN*	*IC/IN*
Number of habitats	X			
Habitat diversity		X	X	X
Total PVI (percentage volume inhabited (by macrophytes))	X		X	X
Sum of macrophyte types		X		
Total sum of vegetation cover types	X	X	X	
Sum of coarse woody debris (CWD), roots & overhanging vegetation	X			X
Total (anthropogenic) pressure index	X	X	X	X
Presence/absence of natural/artificial dominant land cover type				X

These stressor index components were then normalized to values ranging from 0 to 1. Hereby the 5 % percentile was set to 0 and the 95 % Percentile was set to 1. Values smaller than 0 or larger than 1 were set to 0 and 1, respectively. In a second step these values were classified in a scale from 1 (best condition) to 5 (worst condition). The scaling for all stressor index components after Hering et al. (2006) and Vlek et al. (2004) was as follows: 0 - 0.2 = 5, 0.21 - 0.4 = 4, 0.41 - 0.6 = 3, 0.61 - 0.8 = 2, 0.81 - 1 = 1. For the definition of the "total (anthropogenic) pressure index" the scaling was then reversed since this variable correlates positively with increasing degradation. In a next step, several stressor index variants were calculated as the unweighted means from the stressor index components, and tested again with an ANOVA to which degree they mirrored the differences among the 3 alteration types. Hereby variables that showed a cross-correlation (Spearman Rank Correlations) with Rho > 0.8 were not used together in the same stressor index variant, as they describe the same environmental information. This was the case for the variables "number of habitats"/"habitat diversity" and for "total PVI (percentage volume inhabited (by macrophytes))"/"sum of macrophyte types". The variant that reflected best differences among the alteration types, and especially between hard and soft shore modification, was chosen for each biogeographical region

(Table 1). Since the values of the stressor index components were classified in a scale from 1 (best condition) to 5 (worst condition), the values of the stressor index were also in the range from 1 to 5.

Selection of candidate metrics and construction of a multimetric index (MMI)

Invertebrate metrics were calculated based on macroinvertebrate abundances and abundance classes (AC): 1-2 = AC 1, 3-10 = AC 2, 11-30 = AC 3, 31-100 = AC 4, 101-300 = AC 5, 301-1000 = AC 6, > 1000 = AC 7. Metrics based on abundance classes have the advantage of being less influenced by a few dominant taxa with very high densities. First a boxplot of each metric was plotted to check if the respective metric had a narrow range of values, a highly skewed distribution of values and/or many outliers. If one of these cases was true it would be numerically unsuitable. Subsequently, the metrics were correlated with the stressor index via Spearman Rank Correlations. From this dataset a subset of metrics with Rho>0.2, i.e. metrics that correlated well with the stressor index, was chosen and each metric normalized to values from 0 to 1 with the 5 % percentile set to 0 and the 95 % percentile set to 1 again. Values smaller than 0 or larger than 1 were set to 0 and 1, respectively. Finally, for each biogeographical region eight candidate metrics were chosen (Table 2) that (1) correlate with the stressor index with Rho > 0.2, (2) cross-correlate with each other with Rho < 0.8 and (3) equally represent the four metric types diversity (D), taxonomic and functional composition (TFC), abundance (A) and disturbance sensitive taxa (DST) according to the normative text of the EU WFD. From the eight candidate metrics 32 multimetric index (MMI) variants with the unweighted mean of three or four metrics covering at least the metrics types TFC, D and DST, ideally also A, were constructed. These were correlated in Spearman Rank correlations with the stressor index and the best correlating MMI variant was chosen (Table 2). The final multimetric index had values from 0 to 1, and in a second step these values were re-classified to a scale from 1 to 5. The re-scaling for all stressor index components to ecological quality status classes according to the EU WFD was as follows: 0 - 0.2 = 5 ("bad"), 0.21 - 0.4 = 4 ("poor"), 0.41 - 0.6 = 3 ("moderate"), 0.61 - 0.8 = 2 ("good"), 0.81 - 1 = 1 ("high").

Table 2: Selected candidate metrics for the 4 biogeographical regions (D = Germany / DK = Denmark, IRL = Ireland / GB = United Kingdom, S = Sweden / FIN = Finland, IN = northern Italy / IC = central Italy). Core metrics for the final MMI variant are marked in bold and Capital letters. TFC = taxonomic and functional composition, D = diversity, A = abundance, DST = disturbance sensitive taxa. AC = abundance class.

Candidate metric	Metric type	D/DK	IRL/GB	S/FIN	IC/IN
Shannon Wiener diversity	*D*	*x*			
Margalef diversity		*X*	*X*		*X*
No. taxa			*x*	*X*	*x*
No. families				*x*	
% AC Type Lithal	*TFC*			*x*	
% AC Type Pelal			*x*		
% Type POM					
% AC Type POM		*x*			*x*
% Gatherer/ Collectors			*X*		
% AC Gatherer/ Collectors		*X*			
% AC Shredders				*X*	
r/K relationship					*X*
Gastropoda % AC	*A, TFC*				*x*
Odonata %				*x*	*X*
Odonata % AC		*x*			
Chirononomidae % AC		*X*			
Trichoptera %			*x*		
Diptera % AC			*X*		
Crustacea % AC				*X*	
No. ETO taxa	*DST*	*x*	*X*		*X*
No. EPTCBO taxa		*X*	*x*	*x*	*x*
No. Odonata taxa				*X*	

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Test of evenness of phytoplankton as an index for eutrophication

Ute Mischke[1], Laurence Carvalho[2], Geoff Phillips[3], Caridad de Hoyos[4], Christophe Laplace-Treyture[5] & Anne Lyche Solheim[6]

Email address of corresponding author: mischke@igb-berlin.de

[1] IGB, D-12489 Berlin, Justus von Liebig Str. 7
[2] CEH (UK)
[3] Environmental Agency (UK)
[4] CEDEX (ES), 5 CEMAGREF (FR)
[6] NIVA (NO)

Keywords: phytoplankton; diversity; indicator; eutrophication; European lakes

Introduction

The main thesis of this paper is that diversity of phytoplankton is expected to decrease with eutrophication and can help to identify a bloom situation. This may serve for a biological indicator (see Mischke et al. 2010). In situations of blooms it is expected that diversity is decreasing, while the total biomass of phytoplankton is increasing. Diversity indices were tested against eutrophication stress (summer mean, July - September) by total phosphorus (TP) concentration and in lake groups.

Data set

The EU-WISER project generated a trans- European database for phytoplankton in lakes. Phytoplankton taxa from 19 countries and more than 35 different data providers were harmonized by a new European taxa list (1789 taxa). The data were provided by the country authorities based all on counts by Utermöhl technique and transferred to taxa biovolumes. Still, the counting protocol deviated according counted sub-sample size, and level of taxa determination as so the sampling depth integral. Most data were from Nordic lakes (1010 of 1710 lakes). In mean 31 taxa were found per sample and less than 10 taxa only in 3% of total. Concentration of TP range between 1 and 1000 µg/L: In Nordic lakes TP concentrations were much lower (mainly 5-20µg/L TP; median = 10µg/L) than in the Central Europe lakes (20 – 50µg/L TP; median = 44µg/L). To test for correlation of diversity to TP, the lakes were first grouped according the lake types used for the European intercalibration process (very shallow (vs), shallow (s), low, high and very high alkalinity and eco-region like Nordic, Central Baltic and Mediterranean region). In a second step those lake types were put together which showed a significant trend of evenness to TP within one eco-region.

Check for influence of counting strategy on diversity indices

Since up to 100 species were detected in some samples whilst the examination of others revealed only 3 – 10 species, the taxonomic level of determination was strongly uneven in the whole data set. The influence of the different skill and effort among individual phytoplankton analysts to affect the evenness index calculation was of concern. Thus, a sub-data set was constructed by restricting the number of taxa to 20 per sample by selecting only the most 20 abundant taxa.

Correlation of evenness values is high, when using all taxa in the count and comparing with the evenness calculated when only the top 20 most abundant taxa were included ($r^2 = 0.9461$). In general there is only a slight tendency that evenness increases with total number of taxa per sample (x):

$$J' = 0.0017x + 0.5487 \quad (r^2 = 0.0297).$$

For Nordic lake samples an unique counting strategy was applied, so this sub data set was used to check for the distribution on species richness along the TP gradient. The distribution pattern is the same with a decrease of species richness above concentrations of 100 µg L-1 TP. The taxa number per sample is 36.4 and up to 107 taxa in the 1010 Nordic lakes, thereby strongly higher than in the rest of data (mean 24.5 taxa / sample).

Species richness to TP

Species richness was related to productivity of lakes in a hump-shaped distribution (Fig. 1): The number of taxa was low at TP concentrations in a lot of cases in the oligotrophic range, was highest in mesotrophic to eutrophic lakes and decreased in hypertrophic lake systems. Therefore, evenness as a further diversity index was tested.

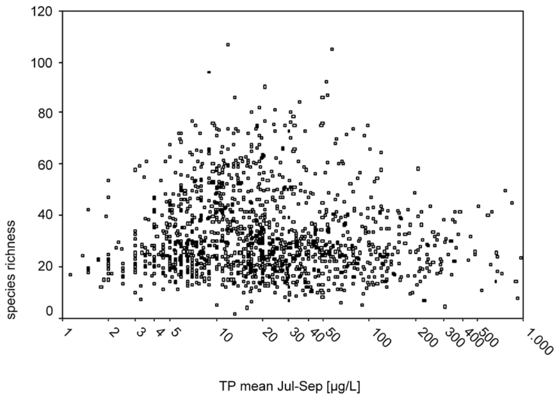

Fig. 1: XY plots of species richness per sample to summer TP concentrations in all samples

Evenness to TP

Evenness (J') was calculated for each single sample according to Pielou's evenness index: J`= H' / H' max, when H´max is the maximum value of H' (Shannon index). Index values near 1 are communities, when all taxa contributed very evenly to total biovolume, versa vice values below 0.2 present communities with high dominance of species. The observed evenness values of phytoplankton covered the whole spectrum of theoretically possible index values (0.03 - 0.98 in mean of Jul – Sept).

Lakes of all lake types, defined for the European comparison of nations method results (intercalibration process), where tested separately for a significant trend in the correlation of evenness to pressure (here TP concentrations)

The relationship of evenness (J') to TP shows the expected negative trend (Fig. 2), but beforehand exclusion of some lakes types were necessary, which had no or a negative trend (all lakes of the Mediterranean and Eastern European region, and Lobellia- shallow lakes of the lowland, L-CB3). When grouping the rest of common lakes types together in two remaining groups, two different steep and significant trends could be observed for the very shallow and shallow lowland lakes in the ecoregion Central European (CB) on the one hand, and for the clear or meso-humic Nordic shallow lakes with low and moderate alkalinity (N like N1, N2a,N3a, N8a,N9, NU,NX).

The linear regression model, which fits best to the data of each lake group is J' = 0.704+ -0.078 * logTP for Central European lakes (CB) and is J' = 0.777 + -0.140 * logTP for Nordic European lakes (N).

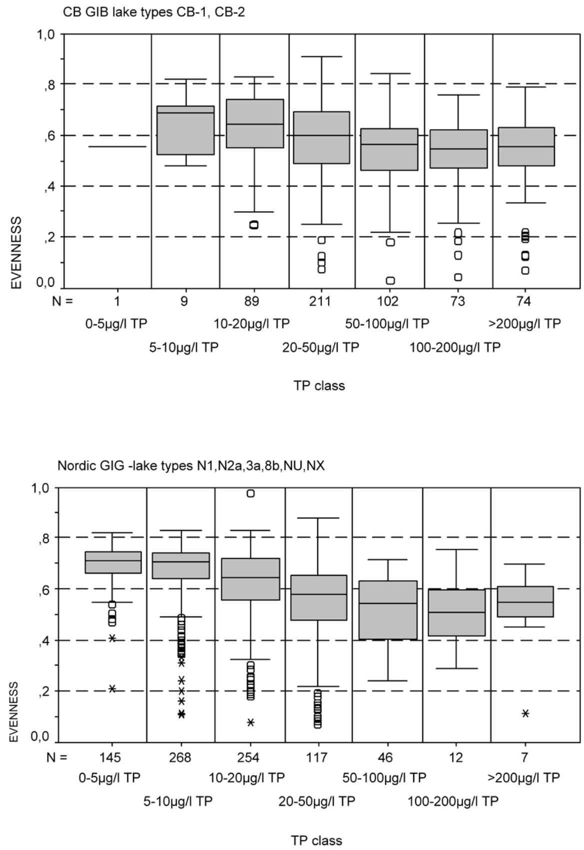

Fig. 2: Box plots of observed evenness in seven TP classes in Northern (left) and in Central European lakes in summer phytoplankton (mean of July to September).

The decline of evenness along the TP gradient was much steeper in Nordic lakes (low to moderate alkalinity; clear or humic) than in the lakes of Central Europe and in the Baltic region (high alkalinity; very shallow or shallow partly stratified).

To define an ecological quality ratio (EQR) by evenness, which distinguish the five status classes requested by the European Water Framework Directive, the linear regression models for N and CB lakes were converted to regression functions. The EQR value 1 was set at the mean evenness within the reference lake group of Nordic lakes and Central Baltic lakes respectively. The EQR value 0.6 (Good/Moderate boundary) was set as the evenness value detected at 25 µg/l TP by linear regression for each eco-region group. The EQR value 0.2 (Poor/Bad boundary) was set as evenness value at 200 µg/l TP. In the next step, the High/Good and the Moderate/Poor boundaries of evenness were determined by simple interpolation between the set points.

Finally the evenness index predicts ecological quality as a normalized ratio (EQR), which using the linear regression along the derived boundary values by following equations, while resulting EQR values >1 are set as 1 (highest status) and EQR values <0 are set as 0 (most bad status):

EQR_CB = 6.8966*J' - 3.469

EQR_N = 2.7933*J' - 0.8804

For Nordic lakes, the predicted EQR values decrease significantly (Spearman 0.001) when correlated to mean TP concentrations. Lakes with TP mean concentrations below 20µg/L are assessed by evenness mainly as high or good status (EQR >0.6) except of the few cases, in which critical gigh biomass densities were observed (N = 7).

Conclusions

The diversity index evenness (J') was calculated and analysed for the summer phytoplankton communities of 1590 European lakes. Evenness distribution exhibits a significant correlation to pressure (here total phosphorus concentration) in several of the most common lake types covering about 80% of all investigated lakes.

If it is accepted that an unbalanced phytoplankton communities can be used as a response to high pressure, the suggested evenness metric is able to detect and assess such degradations even in cases when the total biomass index is not able to detect the pressure status by remaining below the Good/Moderate boundaries.

In Nordic lakes this is a very seldom case, since in this lake group high biomasses are well correlated to low evenness values. So, the loss of evenness with increasing nutrient stress is steep and significant enough to be used as a biological indicator in lakes of the Nordic countries.

Additionally the analysis presented here demonstrates that evenness significantly decreases in almost all CB lakes (L-CB1 and L-CB2) with increasing pressure. Still, very high uncertainties were found in additional test within the geographic intercalibration groups, so the use of the evenness index is not recommended for lakes from the Central and Eastern Europe region as so for Mediterranean lakes.

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Hydromorphological pressures and aquatic macrophytes – how to evaluate the effects of water level fluctuation in Nordic lakes?

Marit Mjelde[1], Seppo Hellsten[2] & Frauke Ecke[3]

[1] Norwegian Institute for Water Research, Gaustadalléen 21, NO-0349 Oslo, Norway
[2] Finnish Environment Institute (SYKE), P.O.Box 413, FIN-90014 University of Oulu, Finland
[3] Department of Aquatic Sciences and Assessment, Swedish University of Agricultural Sciences, Box 7050, SE-750 07 Uppsala, Sweden

Email address of corresponding author: Marit.Mjelde@niva.no

Key words: water level regulation, aquatic macrophytes, hydropower, lakes

Introduction

Hydromorphological pressures in lakes are related to the human need to control water levels of lakes and flows of rivers for production of hydropower, flood prevention, recreation, navigation, and supply of water for agricultural or human consumption. Regulation practices vary among systems and countries and depend on the objectives of regulation.

Macrophytes are one of the key indicators of hydromorphological changes in lakes. Because macrophytes grow in the littoral zone they are sensitive for changes in the water level fluctuation regime. Effects are enhanced in lakes covered by ice, because effects of down-dwelling ice are especially harmful for freezing sensitive plants. The aim of this study was to develop a suitable index for evaluation of effects of water level fluctuation in Nordic ice-covered lakes.

Material and methods

A total of 79 lakes from Finland, Norway and Sweden were used in developing the new waterlevel index (WIc). Of these, 37 were storage lakes (H3), 20 other regulated lakes (H2) and 22 natural (N2) or semi-natural lakes (sN2) (Rørslett 1988). The Finnish dataset included low alkalinity, both clear and humic, lakes. Water level fluctuation varied between 0.1 and 6.8 m. The Norwegian dataset consisted mainly of clear water, low alkalinity lakes, with water level fluctuations between 0.1 and 5.7 m. The Swedish dataset sampled by Wallsten (2010) included low alkalinity lakes in the county of Värmland with a wide range in colour. All lakes in the dataset are oligotrophic to slightly mesotrophic lakes.

The aquatic macrophytes were surveyed by WFD compliant methods ranging from eulittoral zone to deepest growing points. Only real aquatic macrophytes (isoetids, elodeids, nymphaeids, lemnids and charophytes) were included in further analysis.

We used winter drawdown as an indicator of water level regulation amplitude (Hellsten 2001). Winter drawdown was calculated as the average difference between highest water level in October-December and lowest level during the following April-May.

Defining the index

We analysed the sensitive and tolerant species for winter drawdown by using a subset of 66 oligotrophic and low alkalinity lakes from Finland (n=29), Norway (n=25) and Sweden (n=12).

We suggest the following description of sensitive and tolerant species:

Sensitive species: species that prefer or only occur in reference lakes. Decreased frequency and abundance (often disappearance) if water level fluctuations increase. Some of the sensitive species appear to be less effected by winter drawdown. We call these species less sensitive species, since they suffer to some extent of water level fluctuation.

Tolerant species: species with increased frequency and abundance when water level fluctuations increase. Often less frequently occurring in reference lakes.

Identification of sensitive and tolerant taxa is accomplished by analysing the species occurrence along the winter drawdown gradient. To distinguish between sensitive and tolerant species we sort on the 75th percentile, which placed e.g. Isoetes lacustris within the sensitive group and Juncus bulbosus among the tolerant species.

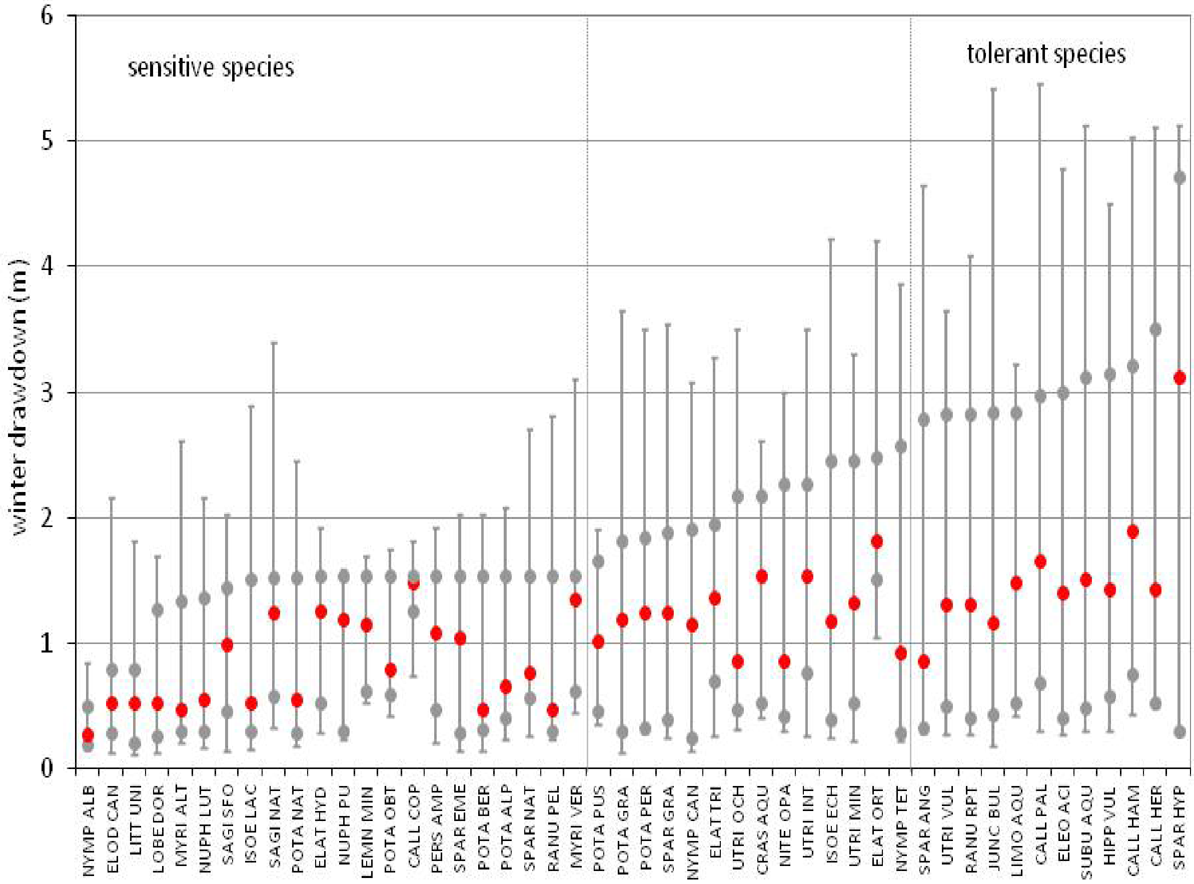

Figure 1: Distribution of sensitive and tolerant species along a gradient of winter drawdown, based on Finnish, Swedish and Norwegian lakes. The graph includes 10, 25, 50, 75, and 90th percentiles.

The level of winter drawdown used to separate the two groups is mainly based on expert judgement. We have used changes in frequency and abundance of well known sensitive or tolerant species to help us decide which level to use. Based on this method we can identify the most sensitive species as: all with 75^{th} percentiles <1.6 m winter drawdown, while the most tolerant species seem to be species with 75^{th} percentiles >2.6 m winter drawdown (Figure 1).

Hellsten and Mjelde (2009) suggested a water level index (WIc) using macrophytes to describe the ecological status or ecological potential for regulated lakes. Based on this preliminary work we have improved the water level index – WIc(i), by improving the determination of sensitive and tolerant species. while the equation is the same as earlier:

$$WI_{C(i)} = \frac{N_S - N_T}{N} \times 100$$

where WIc is the water level regulation index, NS is the number of sensitive species, NT is the number of tolerant species, and N is the total number of species in the lake, including the less sensitive.

Analysis of data

The water level regulation index WIc(i) correlated well with winter drawdown in the storage reservoirs (H3) for all countries (Figure 2, r^2=0.77, 0.67 and 0.73 for Finnish, Norwegian and Swedish lakes, respectively. The weaker correlation for the Norwegian lakes may be explained by the steep littoral zone of some Norwegian lakes being dominated by stones.

Some natural or slightly regulated lakes (H2) showed low index values, especially among the Norwegian lakes. This is mainly due to the littoral zone dominated by stones (see above). However, most natural and slightly regulated lakes have index values higher than -20.

The lakes in the H2 group and the natural lakes (N2 and sN2) normally have smaller water level fluctuation range than the storage lakes. In addition, hydrological regimes are very heterogeneous. Therefore, the correlation between WIc(i) and winter drawdown in these lakes is week. These lakes were therefore not included in the boundary setting assessments.

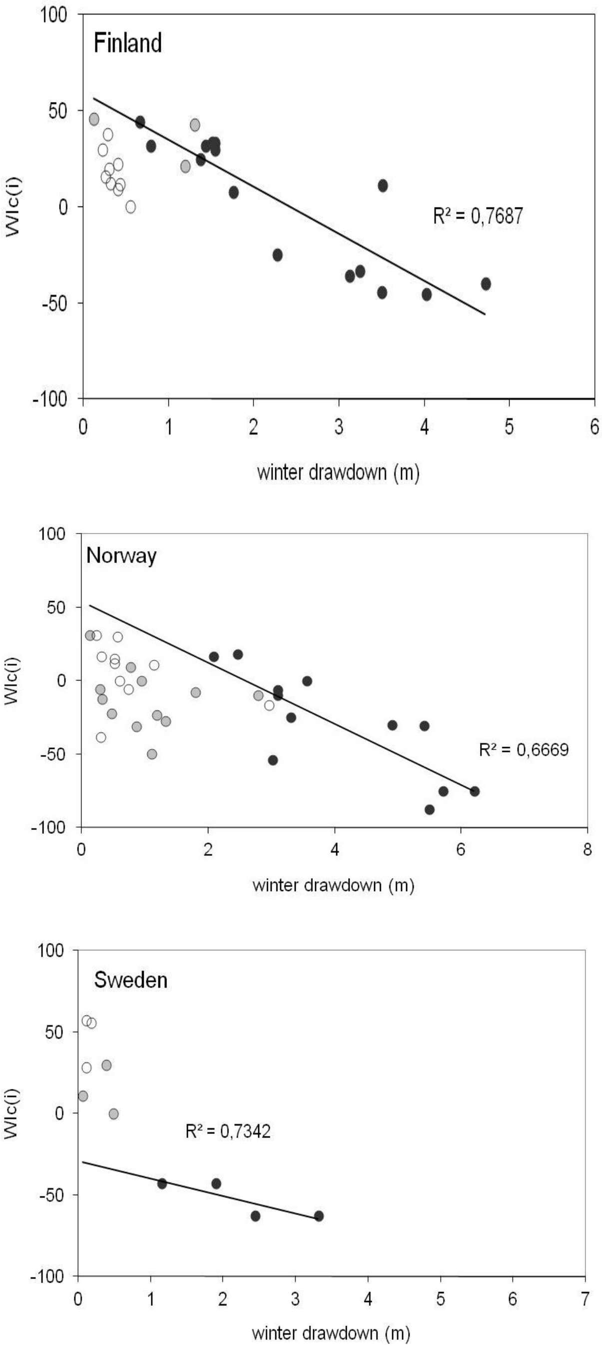

Figure 2: Improved water level regulation index WIc(i), divided by Finland, Norway and Sweden. Dark circles: H3 lakes, grey circles: H2 lakes, and open circles: natural lakes.

The slope for the Swedish lakes is different from the Finnish and Norwegian lakes. The reason for this may be the very low number of species observed in some of the Swedish lakes. The Swedish method is based on virtual transects. If an insufficient number of transects is studied, the resulting dataset might result in an incomplete species list.

Until this dissimilarity is further investigated, the index and suggested boundaries will only be applicable for Finland and Norway. Figure 3 shows the regression between the improved index and the pressure for Finnish and Norwegian storage reservoirs.

Summary

Using a relatively simple division into sensitive and tolerant species for winter drawdown in Nordic lakes produced a tool that could further be utilised for extracting lakes with varying status of ecological potential. General implementation of river basin management plans will support pressure specific tools that can be used in monitoring the effects of possible mitigation methods.

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The WISER Central Database: content, structure and functions

S. Jannicke Moe[1], Roar Brænden[1], Bernard J. Dudley[2], Jan Karud[1], Astrid Schmidt-Kloiber[3], Jörg Strackbein[4], Robert Vogl[3]

[1] Norwegian Institute for Water Research (NIVA), Gaustadalléen 21, 0349 Oslo, Norway
[2] Centre for Ecology & Hydrology (CEH), Bush Estate, Penicuik, Midlothian, EH26 0QB, UK
[3] University of Natural Resources and Life Sciences (BOKU), Institute of Hydrobiology and Aquatic Ecosystem Management, Max-Emanuel-Straße 17, 1180 Wien, Austria
[4] University of Duisburg-Essen (UDE), Faculty of Biology, Universitätsstrasse 5, D-45141 Essen, Germany

Email address of corresponding author: jmo@niva.no

Key words: Biological quality elements (BQE); database structure; data extraction; intellectual property rights (IPR)

Background

The main objective of WISER Workpackage 2.1 Data service has been to assist the WISER partners with obtaining efficient access to relevant project data (http://www.wiser.eu/programme/data-and-guidelines/data-services/). The project data consisted of new data from the WISER field exercises, existing data from previous projects and from ongoing monitoring programmes provided by WISER partners, as well as data provided from external collaborators in the Geographical Intercalibration Groups (GIGs).

A key task of WP2.1 has been to design, construct and manage a Central Database (CDB) holding all these project data. The main purpose of the CDB was to serve WPs in Modules 5 and 6 with data collected by WPs in Module 3 and 4 (Figure 1). The CDB structure was developed in dialogue with the data managers of each WP. In particular, the CDB was designed to enable the following data processing, across all WPs: (1) combination of data on different biological quality elements (BQEs); (2) combination of biological data with environmental data from the same waterbodies; and (3) a consistent approach to uncertainty analysis for different BQEs.

In order to facilitate data flow within the project, all WP data managers were encouraged to use the WISER CDB structure also within their WP (although this was optional). Based on this database structure, the WP data managers were offered templates, code lists, tools and guidelines for data import and export.

This presentation will focus on the content, structure

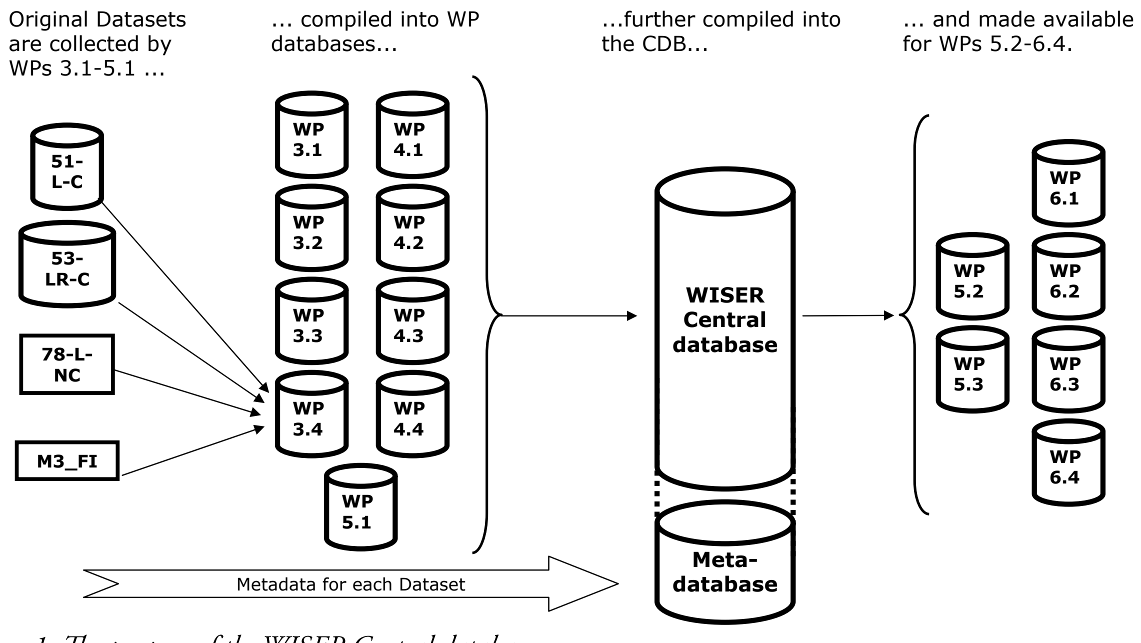

Figure 1. The purpose of the WISER Central database.

and functions of the WISER Central Database. Other poster presentations at this conference will focus on (1) the metadatabase that provides information on availability and accessibility of all project data for the project partners via the project website (http://www.wiser.eu/results/meta-database); and (2) a data extraction tools that facilitates data export into a single table suitable for data analysis.

Content of the WISER Central Database

The Central DB 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. 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. 8300 samples from 405 stations in 69 waterbodies 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

Table 1: Content of the WISER Central Database

Foreground data									
WP	**# Countries**	**Countries**	**# Water-bodies**	**# Stations**	**# Biol. samples**	**# Biol. values**	**# Env. samples**	**# Env. values**	**WP data manager**
3,1	*11*	*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	*10*	*DE, DK, EE, FI, FR, IT, NO, PL, SE, UK*	*28*	*161*	*6.725*	*7.497*	*0*	*0*	*Bernard Dudley*
3,3	*8*	*DE, DK, EE, FI, IE, IT, SE, UK*	*53*	*150*	*96*	*2.159*	*150*	*150*	*Oliver Miler, Mario Lepage*
3,4	*3*	*DE, IT, UK*	*21*	*333*	*452*	*4.867*	*0*	*0*	*Stephanie Pedron, Simon Causse*
4,1	*4*	*BG, ES, FI, IT*	*6*	*43*	*42*	*2.903*	*0*	*0*	*Karsten Dromph*
4,2	*5*	*BG, ES, IT, NO, PT*	*8*	*72*	*331*	*1.881*	*8.357*	*25.847*	*Rosa G. Novoa*
4,3	*4*	*ES, IT, NO, PT*	*10*	*61*	*165*	*8.592*	*56*	*559*	*Karl Norling*
4,4	*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*	

Background data									
WP	**# Countries**	**Countries**	**# Water-bodies**	**# Stations**	**# Bio samples**	**# Bio values**	**# Env samples**	**# Env values**	**WP data manager**
3,1	*21*	*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	*12*	*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	*8*	*BE, DE, EE, LT, LV, NL, PL, UK*	*180*	*635*	*889*	*23.016*	*0*	*0*	*Juergen Boehmer*
3,4	*18*	*DK, EE, FI, FR, IE, IT, LV, LT, NO, PT, RO, SI, ES, SE, UK*	*2.173*	*54.851*	*72.245*	*558.993*	*0*	*0*	*Stephanie Pedron, Simon Causse*
4,2	*2*	*BG, ES*	*32*	*63*	*1.836*	*6.463*	*3*	*3*	*Rosa G. Novoa*
4,4	*4*	*ES, FR, PT, UK*	*67*	*2.363*	*3.416*	*6.165*	*3.229*	*16.007*	*Anne Courrat*
5.1	*10*	*AT, CZ, DE, DK, FR, NL, PL, SE, SK, UK*	*3.085*	*4.349*	*18.152*	*528.623*	*14.558*	*134.602*	*Andreas Melcher, Martin Seebacher*
Sum	*26*		*12.882*	*74.506*	*115.123*	*1.614.870*	*141.634*	*918.837*	

biological samples and ca. 140 000 environmental samples from rivers, lakes and coastal/transitional waters 26 countries.

Structure of the WISER Central Database

We aimed at developing a database structure 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 other WPs. The resulting structure (Figure 2) is somewhat complex, but flexible. One important feature of the CDB is a set of sampling information fields, which in combination provide a common definition of unique samples across all WPs. This definition was important for a consistent uncertainty analysis across WPs.

The "unit" of the CDB is termed "Dataset". Each Dataset is defined by a code (DatasetID) and is represented by a unique record in the metadatabase. The DatasetID is thus critical for linking metadata, such as intellectual property rights (IPR), to the actual data used in data analysis. More information on IPR for each dataset can be found in an overview on the website, under http://www.wiser.eu/results/meta-database/).

The CDB has a hierarchical structure, with tables corresponding to the hierarchical levels of the WISER field campaign: Dataset, Waterbody, Station, Sample, and Value. There are separate sample tables for biological samples and environmental samples. In addition there is a separate table "t_EnvWaterbody" for constant environmental data associated with waterbodies (i.e. without sampling date). This information is kept in a separate table for two reasons: (1) to limit the size of the more fundamental the Waterbody table, and (2) to enable aggregation of environmental information for waterbodies that appear multiple times in the Waterbody table (e.g. reported from different WP databases).

Functions of the WISER Central Database

The CDB has been subject to several updates, because the underlying WP databases have been delivered at different times throughout the project period. During the project period, therefore, WP2.1 dealt with data requests upon demand from temporary CDB versions in MS Access. After the project period, the CDB will no longer be updated. The final CDB version in Oracle will be available from the WISER website. Via the metadatabase, potential data users can search for Datasets with certain properties, mark the Datasets they wish to export, and get these Datasets downloaded as a subset of the CDB. The downloaded file will be an MS Access database containing data in structure of the CDB (Figure 2). In addition, this database will contain a tool with a set of options for extracting the data into a single Excel table. Data users who wish to extract data in different ways than those provided by the tool, can receive guidelines on how to combine tables and set selection criteria etc.

Each biological and physico-chemical value in the

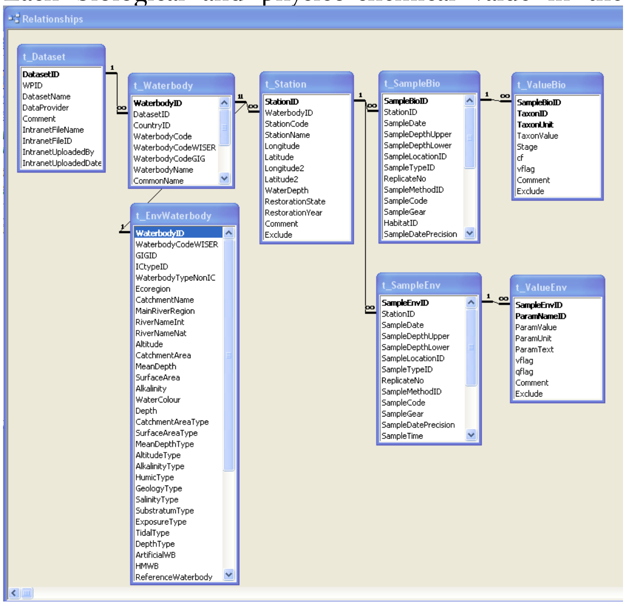

Figure 2. Relationships between tables of the WISER Central database.

CDB will be associated with a DatasetID (as explained above), which should always be included in data extractions. This way, a data user will know which Datasets each value belong to, and therefore have information on the IPR for each data point.

The hierarchical structure of the CDB facilitates aggregation and data analysis at different levels (GIG, country, waterbody, station, sample etc.). Furthermore, the common definition of "samples" for all WPs enables combined analysis of multiple BQEs.

The field "WaterbodyCodeWISER" can be used to identify multiple BQE data from the same waterbody, and to link the biological data to relevant environmental information for this waterbody. The WISER sampling campaign resulted in data from multiple BQEs from a number of waterbodies. In cases where a waterbody is recorded with different code from different WP databases, these records will nevertheless have the same "WaterbodyCodeWISER". For the historical data from lakes and coastal waters, harmonisation of waterbody coding has been carried out for countries with data on more than two BQEs. For data from rivers, however, the waterbody coding is completely harmonised.

Concluding remarks

All use of the WISER data during and after the project period must follow the intellectual property rights (IPR) as stated in the IPR section of the metadatabase and in the contract agreement. The data user is responsible for checking and following the IPR.

Due to the IPR associated with the Datasets, the download of WISER data will only be possible for project partners. Nevertheless, the metadatabase search functionality is also accessible for the public. This way, contact information for relevant WISER partners can be found for each Dataset. External persons who are interested in using the data are encouraged to contact the relevant WISER partners and propose collaboration using these data.

Acknowledgements

We acknowledge all WP data managers (see Table 1) for cooperation and data deliveries.

Climate change, restoration and ecological status in lakes: a Bayesian network modelling approach

S. Jannicke Moe[1], Tuomo Saloranta[2], Olli Malve[3], Niina Kotamäki[3], Anne Lyche Solheim[1]

[1] Norwegian Institute for Water Research (NIVA), Gaustadalléen 21, 0349 Oslo, Norway
[2] Norwegian Water Resources and Energy Directorate (NVE), P.O. Box 5091 Majorstua, NO-0301 Oslo, Norway
[3] Finnish Environment Institute (SYKE), P.O. BOX 140, FI-00251 Helsinki, Finland

Email address of corresponding author: jmo@niva.no

Key words: climate change, restoration, phytoplankton, ecological status classification, modelling

Background

The main objective of work package 5.2 was to address the impact of catchment management and climate change on pressures and ecological status in lakes. Climate change may impact ecological status directly and indirectly in multiple ways (Moe et al. 2010): e.g. by increasing physico-chemical pressures, by impacting the ecological baseline, by reinforcing the ecological response to a pressure gradient, or by reducing the ecological ability to recover. In this modelling study, we have focused on the combined impacts of restoration and climate change on ecological status based on phytoplankton (chlorophyll a). We considered only climate impacts directly on lake processes and leave out potential climate impacts on river basin processes (such as water discharge and nutrient transport). In addition, we modelled the effect of lake restoration in terms of reduced P loading. Our study considered altogether 9 scenarios: 3 levels of restoration (no change; -20% P loading; -40% P loading) combined with 3 levels of climate change (no change; +2 °C air temperature; +4 °C air temperature). We explored the impacts of these scenarios on the lake status class according to a biological quality element (phytoplankton) as well as to supporting physico-chemical elements (total P and total P). Since ecological classification of lakes is dependent on the lake type, we have selected two common lake types of Northern Europe as an example (L-N2a: altitude <200 m; L-N5: altitude 200-800 m).

Modelling approach

For this study we used a Bayesian network (BN) modelling approach. One of proposed the common approaches for all water categories (WPs 5.1-5.3) was development of conceptual models representing driver-pressure-impact-response-recovery chains. A Bayesian network can be developed as a conceptual model (Figure 1), but can also be parameterised and used as a simulation model. In brief, is variable (e.g. Total P, Chl-a) is

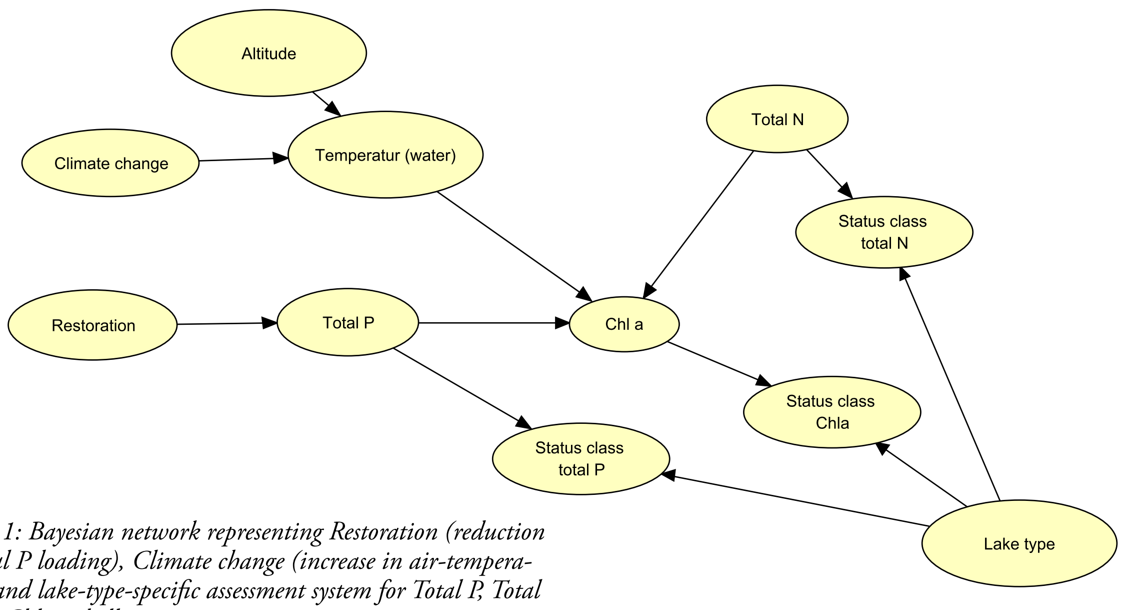

Figure 1: Bayesian network representing Restoration (reduction in Total P loading), Climate change (increase in air-temperature), and lake-type-specific assessment system for Total P, Total N and Chlorophyll-a.

illustrated by a node, which represents a discrete probability distribution (Figure 2). The cause-effect links are illustrated by arrows, which represent contingent probability tables (CPTs, Table 1). This modelling approach has many benefits, especially in relation to environmental risk assessment and management (Moe 2010): it can easily combine data or other information from different sources; it can explicitly model uncertainties (as probability distributions); and it can predict the probability of different outcomes of interest (such as different status classes).

Model construction and simulation

The construction of this Bayesian network model integrated two different lake models and data sources, from NIVA and SYKE respectively. The first part of the BN (effect of restoration and climate change on total P and temperature in lakes) was based on the "NIVA model", while the second part (effects of total P and temperature on Chl-a) was based on the "SYKE model" (http://lakestate.vyh.fi/). The NIVA model used the model code MyLake (Saloranta & Anderson 2007) to simulate

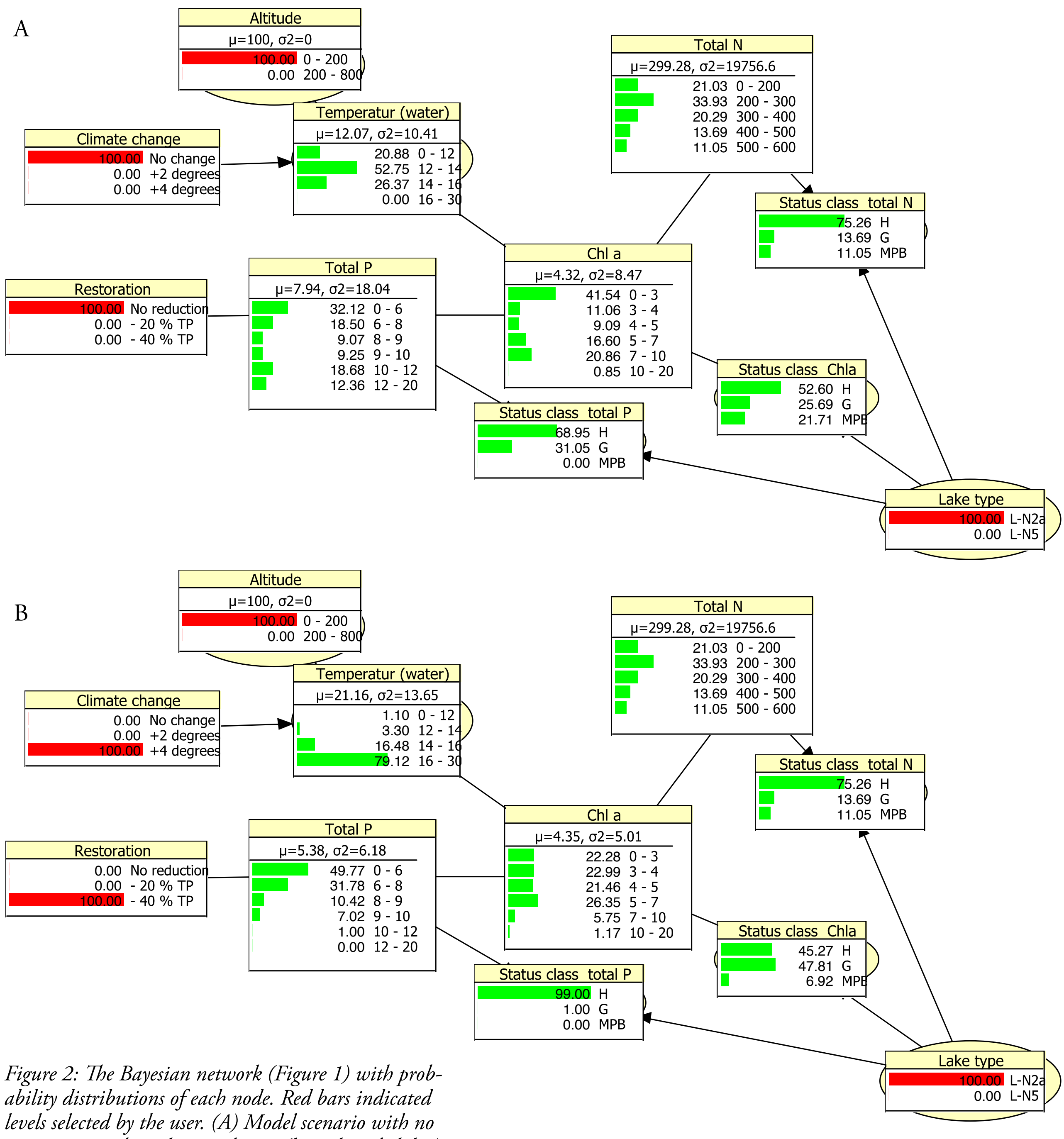

Figure 2: The Bayesian network (Figure 1) with probability distributions of each node. Red bars indicated levels selected by the user. (A) Model scenario with no restoration and no climate change (low-altitude lakes). (B) Model scenario with highest level of restoration and climate change (low-altitude lakes).

climate and restoration impacts for a large number of Norwegian lakes, of which 124 lakes (types L-N2a and L-N5) were selected for this analysis. The SYKE model is a hierarchical model that simulates chlorophyll-a from nutrients and temperature observations, based on lake-type-specific estimations. This model has applied data from WISER WP3.1 (Lakes phytoplankton) from all of Europe, of which 337 lakes (types L-N2a and L-N5) were used for this analysis. Only lakes with complete set of observations were used, which means that the dataset is not representative for the region in general (high-status lakes are overrepresented). For both data sources, average values for summer months were used in this study. Discretisation of continuous variables was based on (1) official class boundaries for TN, TP and chl-a (Finland), and (2) regression tree analysis for identifying threshold responses in the variables. The entries of the conditional probability tables (Table 1) were calculated as the proportion of data points falling into each combination of the parent node levels.

The BN model simulations were run by selecting a scenario and a lake type (marked red in Figure 2), and recording the resulting status classes according to TN, TP and chl-a. The model also provides expected value (μ) of e.g. chl-a, but with this modelling approach the probability distribution is more interesting than a point estimate.

Results

The predicted levels of nutrients and chl-a in this model exercise depends on many assumptions, of which not all can be justified. The most relevant result is therefore not the absolute probabilities, but the changes in probabilities (percentage points) across climate and restoration scenarios (Figure 3). TN was not affected by restoration or climate scenarios in this model, and remained as shown in Figure 2 across all scenarios. TP status class responded to restoration (reduction of P loading) by increased probability of High status. The highest

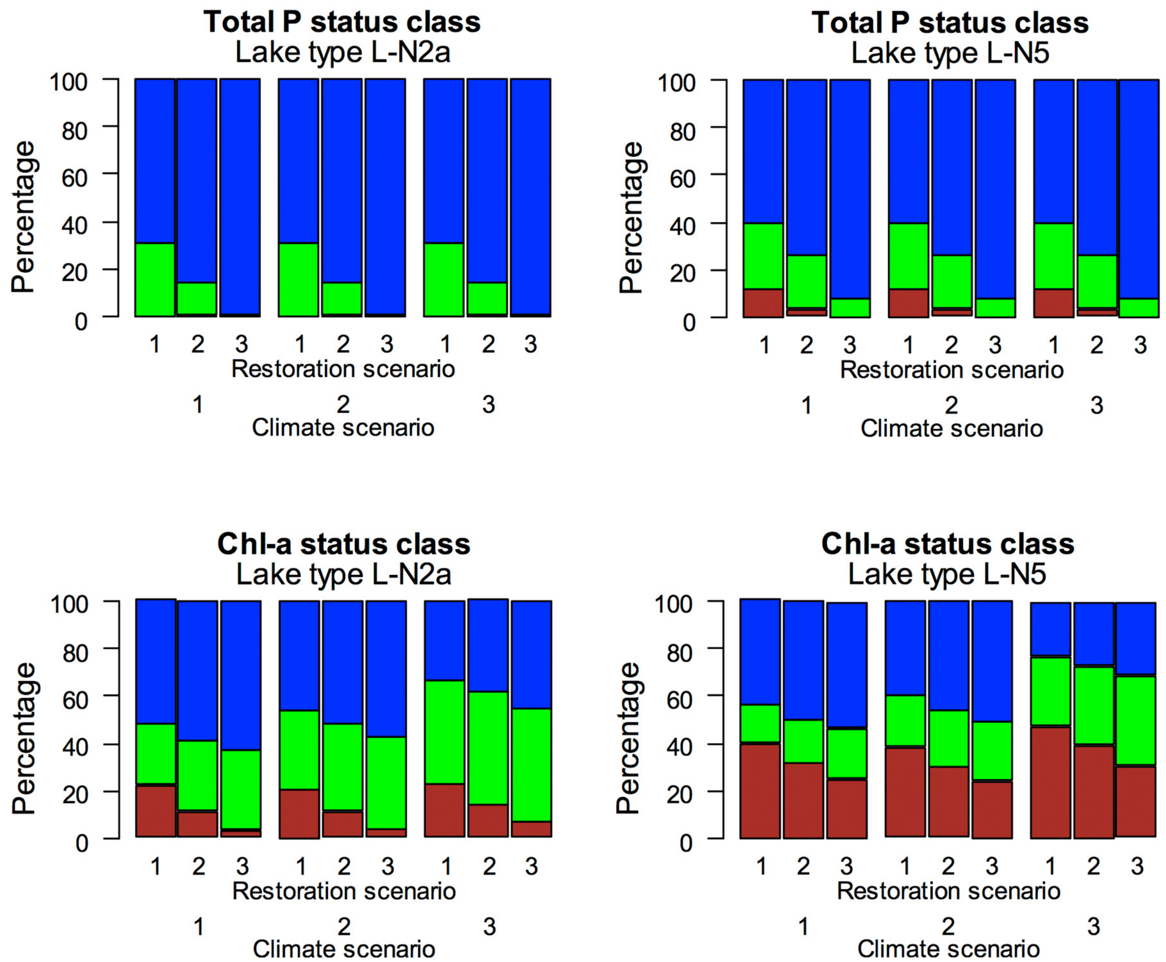

Figure 3. Probability distribution of status classes for Total P (upper panel) and Chlorophyll-a (lower panel), for low-altitude (left) and high-altitude (right) lakes respectively. Each plot shows the outcome of the 3x3 scenarios for Restoration and Climate change levels.

restoration level resulted in 30 percentage point (pp) increase in probability of High status for lake types L-N2a, and 32 pp increase for L-N5. TP was not affected by the climate scenarios, in accordance with the model settings. Chl-a status class also responded to reduction of P loading by increase in probability of High status, although to a lesser degree than for TP (L-N2a: 10-12 pp increase; L-N5: 8-11 pp increase). At the same time, the risk of less-than-good status was reduced by 14-19 pp. Chl-a was also impacted by climate change: +4 degrees resulted in an 18-20 pp reduction of High status probability for L-N2a, and 22-23 pp reduction for L-N5. The increase in risk of less-than-good status, however, was only 1-7 pp in this model.

(A)

Temperatur (water)

Climate cha...	No change		+2 degrees		+4 degrees	
Altitude	0 - 200	200 - 800	0 - 200	200 - 800	0 - 200	200 - 800
0 - 12	0.209	0.566	0.066	0.242	0.011	0.121
12 - 14	0.527	0.434	0.099	0.242	0.033	0.091
14 - 16	0.264	0	0.527	0.515	0.165	0.242
16 - 30	0	0	0.308	0	0.791	0.545

(B)

Status class Chla

Lake type	L-N2a						L-N5					
Chl a	0 - 3	3 - 4	4 - 5	5 - 7	7 - 10	10 - 20	0 - 3	3 - 4	4 - 5	5 - 7	7 - 10	10 - 20
H	1	1	0	0	0	0	1	0	0	0	0	0
G	0	0	1	1	0	0	0	1	1	0	0	0
MPB	0	0	0	0	1	1	0	0	0	1	1	1

Table 1. Examples of conditional probability tables (CPT). (A) The conditional probability distribution of the node "Temperature (water)" (°C) depends the levels of both "Climate change" and "Altitude" (m). (B) The conditional probability distribution of "Status class Chla" (H=High; G=Good; MPB=moderate, poor or bad) depends on both Lake type and Chl a (µg/l).

Concluding remarks

This study has focused on high status of lakes, because the source dataset was dominated by good- and high-status lakes. According to the phytoplankton indicator in this BN model, a climate change of +2 °C would almost counteract the benefit from 20% P loading reduction, while +4 °C would more than outweigh the benefits from 40% P loading reduction. The risk of not meeting good ecological status was less affected in this study. Although "good ecological status" is the main WFD management goal, preservation of high status is also a WFD requirement.

As mentioned, this modelling exercise considers only climate change impacts on lake processes, notably phytoplankton growth rate. In reality one can expect additional impacts on river basins such as increased P loading. This study has therefore only explored one the many potential climate change impacts on ecological status of lakes.

The Bayesian network modelling approach presented here is very general, and can easily be extended to include more lake types and other biological quality elements, as well as different scenarios. Based on the work presented here, the BN methodology has also been adopted by the ongoing EU project REFRESH, as a common approach for linking ecological responses to physico-chemical pressures for all water categories and all biological groups.

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Intertidal seagrass (*Z. noltii*) along anthropogenic pressure gradients – degradation and recovery trajectories from the Mondego estuary, Portugal

Joao M. Neto, Dimitri V. Barroso, Pablo Barría, Joao Carlos Marques

IMAR - Institute of Marine Research (C.M.A.). Department of Life Sciences, University of Coimbra, Largo Marquês Pombal, 3004 - 517 Coimbra, Portugal

Email address of corresponding author: jneto@ci.uc.pt

Key words: seagrass metrics, degradation trajectory, recovery trajectory, ecological assessment, pressure response

Introduction

The WISER project aims to support the implementation of the Water Framework Directive (WFD – Directive 2000/60/EC; European Council 2000), namely by testing the efficiency of assessment tools and / or helping to improve existing tools for the assessment of the ecological status of European surface waters based on the different biological quality elements. The WISER competence is not completed with the tools validation, moreover, it is expected to contribute for a better understanding of ecological processes influencing environmental quality changes. Are here included not only degradation but also the recovery trajectories, which are often moving through considerably different lines.

Although the study of the different paths followed when degradation or recovery are taking place doesn't constitute a completely new scientific issue (references), it's always important to confirm or to add new data into the general understanding of the topic. It's also known that every single ecosystem constitutes a particular case, where the differences observed in the distribution and in the interrelation existing between species and the abundance of their individuals (Franco et al., 2011), contribute to increase uncertainty about the general applicability of previous results and conclusions. The confirmation that degradation and recovery trajectories are not coincident it's relevant but it's needed to validate that the two processes are developing under the comparable environmental conditions (e.g., same pressure level).

From the above mentioned, the present work aims:

– to analyse the response of the intertidal seagrass *Zostera noltii* metrics against anthropogenic pressure;
– to compare the degradation and recovery trajectories followed up and down on the quality scale by this biological quality element;
– to validate the Seagrass Quality Index (SQI), developed during the WISER project, as a WFD compliant assessment tool.

Materials and methods

Study site

The study area is a southern Europe Atlantic estuary located at the western coast of Portugal (Figure 1). The Mondego estuary (40º08'N, 8º50'W) is a shallow Transitional Water (TW) classified as a mesotidal well-mixed estuary, with irregular river discharges and included in the Portuguese A2 type (Bettencourt et al., 2004), and as NEA 11 in the WFD (2000/60/EC). The southern canal of the estuary, where seagrass meadows can be found, constitutes a subsystem with 7 km length, 0.5 km width, 2 to 4 m depth and 2.57 km^2 in area. The marine influence is strong, and the average tidal amplitude of 1 to 3 m allows up to 75 % of this subsystem to be air exposed during low tide. (Neto et al., 2010)

Due to its regional economic value, the Mondego basin has been subjected to several physical modifications over the years (Neto et al., 2010). In this sense, the estuary has been continuously receiving high nutrient loads from the Mondego River catchment area, particularly those caused by the direct runoff from the 15,000 ha of cultivated land in the lower river valley (Neto et al., 2008). The estuary supports industrial activities, salt works, mercantile and fishing harbours, as well as the urban pressures from Figueira da Foz, a centre of seasonal tourism activity

Environmental evolution

Two distinct time intervals could be observed throughout the study period, 1986 to 2009. A first period ends in 1997 and is characterised by a general degradation process occurring in the south canal of the Mondego. A

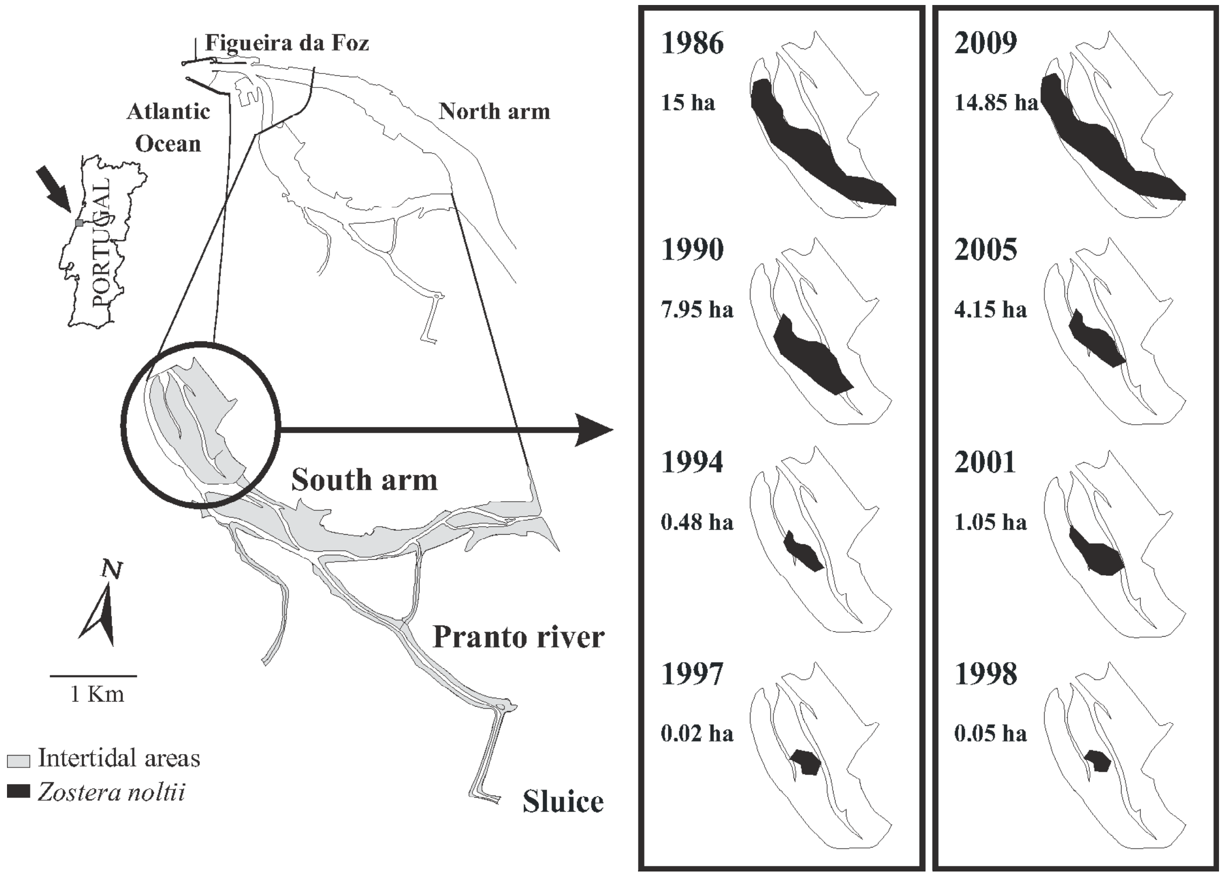

Figure 1. The Mondego estuary. Sampling area in the south arm (circle) and the Zostera noltii bed extent along the study period.

second period, from 1998 until 2009, is characterised by the implementation of several mitigation measures that resulted in the beginning of the ecological recovery of the south canal.

The last years of the first period were characterized by an intense anthropogenic disturbance (e.g., margins' regularization in 1990 and 1992, in the north arm), which culminated in a complete interruption of the communication between north and south arms of the river. Eutrophication symptoms were then visible in the south arm, mainly as proliferations of opportunistic green macroalgae (Martins et al. 2001, Marques et al. 2003) and the reduction of the seagrass cover area, possibly due to the decrease in dissolved oxygen and the increase in nitrite and ammonia concentration.

The second period started with the implementation of experimental mitigation measures (1997 and 1998) into the south arm. The communication between the two canals was re-established through a 1 m^2 section which allowed the water to periodically flow from 1.5 to 2 hours before and after each high tide peak (Neto et al., 2010). After the positive results obtained with the experimental reestablishment of the upstream communication between the two estuarine arms, this connection was widened in 2006.

Pressure data

Following the proposal of Aubry and Elliott (2006), three categories of indicators were considered to assess the anthropogenic pressures in the sampling site: a) hydromorphological changes (represented by the 'land claim' and the 'shore line re-enforcement'); b) resource use change (represented by the 'maintenance dredging area and volume', 'maintenance disposal area and volume', 'other fisheries near shore disturbance', 'marina development' and 'tourism and recreation'; and c) environmental quality and its perception (represented by 'nutrients concentration' and 'natural turbidity'). The selected pressure indicators (Table 1) were the ones considered as potentially significant on influencing the quality of the seagrass meadows.

Biological data

A long-term data series from the Mondego estuary (1986 to 2009) was used to provide information on the basic structural parameters 'bed extent', 'biomass' and 'shoot density' of *Zostera noltii* meadows. Sampling was performed at the intertidal area of the south arm of the Mondego estuary, during low tide and using a manual

Table 1: Categories, indicators and criteria used to assess anthropogenic pressures in the Mondego.

Pressure			Scores					
Category	Indicator	Criteria	No change (0)	Very low (1)	Low (3)	Medium (5)	High (7)	Very high (9)
Hydromorphological changes	Land claim (ha)	Consider both: mudflats and tidal marshes. This indicator includes both anthropogenically induced changes (land claim) and natural variations, since the 1900-1950s or before big morphological changes occurred (since when trustful maps are available).	No change	<0.5 % lost	<1% lost	<5% lost	<10%lost	≥ 10% lost
	Shoreline re-enforcement (%)	Percentage of the shoreline or estuarine margin that suffered re-enforcement work.	No change	<5%	<30%	<60%	<90%	≥ 90%
Resource use change	Maintenance dredging area (ha)	The annually subtidal dredged area in relation to total area of estuaries (or WB).	No dredging	<1%	<10%	<30%	<50%	≥ 50%
	Maintenance dredging volume (tons)	The amount of material dredged annually from estuaries (1 m3 of sand dredged is equivalent to 2 tons).	No dredging	< 5000 tons	<100,000 tons	< 1 million tons	< 4 million tons	≥ 4 million tons
	Maintenance disposal area (ha)	The area designated for disposal in estuaries (or WB) or length affected by disposal (for tidal rivers) as suggested within the Water Framework Directive for the designation of Heavily Modified Water Bodies (HMWB).	No disposal	<1%	<10%	<30%	<50%	≥ 50%
	Maintenance disposal volume (tons)	Represented by the total tonnage annually disposed in estuaries.	No disposal	< 5000 tons	<100,000 tons	< 1 million tons	< 4 million tons	≥ 4 million tons
	Other fisheries nearshore disturbance	Percentage of the length of coast or estuarine (or WB) area affected by fishery.	No fishery activities	< 10%	<30%	<60%	<90%	≥ 90%
	Marina Development	The intensity of marina development is measured by the number berths / km2 of the WB.	No marina	< 100 berths / km2 WB	<150 berths / km2 WB	<300 berths / km2 WB	<500 berths / km2 WB	≥ 500 berths / km2 WB
	Tourism and recreation	Percentage of the length of coast (riverbank) or estuarine (or WB) area affected by tourism and recreation activity.	None	< 10%	<30%	<60%	<90%	≥ 90%
Environmental quality and its perception	Nutrients (μmol/L)	Quantified as the DIN winter median concentration (μmol/L)	< 6.5	< 10	< 30	< 60	< 90	≥ 90
	Natural turbidity	Measured as the mean secchi disk transparency (m) during growing season (May to September).	< 0.5	< 1	< 1.5	< 2	< 2.5	≥ 2.5

corer (13.5 cm Ø). Samples were randomly collected inside the *Zostera* meadow to provide data on biomass and shoot density. The bed extent mapping was based on field observations (GPS to register the meadows perimeter), vertical photographs and GIS methodology (ArcView GIS version 8.3). Depending on the purpose data were collected from twice a month during several year to a lower frequency of only one to three sampling events concentrated in the growing season. Samples were sorted in the laboratory, the shoots counted and the biomass determined as dry weight (g DW after weight stabilisation at 70 ºC).

Metrics and quality assessment method (SQI)

The Seagrass Quality Index (SQI) includes three different metrics: 1) species richness, as the number of taxa, 2) the bed extent, as the areal cover of the meadows, and 3) the shoots density, as the number of shoots per m^2.

The deviation from the reference condition is calculated for each metric, converted in a scale 0 – 1. After this first round of calculations, the EQR is obtained through the use of the combination rule expressed in equation 1:

$$EQR = (T/5)*0.2 + BE*0.3 + SD*0.5$$

where T is the no. of taxa, BE is the bed extent / bed extent reference condition, and the SD is the shoot density / shoot density reference condition.

An equidistant scale translates the EQR obtained into the EQS classes.

Data analysis

Data on the structural parameters of the seagrass were analysed along the study period. The response of the bed extent and biomass structural parameters to the different levels of pressure was also analysed, both towards degradation and after the implementation of the first (experimental) mitigation measures.

The response of the SQI method, was also tested against the different pressure levels. The ability of the SQI in reporting into the five ecological quality classes (bad, poor, moderate, good and high) (WFD, 2000/60/EC) was also examined and compared to the pressure level acting at the moment.

The correlation between biological data (metrics and the SQI EQRs) and the anthropogenic pressures (total pressure and the sum of resources change plus environmental quality) was tested through the Pearson product moment correlation coefficient, with StatSoft, Inc. (2004) STATISTICA (data analysis software system), version 7.

Results

The response of the structural parameters (bed extent and biomass) was in agreement with the pressure level (Figure 2), but through different trajectories when under degradation or recovery.

The environmental quality, expressed by the SQI, reacted as expected in agreement with the different pressure levels. Although the reduction registered after 1997 on the pressure level, the recovery was slower than degradation and, apparently, some initial inertia was needed to combat so the positive results could be detected.

The SQI is, apparently, on a good position to pass the validation test. The response of the method against the pressure levels was adequate and it was also able to report adequately into the five used quality classes (high, good, moderate, poor, bad).

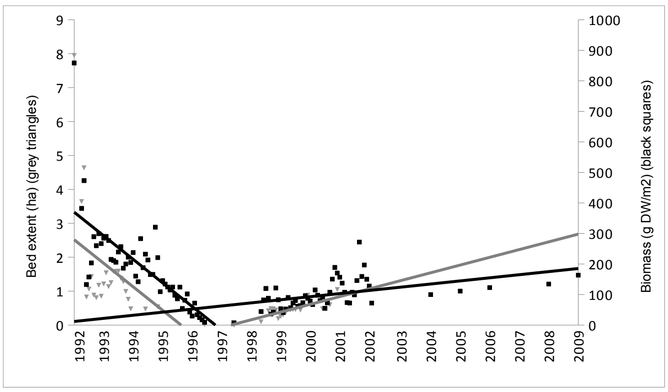

Figure 2: Structural parameters (bed extent, biomass) along the study period. Towards degradation and after implementation of mitigation measures.

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Experiences from the intercalibration exercise of fish-based assessment of ecological status for northern lakes

Mikko Olin[1], Kerstin Holmgren[2], Trygve Hesthagen[3], Fiona Kelly[4], Martti Rask[5]

[1] University of Helsinki, Department of Environmental Sciences, Aquatic Sciences, P.O.Box 65, FI00014 Helsinki, mikko.olin@helsinki.fi
[2] Swedish University of Agricultural Sciences, Department of Aquatic Resourses, Institute of Freshwater Research, Stångholmsvägen 2, SE-178 93 Drottningholm, Sweden
[3] Norwegian Institute for Nature Research, PO Box 5685, N-7485 Trondheim, Norway
[4] Inland Fisheries Ireland, Swords Business Campus, Swords, Co. Dublin, Ireland
[5] Finnish Game and Fisheries Research Institute, Evo Game and Fisheries Research, FIN-16970 Evo, Finland

Email address of corresponding author: mikko.olin@helsinki.fi

Key words: fish-based assessment, ecological status, Water Framework Directive, intercalibration

Background

Intercalibration (IC) of assessment tools developed for each biological quality element is required according to the Water Framework Directive (WFD). In the second phase of the IC-work, the national lake fish assessment tools from four northern GIG countries (Finland, Ireland, Norway and Sweden) were intercalibrated. Only the Finnish and Irish methods passed the feasibility check, showed a sufficient pressure response, and met the comparability criteria of the IC guidance. A summary of the main characteristics of the national methods, the steps and output of the intercalibration exercise, and some problems encountered during the process are presented.

National methods

The Finnish EQR4 method mainly targets eutrophication pressure. It is calculated as the average of three or four of the variables (Table 1). Reference values and class boundaries are calculated from type-specific reference lakes (n=10-32) of 10 out of 12 national lake types that are mainly based on water colour, mean depth, and surface area of the lakes (the method does not yet cover the lake types "High altitude lakes" and "Lakes with low retention time"). The ecological classification by the metric "indicator species" is defined as expert judgement based on the presence, absence/extinction of certain sensitive species like whitefish (*Coregonus lavaretus*) and burbot (*Lota lota*). For details, see LNF_Milestone_6, Rask et al. (2010).

The Irish method FIL2 targets eutrophication and general land use pressures. Of the 13 variables (Table 1), 5-6 variables are used in each of the four lake types which are based on water alkalinity and mean depth. Ecological status is determined by using discriminant classification rules and a generalised linear model. For details, see LNF_Milestone_6, Kelly et al. (submitted for publication).

The Norwegian FCI index requires data on species occurrence and evenness; both historic and current data, and species status (losses / changes in abundance) for different species categories. Therefore, data are applied from multiple sources (Table 1). FCI mainly targets acidification pressure. The reference condition is defined as an unchanged and healthy population, and is site specific. For each fish community, this value is obtained by grouping the species into three categories; dominant, subdominant, and rare. Secondly, population status in terms of changes in abundance relative to the reference condition is assessed by grouping the species into three categories; unchanged/no damage, marked change (either increased or decreased), or lost. The fish community index FCI is then defined as the relative deviation from the reference condition. For details, see LNF_Milestone_6, and Anonymous 2009.

The Swedish EQR8 is the most ambitious of the four national methods, targeting acidification, eutrophication and general ecological degradation. The mean of three to eight of the variables (Table 1) is used for classification. Reference values are obtained by modeling according to lake altitude, area, maximum depth, annual mean air temperature, location below or above the highest coast line after deglaciation. The metric values are expressed as standard residuals from lake-specific reference values (Z-values), transformed to P-values. For details, see LNF_Milestone_6, Holmgren et al. (2007).

Table 1: The four national lake fish assessment methods in NGIG.

Country / method / pressure	Variable	Data source
Finland / EQR4 / eutrophication	*BPUE: total biomass per unit effort* *NPUE: total number per unit effort* *CYPRINIDS %: biomass share of cyprinids** *INDICATOR SPECIES: presence/absence*	*Std gillnetting* *Std gillnetting* *Std gillnetting* *All available*
Ireland / FIL2 / eutrophication, general land use	*TOT_BPUE: sum of mean BPUE*[#] *NAT_BPUE: sum of mean BPUE of native species* *PERCH_BIO: mean perch BPUE* *RHEO_BIO: % rheophilic individuals*[#] *SPE_EVEN: Species evenness/dominance*[£] *ROACH_BPUE: mean roach BPUE* *BREAM_%_IND: % composition of bream*[$] *PHYT_%_BIO: % phytophilic individuals*[#] *2_%_BIO: % biomass of non-native species*[¤] *CYP_BIO: % biomass of cyprinid species*[#] *RUDD_%_IND: % composition of rudd*[&] *MAX_L_DOM_BIO: Max. length of dominant species*[#] *LITH_IND:% lithophilic individuals*[˜]	*Std gillnetting and fyke netting, per linear metre of net used*
Norway / FCI / acidification	*FCI index: historical and present occurrence, evenness, and status (losses / changes in abundance) of fish species*	*Interviews, reports, test-fishing, water chemistry, modeling*
Sweden / EQR8 / acidification, eutrophication, general degradation	*Number of native species* *Simpson's D (abundance)* *Simpson's D (biomass)* *Relative biomass of native species (BPUE)* *Relative abundance of native species (NPUE)* *Mean mass of native species* *Piscivorous percids biomass %*[§] *Perch / Cyprinids biomass ratio*	*Std gillnetting*

**incl. species that favour eutrophic conditions (roach, bleak, rudd, bream, white bream, blue bream, crucian carp, tench). The variable is not included in the EQR4 if the fore mentioned species are not present in fish fauna.*

[#] *based on BPUE excl. eels and adult salmon*

[£] *(1/D=1/(Nmax/Ntot) (Nmax= no. inds represented by the most abundant species, Ntot=total number of individuals in the sample (eels captured in fyke nets excluded) (Based on total number of fish captured)*

[$] *based on CPUE (BREAM_CPUE/TOTAL_CPUE*100)*

[¤] *Species group 2 (non-natives influencing ecology): roach, perch, pike, bream, dace, carp, rainbow trout, chub, minnow*

[&] *based on CPUE (RUDD_CPUE/TOTAL_CPUE*100)*

[˜] *excl. eels and adult salmon*

[§] *The proportion of potentially piscivorous perch is 0 at fish length less than 120 mm and 1 at length above 180 mm. At intermediate length the proportion is calculated as 1 – ((180 – length) / 60). Individual mass of perch (g) is estimated as a * length (mm)^b, where a = 3.377 * 10^{-6}, and b = 3.205. Each individual mass is multiplied with the length-specific proportion piscivorous perch. The sum of the products is the biomass of piscivorous perch, which is then added to any biomass of pikeperch. Finally, the total sum of piscivorous percids is divided by the total biomass of all species in the catch.*

Lake data

The IC dataset of the northern GIG group was based on data delivered to the cross-GIG database since 2009; a total number of 1577 lakes from northern GIG countries. As all participating countries use standard gillnet sampling procedure (EN 14757), the comparability of the data was considered to be sufficient. After a pilot study using a set of 640 lakes (Holmgren et al. 2010), it was clear that a successful IC exercise would not be possible without a more detailed determination of IC common lake type and without selecting common pressures for all participating countries. Therefore, we finally ended up with lakes corresponding to the Finnish lake types 1 (mean depth > 3 m, oligohumic, < 40 km^2 in area) and 2 (mean depth > 3 m, humic, < 5 km^2), totalling 169 non-acidified and non-limed lakes, and eutrophication w as considered to be the most important common pressure. Out of the 169 lakes, 106 were reference lakes, that fulfill the reference criteria of the cross-GIG lake fish IC group (Causse et al. 2011), and 63 lakes were impacted by eutrophication.

Ecological classification of the common data set by the four national methods

Of the four national methods, the Swedish method was found to be the strictest as 80% of the impacted lakes were assigned less than good status (Table 2). The

Table 2: Ecological classification of the common data set (169 lakes) by the Finnish (EQR4), Irish (FIL2), Swedish (EQR8) and Norwegian (FCI) methods. Error % = the risk of misclassification of reference lakes (R) to a status worse than good or impacted lakes (I) to status good or high.

	EQR4	EQR4	FIL2	FIL2	EQR8	EQR8	FCI	FCI
%	R	I	R	I	R	I	R	I
High	*55.2*	*12.4*	*37.8*	*19.2*	*2.8*	*0.6*	*50.0*	*26.7*
Good	*28.7*	*33.3*	*25.2*	*23.7*	*40.6*	*19.2*	*25.0*	*40.0*
Moderate	*13.3*	*32.2*	*9.8*	*10.2*	*19.6*	*23.2*	*20.0*	*33.3*
Poor	*2.8*	*17.5*	*0.0*	*0.0*	*30.8*	*24.9*	*0.0*	*0.0*
Bad	*0.0*	*4.5*	*0.0*	*0.0*	*6.3*	*32.2*	*0.0*	*0.0*
Poor/Bad	*2.8*	*22.0*	*27.3*	*46.9*	*37.1*	*57.1*	*5.0*	*0.0*
Total	*100.0*	*100.0*	*100.0*	*100.0*	*100.0*	*100.0*	*100.0*	*100.0*
Error %	*16.1*	*45.8*	*37.1*	*42.9*	*56.6*	*19.8*	*25.0*	*66.7*

Norwegian method was the least strict as the corresponding proportion was only 33%. The Finnish and Irish methods classified the impacted lakes rather similarly, the percentage was 54% and 57%, respectively.

The risk of misclassification of reference lakes to a status worse than good was lowest in the Finnish EQR4 (16 %, Table 2) and highest in the Swedish EQR8 (57 %). The risk of misclassification of impacted lakes to good or high status was lowest in the Swedish EQR8 (20 %) and highest in the Norwegian FCI (67 %).

Intercalibration exercise

All four national classification methods were considered compliant with the WFD requirements but some remarks are worth mentioning. Age structure of fish was included indirectly, mainly through length frequency distribution of fish, by each tool except the FCI. IC common lake types were not applied but national lake types. Lake type specific reference conditions were used in EQR4 and FIL2. In EQR8, lake specific reference values were modeled, and in FCI estimated for each site.

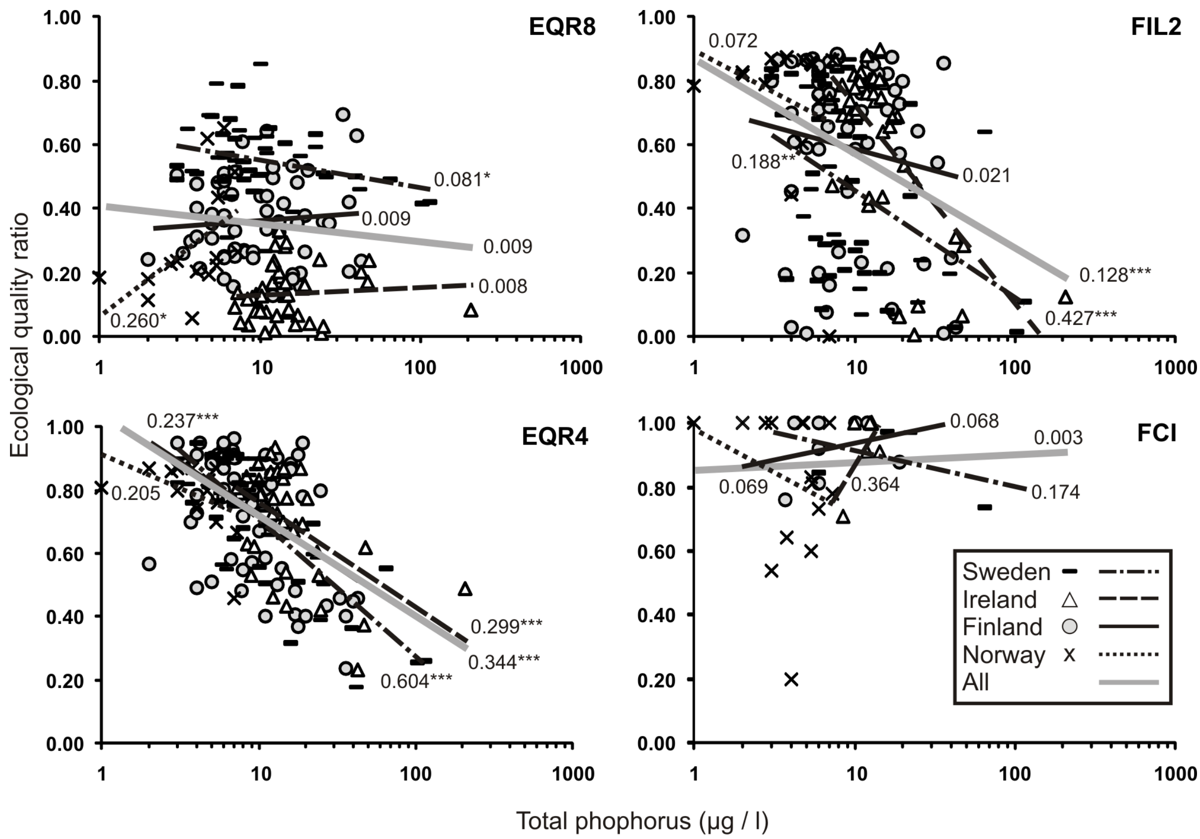

*Figure 1: EQR values of the four national methods (Finnish EQR4, Irish FIL2, Norwegian FCI and Swedish EQR8) in relation to total phosphorus concentration in the lakes of common IC type (n=169, except for FCI n=35). Coefficients of correlation and P-values (<0.05 = *, <0.01 = ** and <0.001 = ***) of the regression analyses for country-specific and total lake data are shown.*

Intercalibration was considered feasible in terms of lake typology, pressures and assessment concept between the Finnish, Irish and Swedish methods. The Norwegian method was excluded from IC due to targeting different pressure (acidification), and problems in obtaining fish data with sufficient quality from multispecies communities.

The pressure response of the methods was examined by using total phosphorus (TP) concentration as the indicator of eutrophication. Only the Irish and the Finnish methods responded in a meaningful way and showed statistically significant TP-correlations with fairly similar slopes (Fig. 1). The Norwegian method did not respond to the eutrophication pressure. The Swedish method displayed a weak response for Swedish lakes, but no response when using all data (Fig. 1). We also checked the correlation of EQR4, EQR8, and FIL2 classification output (benchmark standardized values by subtraction, Birk et al. 2011) to a pseudo common metric (PCM), i.e. to the mean of the output from the other indices (Fig. 2). As the Swedish EQR8 did not correlate with the PCM (whereas EQR4 and FIL2 had the correlation), it was not possible to intercalibrate the Swedish method.

Thus, only the Finnish and Irish assessment method could be intercalibrated. For calculations the IC option 3a (direct comparison of two methods, Anonymous 2010) was applied. The two methods met the comparability criteria required for intercalibration (Fig. 3), including acceptable correlation (slope between 0.5-1.0, r>0.5 and p<0.01) and class agreement (difference <1.0 classes). Only the Good/Moderate boundary of the Irish FIL2 had a slightly too higher than acceptable boundary bias, and the boundary was changed from 0.53 to 0.51. This small adjustment had practically no ecological significance.

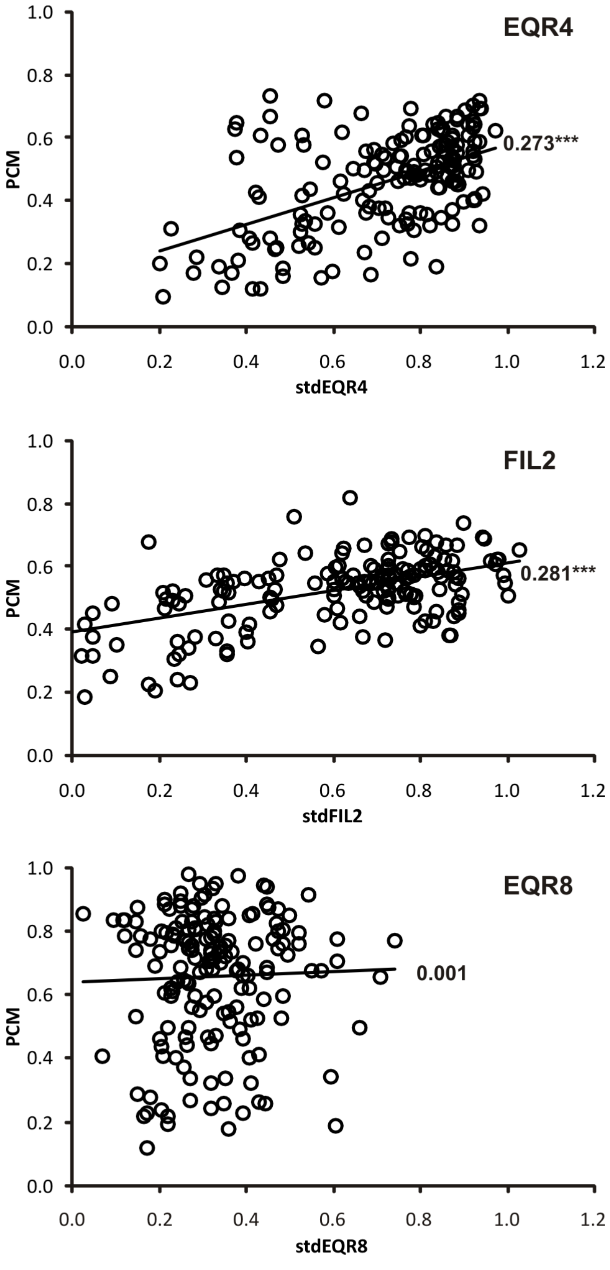

Figure 2: The relation of classification results from Finnish, Irish, and Swedish assessment methods to the pseudo common metric (PCM = average classification of the other methods).

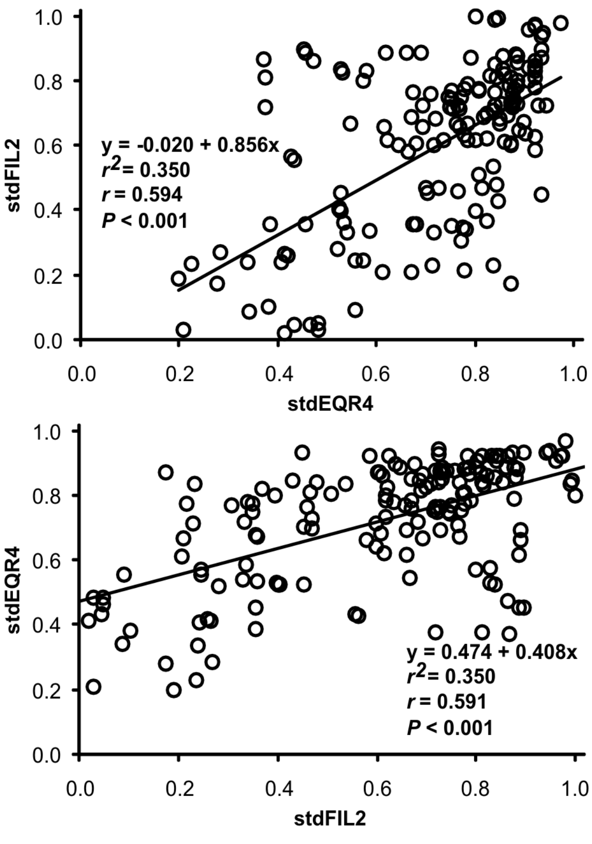

Figure 3: Direct comparison of benchmark standardized EQR values of EQR4 and FIL2 methods applied to the the common IC type data.

Discussion

Generally, the fish based classification of the N-GIG lakes in a comparable way has been a challenging task. The original fish fauna of Ireland and Norway is much poorer in species number compared to Finland and Sweden. Further, the main pressure differs between countries, being eutrophication in Finland and Ireland, acidification in Norway and either acidification or eutrophication in Sweden.

The alarming differences in the output of the Finnish and Swedish methods were first recorded during the TRIWA II Interreg project where 14 pristine lakes of Torne River Basin (8 from Finland, 6 from Sweden) were classified with the both methods (Sairanen et al. 2008). The ecological status of the lakes using the Finnish EQR4 method was 1 to 2 classes higher than the Swedish EQR8 classifications. Classification results for the same lakes obtained from benthic macroinvertebrates, phytoplankton, and water chemistry (Elfvendahl et al. 2006) were close to the output of EQR4.

Several potential reasons for the different results can be suggested. In the Swedish EQR8 all variables are two-tailed and therefore sensitive also to low values of test fishing catches in order to detect the effects of acidification. In the Finnish EQR4 only the variables total biomass and number of individuals are two-tailed. There are also essential differences in the reference lake material of the countries: the Swedish lake dataset consists of more oligotrophic and acid sensitive highland lakes whereas the Finnish reference lake set is dominated by more lowland productive lakes. As a result, EQR4 can be thought to be more reliable for classification of eutrophicated lakes whereas EQR8 may be better for acidified lakes. Furthermore, as EQR8 is targeting two almost opposite pressures acidification and eutrophication, the variables which are sensitive mainly to acidification (e.g. number of species and diversity indexes) may reduce the ability to detect eutrophication pressure.

In conclusion, the good comparability between the Finnish and Irish methods is promising when taking into account the differences of the original fish fauna of the two countries and their location in different ecoregions, as well as methodological differences in the variables and boundary setting. Respectively, it should not be impossible to adjust the methods of Finland and Sweden to respond in similar ways to similar pressures in the boreal lakes of ecoregion 22 including also lakes in south eastern Norway.

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Benthic macrophyte changes across an anthropogenic pressures gradient in Mediterranean and Black Sea water systems: structural vs. functional approaches

S. Orfanidis[1], K. Dencheva[2], K. Nakou[1] & A. Basset[3]

[1] National Agricultural Research Foundation, Fisheries Research Institute, 640 07 Nea Peramos, Kavala, Greece (sorfanid@inale.gr)
[2] Institute of Oceanology, Bulgarian Academy of Sciences, PO Box 152, 9000 Varna, Bulgaria (dencheva@io-bas.bg)
[3] Di.S.Te.B.A., Centro Ecotekne, Prov. Lecce-Monteroni 7, University of Lecce, 31 00 Lecce, Italy (alberto.basset@unile.it)

Email address of corresponding author: sorfanid@inale.gr

Introduction

Biotic metrics of benthic macrophytes represent an effort to describe different and complex aspects of communities or other different biological organizational levels by integrating them in a formula producing a single numerical output (Orfanidis et al., 2011). This approach may distinguish responses of human impact from natural variability when supported by quantitative data enabling to study communities' heterogeneity (see Orfanidis et al., 2008) and to identify correlations and casual relationships between the biotic and abiotic data.

The aim of this paper was to test the relationship among different structural and functional metrics with key abiotic parameters and total pressures in Mediterranean (Lesina Lagoon, Italy) and Black Sea (Varna Lake and Bay, Bulgaria) water systems. A verification of metrics responses to natural or anthropogenic ecological processes at an international scale across different water typologies is intended.

Material & Methods

Sampling and data analysis efforts

Sampling in Lesina Lagoon (Figure 1a) was undertaken between 21st and 23rd September 2009. A 0.0225 m^2 Ekman grab was used to collect twelve random samples in each site at 0.6 to 1.2 m depth. 84 samples were selected in total. In the laboratory the samples were sorted out and the species were identified to functional group level and as much as possible to species level. In order to estimate % coverage a transparent double bottom square PVC container, filled with sea water and having at its bottom a square 15x15 cm matrix divided in 100 squares, was used. The surface covered by each sorted taxon in vertical projection floating in sea water was quantified as % of coverage.

Sampling in Varna Lake and Bay (Figure 1b) was undertaken between 8th and 10th September 2009. At each site different number (from 7 to 42) square frame (0.01 m^2) samples were collected. Samples were taken from 0-2 m depth with the help of diving technique. 122 samples were selected in total. Visual assessment of

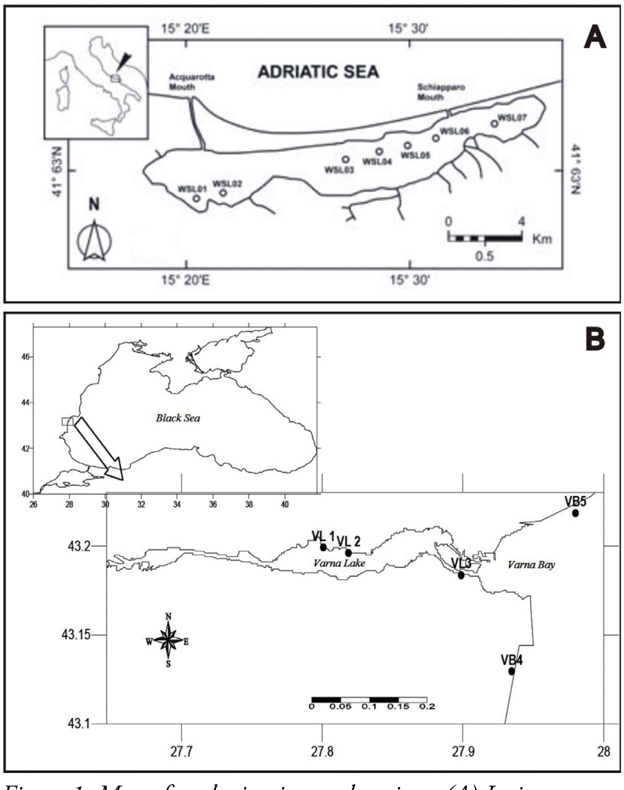

Figure 1: Map of study sites in two locations: (A) Lesina lagoon, (B) Bulgaria-Varna Lake and Bay.

total percent cover of the communities of every depth layer was carried out. In laboratory conditions all benthic macrophyte samples were washed and sieved to remove sediments. Macrophytes were sorted and identified to the lowest possible taxonomic level under microscope when needed. Species were dried for a while on a filter paper and weighted (fresh weight).

Metrics and methods calculation

Nine metrics related to community structure [species number, Shannon-Weaver index (H', $\log_2$), Pielou's evenness index (J'), % of total coverage, and dry biomass (g/m^{-2})] and function (ESG I % coverage, ESG II % coverage, EI and EEI) were estimated. Fresh weight biomass data from Bulgarian coasts were multiplied by factor 0.09 to be transformed to dry biomass. The abundance of the two Ecological State groups (ESG I, ESG II) and the Ecological Evaluation Index (EEI-c) for each site were calculated according to Orfanidis et al. (2011). The calculation of EEI-c in site 7 was modified by introducing a new group (ESG IIC) that includes species of fresh water affinity such as *Potamogeton* sp. This species is valued similarly to opportunistic species (ESG IIB) since its existence in the site 7 is explained by low salinity (close to 10 PSU) that prohibits their growth. A modification of EEI-c index in conformity with Black Sea peculiarities was also developed. Index values are represented as biomass percent ratio of late-successional (sensitive species) divided by biomass of sensitive and opportunistic-tolerant species. Total cover value of macrophyte communities from every site was multiplied by percent biomass values to obtain final EI, presented as continuous numerical values from 0 to 10. For example, for 80% sensitive species biomass EI equals to 8 and for 65% biomass to equals to 6.5. EQR value has been calculated as current obtained EI value divided by referent value (10). EQR values for different Ecological Status Classes are following: (0-0.2= bad; 0.2-0.4=poor; 0.4-0.6=moderate, 0.6-0.8=good and 0.8-1=high ESC.

Statistical treatment

Only samples with a % coverage >10% were analyzed. The response of metrics and assessment method to the pressure gradient was evaluated using Spearman rank correlation coefficients (ρ) at p=0.01 after a log (x+1) transformation of the data. PCA analyses were performed on log (x+1) transformed data.

Results & Discsussion

PCA components 1 and 2, which explains 86.2 % of the total variability (component 1=65.7%, component 2=20.5%), indicated that the sampled sites belong in a gradient rather than in distinct environments and water types, in terms of depth, salinity, grain size etc (Figure 2).

A strong correlation was found between the total pressures and the functional indices EI and EEI-c (Table 1). A strong non-linear relationship between the total pressures and the EI and EEI-c was also identified (Figure 3). These results indicate a different behavior between the structural and the functional indices as has been earlier documented by Orfanidis et al. (2008). Therefore, the functional indices are better indicating the pressures and thus the ecological status of water systems across different typologies, while the structural indices are better indicating the lagoon confinement.

Using the EEI methodology the studied sites were classified as: VB4 - "bad" ESC; VL1, VL2, VL3, WSL01,

Table 1: Spearman rank correlation coefficient between key abiotic and biotic metrics in Lesina lagoon (a) and Bulgarian coasts (b). Underlined values show significant correlation at $p<0.01$.

	Total coverage (%)	Total dry biomass (g/m^2)	Species No	J'	H'	ESG I%	ESG II%	EI	EEI-c
Depth (m)	*-0.18*	*0.16*	*0.63*	*0.41*	*0.48*	*0.31*	*-0.13*	*0.05*	*0.13*
Temperature (°C)	*0.10*	*-0.17*	*0.06*	*0.00*	*-0.04*	*-0.05*	*0.24*	*-0.13*	*-0.02*
Salinity (PSU)	*-0.18*	*-0.50*	*-0.15*	*-0.61*	*-0.59*	*-0.01*	*-0.02*	*0.28*	*0.36*
Oxygen saturation (%)	*0.14*	*0.38*	*0.04*	*-0.09*	*0.05*	*0.35*	*0.01*	*0.07*	*0.17*
Turbidity	*-0.12*	*0.21*	*0.63*	*0.42*	*0.50*	*0.32*	*-0.09*	*0.05*	*0.12*
Organic content (%)	*0.40*	*0.05*	*-0.75*	*-0.65*	*-0.71*	*0.46*	*-0.35*	*0.76*	*0.73*
Gravel (%)	*-0.12*	*-0.03*	*0.22*	*0.35*	*0.34*	*-0.31*	*0.47*	*-0.62*	*-0.49*
Sand (%)	*0.14*	*0.56*	*0.21*	*0.15*	*0.26*	*0.76*	*-0.33*	*0.37*	*0.56*
Mud (%)	*-0.12*	*-0.48*	*-0.32*	*-0.72*	*-0.70*	*-0.10*	*-0.22*	*0.42*	*0.29*
Total pressures	*-0.37*	*-0.35*	*0.32*	*0.46*	*0.40*	*-0.83*	*0.40*	*-0.90*	*-0.95*
Distance to the pressures (km)	*0.21*	*0.67*	*0.34*	*0.55*	*0.61*	*0.62*	*-0.33*	*0.18*	*0.29*

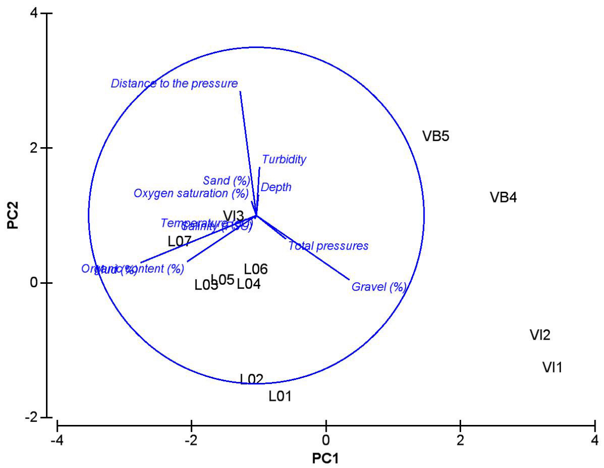

Figure 2: Principal Component Analysis (PCA) using the total pressure and the abiotic data together with the sampling sites of Lesina lagoon and Bulgarian coasts.

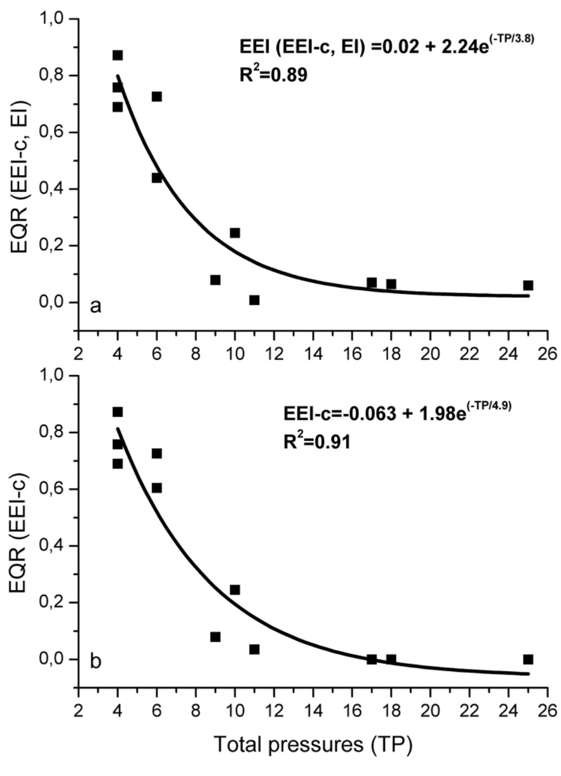

Figure 3: The relationship between a) EEI (EEI-c, EI), and b) EEI-c EQR's with total pressures across the sampling sites of two locations (Lesina lagoon, Varna lake and bay).

WSL02 - "low" ESC; VB5 - "moderate" ESC; WSL03, WSL04, WSL07 - "good" ESC; and WSL5 -"high" ESC. While the mean value of EEI-c index (0.56) classifies the Lesina Lagoon in "good" ESC, the mean value of EI (0.13) classifies the Varna Lake and Bay in "bad" ESC. Indeed the lagoon of Lesina seems to experience a low vulnerability to human activities, especially the central and eastern bases (Vignes et al., 2009). On the opposite Varna lake is very eutrophicated and polluted ecosystem and both biotic and abiotic parameters advocate for worse conditions (Dencheva, 2010). These contaminated waters enter the bay and the main current in south direction contributes to the deterioration in this part of the bay too.

The structural diversity indices, as in other lagoons (Middelboe et al., 1998), were in general low. A decrease in macrophytes diversity of from the entrance to the inner parts coastal lagoons suggests the existence of physiological stress due to strong salinity gradients or spore, fragment or propagule dispersal restriction (confinement) or interactions between them (Coutino and Seeliger, 1984).

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Estimating acceptable phosphorus and nitrogen loading levels with Lake Load Response (LLR) model tool

Anita Pätynen

Finnish Environment Institute, Modelling and Assesment Unit, Survontie 9, FI-40500 Jyväskylä, Finland
Department of Biological and Environmental Science, PL 35, FI-40014 University of Jyväskylä, Finland

Email address of corresponding author: anita.patynen@ymparisto.fi

Introduction

Poor lake water quality, determined as elevated phosphorus, nitrogen and chlorophyll a concentration, is usually an outcome of excessive nutrient loading. After the implementation of Water Framework Directive (WFD) all EU countries faced the requirement to cure these water quality problems (among with the ecological ones). Nowadays most of the external loading comes from diffuse sources, mainly from agriculture, and besides changes in land use practices, loading reduction often requires costly management actions. The basis for successful lake management is an idea of the tolerable loading to the lake, thus knowing what kind of loading reductions is needed to achieve the phosphorus, nitrogen and/or chlorophyll a concentration limits for good water quality. As it is time consuming and often impossible to monitor one lake to get an idea of the relation between loading and in-lake concentrations, some helpful equations and models are generated from larger lake data sets.

In practice the lack of sufficient input data, required expertise and sometimes expensive model or software licenses prevent the usage of very complex models, although they could perhaps provide more detailed and accurate description about the modeled processes. That is why there has been a need to develop simpler models to a stage where they are still easy to use, but do not give too general results, with no possibility to estimate their accuracy. One of those is the Lake Load Response (LLR) internet tool that has been developed within the Finnish Environment Institute. It is based on the LakeState (LS) model, that consists of three component models: Chapra's (1975) model for retention of total phosphorus and nitrogen, the hierarchical, linear regression model for chlorophyll a (Malve, 2007) and the logistic regression model for phytoplankton biomass (Kauppila P., Lepistö L., Malve O. & Raateland A., unpublished). In the LS model the mechanistic and statistic approach are combined using Bayesian inference with Markov Chain Monte Carlo simulation methods. This way, predictions about the water quality as well as about the model error can be made on a statistical basis, which gives more confidence into lake management planning.

The strength of LLR is particularly in the strong statistics and low data requirements that enable the use of LLR for less studied lakes. There are however, some noteworthy issues when using this kind of simple, "black box" model that takes in data and hands out a result. Here I point out some of those through estimates for chlorophyll a concentrations.

Materials and methods

LLR can be freely applied through Internet interface (lakestate.vyh.fi). Besides basic information about the lake (volume, mean depth and lake type that is used in hierarchical chlorophyll a model) the user needs to give a data chart with values for incoming phosphorus and nitrogen loading, in-lake phosphorus and nitrogen concentrations and outflow as averages for the lake's retention time. With this information LLR estimates the needed loading reduction (or as well, how much extra loading the lake tolerates) to achieve the lake type specific phosphorus and nitrogen concentration limit for good water quality, or to keep the phytoplankton biomass below the good water quality level. If reducing chlorophyll a concentration is the main interest, another data chart including chlorophyll a, total phosphorus and total nitrogen concentrations from the growing seasons is needed. For detailed information about LLR and its usage, see LLR web pages.

LLR gives estimates based on the data from the study lake alone (Lake Specific model), and as a comparison, utilizing regression drawn from larger lake data sets (Finnish Lakes or European and North American Lakes Model; in this study the Finnish Lakes model was tested besides the Lake Specific Model) As a default LLR uses 50% prediction probability in its estimations. This defines the level of reliability and tells the user how certain it is that the estimated loading reduction leads to desired end result. If not satisfied with 50-50 situation, the user can set the prediction probability higher and get more confidence to the estimates. However, more confidence means more reduction to the loading, but this way LLR provides an opportunity to weight between the costs and confidence and find the optimal combination.

I tested LLR for Lake Pyhäjärvi in SE Finland (volume 849 $10^6 m^3$, mean depth 5.47 m). The input data was from years 1980-2003, and with theoretical retention time of four years that averaged to six lines of data.

Results

The average phosphorus surface loading into the lake in 1980-2003 was ca. 0.1 g m^{-2} a^{-1} (42 kg/d) and average nitrogen surface loading ca. 2 g m^{-2} a^{-1} (890 kg/d). With the Lake Specific Model and 50% prediction probability this combination was estimated to lead to growing season chlorophyll a concentration of 7 mg l^{-1} (Fig. 1). As the type specific limit of good lake water quality for Lake Pyhäjärvi is 7 mg l^{-1}, there does not seem to be a need for loading reduction. With the Finnish Lakes Model the estimated chlorophyll a concentration was over the limit, 12 mg l^{-1}. According to monitoring data, the average chlorophyll a concentration in growing season during 1980-2003 was ca. 7 mg l^{-1} (HERTTA database of Finnish Environmental Administration).

Discussion

According to the Lake Specific Model, there does not seem to be problems with excessive phytoplankton growth in Lake Pyhäjärvi, but the chlorophyll a concentration is almost on the good/moderate class boundary. To some extent this is true, as the lake is classified mesotrophic. However, the biggest indisputable weakness in

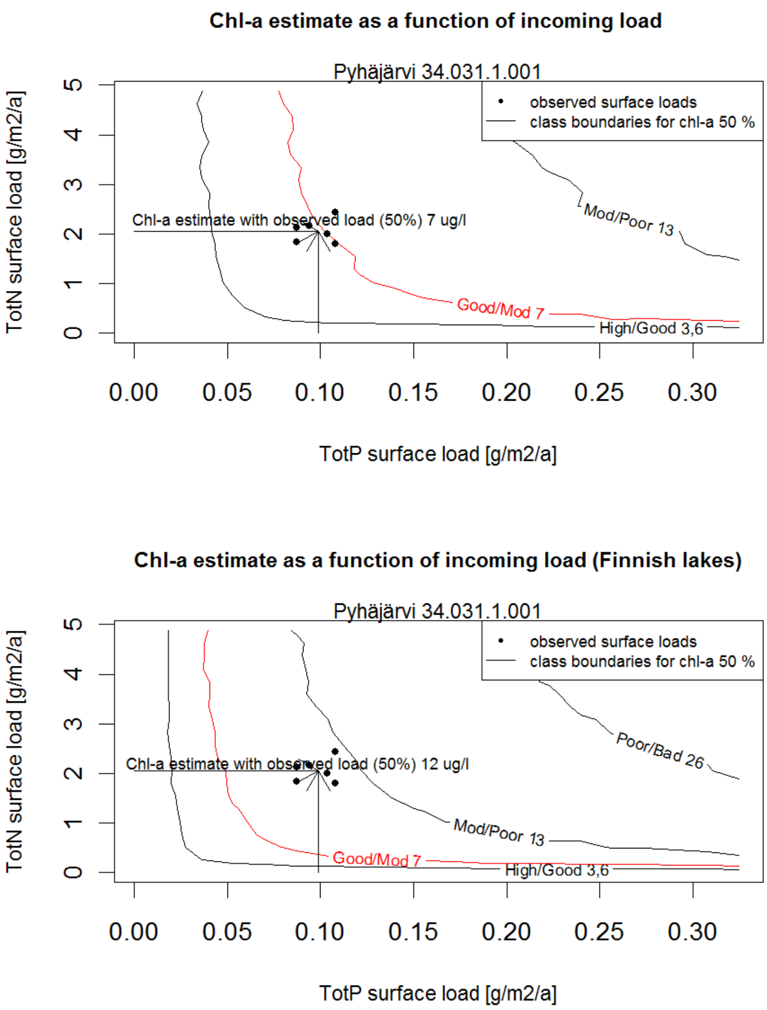

Figure 1: Chlorophyll a estimates of LLR model tool for Lake Pyhäjärvi with Lake Specific Model (upper) and Finnish Lakes model using 50% prediction probability. The phosphorus and nitrogen loading combinations that keep the chlorophyll a concentration below good water quality limit can be read following the Good/Mod line.

average chlorophyll a estimates is the lack of information about the important seasonality; the true problems during warmest periods in summer easily vanish. Because the classification of lakes is also done based on growing season averages, LLR of course meets the needs of management planners dealing with requirements of WFD.

It is worth noticing that internal loading (mostly resuspension due to wind and fish) seems to be very important for the growth of phytoplankton in Lake Pyhäjärvi (e.g. Ekholm et al., 1997; Tarvainen et al., 2010). Thus, the external loading may not be enough to explain the changes in chlorophyll a. In the Finnish Lakes Model the retention of phosphorus decreases when external loading increases, indicating internal loading, but the Lake Specific Model does not take it into account. This may lead to underestimation of the problem, or overestimating the need for external loading reduction, and demonstrates how important it is to know the modeled system well enough, even if the simple model does not really require it.

Especially the loading values are difficult to estimate right, which may have big impact on the results of Finnish Lakes or European and North-American Lakes Model (based on regressions). The lake specific model is not sensitive to loading in that sense, that it would overestimate the chlorophyll a concentrations (Pätynen, 2009). If there is enough data about other variables, the problem with missing loading data can be avoided in Lake Specific model by thinking the loading reduction as percentages instead of absolute values. This of course has to be known, otherwise there is a risk of doing strange predictions from the results of Lake Specific model, if it fits the in-lake concentrations to completely wrong loading values.

It has been shown, that regressions drawn for lakes in a certain area do not necessarily work for lakes in other areas (Phillips et al., 2008). There is a lot of scatter especially in the chlorophyll a – nutrient regressions, because of the many other things affecting the chlorophyll a concentration. As noticed, even the Finnish Lake Model overestimated a bit the chlorophyll a concentration in Lake Pyhäjärvi. However, the hierarchical model structure in chlorophyll a model tries to address this problem by weighting data from the study lake, if it is available. In addition, the general lake data is divided to groups according to the different lake types. This alone is discerned to increase the accuracy of chlorophyll a estimates (Malve, 2007; Phillips et al., 2008).

In practice, the effects of loading reduction may take even 10 years to appear (Jeppesen et al., 2005; Søndergaard et al., 2005), depending e.g. on the loading history of the lake. Also, things like changing climate can create new challenges (for this reason, the effect of temperature in added to the new version of LLR within the EU WISER project), and there is a lot of variation in the nature that can not be taken in to account in simple models. Nevertheless, the management work has to start somewhere, soon and be justifiable to the decision makers and public. That is why it is highly important to provide tools for some basic estimates that are still not just separate values, but have gone through careful evaluation about their reliability.

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Disentangling the multiple stressors affecting benthic invertebrate assemblages in three European inland water ecoregions – a study of Slovenian rivers with implications for management

Maja Pavlin[1], Sebastian Birk[2], Daniel Hering[2], Gorazd Urbanič[1, 3]

[1] Institute for Water of the Republic of Slovenia, SI-1000 Ljubljana, Slovenia
[2] University of Duisburg-Essen, Department of Applied Zoology/Hydrobiology, D-45117 Essen, Germany
[3] University of Ljubljana, Biotechnical Faculty, Department of Biology, SI-1000 Ljubljana, Slovenia

Email address of corresponding author: maja.pavlin@izvrs.si

Key words: *ecoregion, mountains, lowland, Alps, Dinaric, river, multiple stressors, land use, eutrophication, organic pollution, variance partitioning, river-basin management*

Introduction

Running waters are some of the most degraded ecosystems on earth (Giller & Malmqvist, 1998). In most cases, rivers are simultaneously affected by multiple stressors – that interact with each other and are closely related across spatial scales (e. g; Buck et al., 2004; Ormerod et al., 2010). Biogeographical as well as longitudinal, vertical, lateral and time variability patterns are typical of lotic communities in anthropogenically undisturbed conditions (Illies & Botosaneanu, 1963; Vannote et al., 1980; Urbanič & Toman, 2007). Accurately assessing the effects of multiple human-caused stressors on freshwater ecosystems is an essential step in the development of efficient decision support tools for environmental managers (Statzner & Bêche, 2010). Ecoregions are the river typology units that are often used as a frame to define regional water quality and management-goals in the USA and in Europe (e.g. European Parliament, 2000; Loveland & Merchant, 2004; Tison et al., 2005; Urbanič, 2008a). We investigated the variability of natural factors and three stressor groups affecting benthic invertebrate assemblages in rivers of three European inland water ecoregions: the Alps, Dinaric western Balkan and Pannonian (Hungarian) lowland (Illies, 1978; Urbanič, 2008a). We compared how similar and distinct the invertebrate communities in Slovenian rivers were affected by *land use, eutrophication* and *other stressors* (including organic pollution, sedimentation and hydromorphological alteration).

Study area

Rivers in Slovenia drain an area of 20,273 km^2 and extend over four European inland water ecoregions: the Alps, Dinaric western Balkan, Pannonian lowland (Hungarian lowland sensu Illies) and Po lowland (Illies, 1978; Urbanič, 2008a). Carbonate bedrock geology prevails in 50 % of the area, approximately 40 % is siliceous and less than 10 % flysch, but high local variability is present. Based on the predominant geology, altitude and slope of the catchment area, the ecoregions are subdivided into 16 bioregions (Urbanič, 2008b). Within each bioregion, river types are described: by river-size and additional typological descriptors; e.g. karst spring, periodical flooding, intermittency and lake influence (OGRS, 2009; Urbanič, 2011). Slovenia is dominated by forest (over 60 % coverage), which ranks it among the most forested countries in Europe. However, land use categories are not evenly distributed throughout Slovenia (Petek, 2004).

Methods

We selected 260 river sites in three main ecoregions in Slovenia: 71 sites in the Alps, 106 sites in the Dinaric western Balkan and 83 sites in the Pannonian lowland (Fig. 1). The sampling sites covered (near) natural to highly distorted conditions reflecting the various levels of perturbance caused by hydromorphological alteration, pollution and/or catchment land use. Physico-chemical variables and benthic invertebrates were sampled between 2005 and 2008 during low to medium discharge at each site on a single occasion. Rivers were sampled between May and October, except for the 14 large rivers where winter samples were taken due to the natural hydrological conditions. The sampling procedure followed the standard protocols for monitoring of chemical substances (OGRS, 2002) and the standardized Slovenian river bioassessment protocol for benthic invertebrates (OGRS, 2009). Benthic invertebrate samples were processed (Petkovska & Urbanič, 2009) and determined to the taxonomic level used for the ecological status assessment in Slovenian rivers (OGRS, 2009). We classified each sampling site into one of the five hydromorphological (HM) alteration classes (WMI,

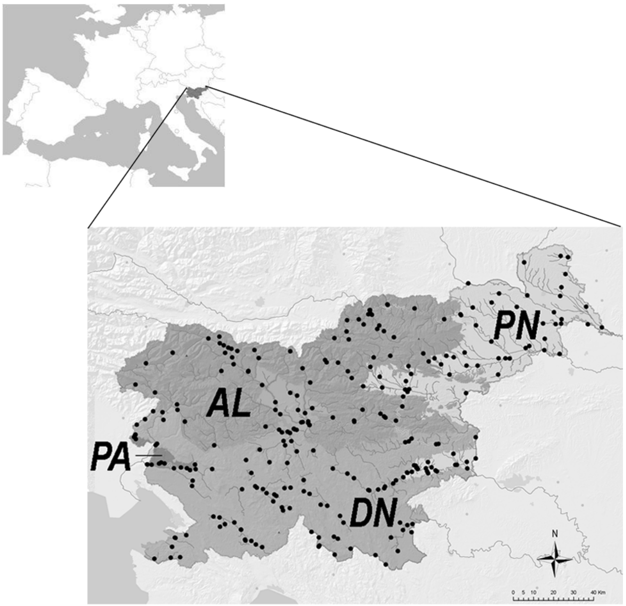

Figure 1: The European inland water ecoregions in Slovenia: AL – Alps, PN – Pannonian lowland, DN – Dinaric western Balkan and PA – Po lowland. Dots present the river sampling sites.

2002). Four land-use categories at the catchment and sub-catchment scale were distinguished (CLC, 2007).

We compiled four groups of explanatory variables describing the typological variability and the main anthropogenic stressors affecting river ecosystems (e.g. Leuven & Poundevigne, 2002; Snyder et al., 2003; Allan, 2004, Dudgeon et al., 2006; Heathwaite, 2010). *Typology* variables were determined based on literature data of the Slovenian river typology (see Urbanič, 2008a, b; Urbanič 2011). The stressor variables were compiled into three groups relevant in river management. The *land use* group consisted of four main land use categories at two spatial scales, with intensive and non-intensive agricultural practices distinguished. A group of potential *eutrophication* variables comprised all available data on inorganic phosphorus and nitrogen compounds in the water. The group of *other* stressors included parameters of organic pollution, hydromorphological alteration and other indicators of human pollution that comprised water concentrations of chloride, sulphate ions (Barendregt & Bio, 2003) and total suspended solids (Tab. 1).

Canonical Correspondence Analysis (CCA; ter Braak & Prentice, 1988) and partial Canonical Correspondence Analysis (pCCA; ter Braak, 1988) were applied to investigate the variability of the invertebrate communities among sampling sites. In all analyses, we used the option "down-weighting of rare species (taxa)" in CANOCO 4.5 (ter Braak & Šmilauer, 2002). Automatic

*Table 1: Environmental variables of the variable-groups: T – typology, E – eutrophication, L – land use and O – other stressors. Explanatory-variable-ranks in decreasing order of their marginal effects are given for each ecoregion : AL – Alps, PN – Pannonian lowland and DN – Dinaric western Balkan. *forward-selected variables.*

Environmental Variable	Variable group	AL	PN	DN
Catchment size class	*T*	*3**	*1**	*4**
Intermittentency	*T*	*22*	*-*	*10**
Karst spring upstream	*T*	*22*	*-*	*4**
Altitude	*T*	*9**	*2**	*21*
Slope	*T*	*1**	*11**	*1**
Alkalinity	*T*	*20**	*16*	*26**
Total Phoshorus	*E*	*13*	*8**	*7*
Orthophosphate	*E*	*11*	*8**	*10*
Total Nitrogen	*E*	*11**	*24*	*10**
Ammonia	*E*	*18*	*16*	*10*
Nitrite	*E*	*13*	*16*	*7**
Nitrate	*E*	*9**	*16*	*10**
BOD5	*O*	*15*	*16*	*17*
Oxygen saturation	*O*	*27*	*16*	*10*
Total Organic Carbon	*O*	*15**	*13*	*2**
Chloride	*O*	*4**	*6**	*21*
Sulphate	*O*	*20*	*3**	*7**
Hydromorphological alteration	*O*	*22**	*10*	*21*
Total Suspended Solids	*O*	*18**	*13*	*21*
Land use - Subatchment				
Urban	*L*	*4**	*11*	*17*
Natural and semi-natural	*L*	*8*	*16*	*17*
Intensive agriculture	*L*	*26*	*13*	*17*
Non-intensive agriculture	*L*	*22*	*25*	*27*
Land use - Catchment				
Urban	*L*	*1**	*3**	*2**
Natural and semi-natural	*L*	*7**	*6**	*10**
Intensive agriculture	*L*	*6*	*3*	*4**
Non-intensive agriculture	*L*	*15*	*16**	*21*

forward selection procedure was applied within each explanatory variable-group to avoid over-estimation of the explained variance (Borcard et al., 1992; Økland & Eilersen, 1994). Explanatory variables were ranked in decreasing order of their marginal effects (λ_1). We partitioned the benthic invertebrate variability into individual and combined effects (Økland & Eilersen, 1994; e.g. Sandin & Johnson, 2004) of the explanatory variable-groups: (a) each stressor-group and *typology* group; (b) stressor-groups: *land use, eutrophication* and *other stressors*. A more specific method-description provide Pavlin et al. (2011).

Results

Typology and *stressor-groups* individually explained considerable shares of the explained invertebrate variability in each of the ecoregions: Alps, Dinaric western Balkan and Pannonian lowland (from 54 % in the Alps up to 95 % in the Dinaric western Balkan) . This indicated

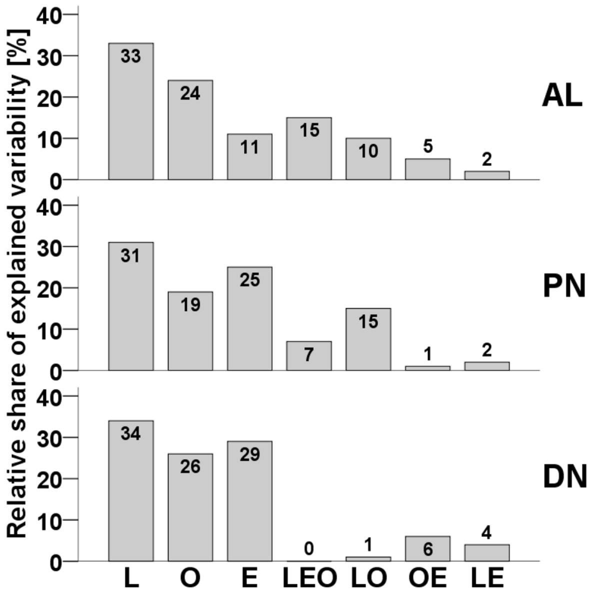

Figure 2: Relative amount of variance explained by each individual group of anthropogenic stressors: L –land use, E – eutrophication, O – other stressors and their combinations in three Ecoregions: AL – Alps, PN – Pannonian lowland, DN – Dinaric western Balkan.]

the presence of anthropogenic land use, eutrophication and other stressor-issues in the Alpine, lowland as well as Dinaric rivers (catchments).

Environmental variables had diversified explanatory power for the assemblages in different ecoregions (Tab. 1). In each of the studied ecoregions, we could relatively well discern among the individual impacts of *land use, eutrophication* and *other stressors* (organic pollution, hydromorphological alteration) on the invertebrate fauna. The responses to the three-stressor groups were best discerned for the rivers of the Dinaric western Balkan but least efficiently for the rivers of the ecoregion Alps (Fig. 2). We could well separate between the responses to nutrient enrichment and other stressors in the rivers of the Dinaric western Balkan and the Pannonian lowland, but not in the Alps. Benthic invertebrate responses to catchment land use were in a meaningful share not explained by the in-stream stressor variables, especially in the Alpine rivers. The responses of the invertebrate community to land use, eutrophication and other stressors were more similar in rivers of the Pannonian lowland and the Alps, compared to the rivers of the Dinaric western Balkan ecoregion (Fig. 2).

Discussion

Effects of environmental stressors are usually tested individually (e.g. Beketov, 2004), but in nature organisms are often exposed to several stressors simultaneously (Folt et al., 1999). In river basin management, it is of special importance to recognise the reasons of ecosystem impairment before searching for possible measures. Ecoregions that are widely used as river-management units (e.g. Clarke et al, 1991; Wickham et al., 2005), are characterised not only by specific natural features but also by a suit of diverse stressors (Loveland & Merchant, 2004;Urbanič & Toman, 2007; Urbanič, 2008a) . Pavlin et al. (2011) recently found benthic invertebrates responded diversely to stressor-groups in Slovenian rivers at a cross-ecoregional scale. Here, we showed important effects of all three stressor-groups (*land use, eutrophication* and *other stressors*) in rivers of each ecoregion in Slovenia. Benthic invertebrate assemblages of different ecoregions responded to the same stressor groups diversely. However, the main pattern remained: the combined stressor effects were low compared to their pure effects, in rivers of all ecoregions but especially in the Dinaric western Balkan.

Nutrient enrichment and organic pollution in particular are often closely related (Friberg et al., 2009). Authors often report the responses of benthic invertebrates to nutrient concentrations (Skoulikidis et al., 2004; Camargo et al., 2004; Smith et al., 2007), but either tend not to consider parameters of organic pollution at the same time, or handle the effects of the two stressors together (e.g. Yuan, 2004). Pavlin et al. (2011) found nutrient enrichment and other stressors (including organic pollution) affect benthic invertebrates differently, in Slovenian rivers. Here, we could clearly distinguish the eutrophication effects from other stressors in rivers of the Dinaric western Balkan or the Pannonian lowlands. In Alpine rivers, the combined effects of eutrophication and other stressors' were prominent. The effects linked to some stressor-groups were more efficiently discerned in some ecoregions, than others were (Fig. 2).

In the multiple stressor-environments, specific combinations of several stressors occur at different intensities. The stressor-effects can thus be unpredicTab. at new regions due to the specific natural characteristics and the un-repeaTab. stressor-combinations that co-occur. Studying the stressor-effects on the river ecosystem at the ecoregion or lower spatial scale can give refined insights to locally relevant management-issues. At the ecoregion level, we have recognised diverse responses to the same stressor groups: land use, eutrophication and other stressors. We emphasize the importance of locally conducted studies (e.g. ecoregion specific) to guide the river-management.

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A phytoplankton trophic index to assess the status of lakes for the Water Framework Directive

Geoff Phillips[1], Anne Lyche Solheim[2] & Birger Skjelbred[2]

[1] Environment Agency, Kings Meadow Road, Reading, Berkshire, RG1 8DQ, UK
[2] NIVA, Gaustadalléen 21, N-0349 Oslo, Norway

Email address of corresponding author: Geoff.Phillips@environment-agency-gov.uk

Key words: phytoplankton, freshwater, lake, classification, Water Framework Directive, phosphorus

Abstract

Improvements in waste water treatment systems are starting to result in improvements to rivers and lakes in Western Europe, however the impact of anthropogenic nutrient sources remains one of the key concerns for the management of European lakes. The Water Framework Directive (WFD) provides a mechanism through which further progress can be made on this aspect of water management. The Directive requires a classification of the ecological status of lake phytoplankton which includes an assessment of its taxonomic composition. All European countries are required to develop assessment systems, these range from simple metrics, such as the proportions of Cyanobacteria or Chrysophytes, to more sophisticated trophic indices based on trophic scores of taxa along a nutrient gradient. One of the requirements of the WFD is to compare these different national systems to demonstrate that each country provides a status assessment that is similar for a given level of pressure. One way to make this comparison is through the use of an independent common metric. We present here a new pan-European phytoplankton taxonomic index (PTI) that can be used for this purpose to assess lake status.

The metric was developed from a dataset containing data from 21 European countries and over 1500 lakes. We selected a training set of data from the summer period (July – September) and used Canonical Correspondance Analysis with a single constraining environmental variable, total phosphorus to produce a set of taxa optima from the 1st ordination axis. These optima were then used to generate sample scores using a weighted average of the proportion of the biovolume of each taxa present in the sample. The resulting index was shown to have a good relationship with pressure measured as total phosphorus (GAM model $R^2 = 0.667$ $p<0.001$), but was different for lakes of low, moderate and high alkalinity. To allow for this the index was converted to an Environmental Quality Ration (EQR) by dividing by type specific reference values, derived from a model which used a population of reference lakes to predict reference PTI values.

The metric was subsequently successfully used as the taxonomic component of a common metric to compare the status of European phytoplankton assessment systems through the intercalibration process.

Time response of seagrass indicators to discrete disturbances

G. Roca[1], S. Columbu, S. Farina, A. Gera, Jordi F. Pagés, J. Romero, T. Alcoverro

[1] Center for Advanced Studies of Blanes (CEAB-CSIC), Accés Cala St. Francesc 14, Blanes 17300, Spain

Email address of corresponding author: groca@ceab.csic.es

Key words: seagrass, BACI, monitoring, indicator response

Abstract

Most water quality bioindicators have been tested to discriminate between different water quality conditions, however, their ability to detect changes, particularly after conditions have been modified, has received much less attention. Therefore, it is essential to reinforce or modify some of the ongoing environmental water policies, which are in part based on those monitoring programs. Negative disturbances in ecosystems are often diffuse in space and time. However, some processes, such as the construction of marine infrastructures generate discrete perturbations of high intensity, short spatial extension and duration. Those reduced dimensions generate excellent scenarios for studying the response capacity of indicators in seagrass beds.

Here we take advantage of the construction of a new harbour in Blanes (North East coast of Spain) that is located close to a *Posidonia oceanica* meadow which allows us to test in situ the response capacity and potential recovery of most seagrass indicators used for water quality monitoring within the WFD and in other monitoring networks.

Specifically, we have studied the response of 22 commonly used bioindicators in *Posidonia oceanica* monitoring programs before and after the construction of the harbour lasted for 2 months. The three main questions included: (i) to determine which environmental drivers (i.e. light, sediment) were affected directly by the building of the new harbour; (ii) to find out which indicators were able to respond to those pressures (early indicators) and (iii) to determine if the indicators affected by the construction where able to recover its basal states after the ceasing of the disturbance. In order to answer these questions, we used a beyond BACI design. We sampled one impacted site and three control locations one time before the impact, and three to four times after the impact, depending on the variable. At each sampling (before and after) and site (3 controls and 1 impacted) we measured 22 seagrass indicators encompassing structural, morphological, community and physiological variables (Table 1). A total of 5 to 12 replicates per site and time were obtained for each variable (Table 1). Additionally, during and after the impact (1 year) we measured light availability with quantum light sensors and sediment deposition using sediment traps at each site (Table 1).

Fine sediment deposition increased 30 times at the impact site compared to the control sites (BACI analysis, P<0.05). Light was also significantly reduced close to the impacted site by an order of magnitude when compared to the control sites (BACI analysis, P<0.05) (Fig.1).

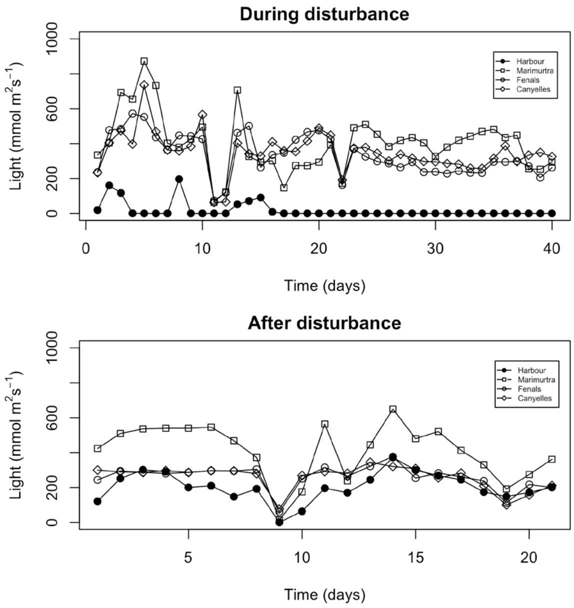

Figure 1: Light availability at canopy level of the four sites (one impacted black circles and 4 control, the rest) during the disturbance (may 2010) and after the disturbance (May 2011).

One month after the impact no clear differences were detected (although trends were detectable) for the studied variables (BACI analysis, P<0.05) (Fig. 2). However, two months after the impact some physiological variables such as sucrose, starch and total carbohydrate, were reduced significantly at the impacted station (BACI analysis, P<0.05) (see Fig. 2). Fe and Mn also experienced a substantial increase in the meadow next to the harbour after the first month of the disturbance

Table 1: a. Metrics belonging to physiological level vary within the tissue age and are influenced by the presence of epiphytes. To avoid these sources of variability and to work on sufficient material for all the physiological metrics analyses, the medium and basal part of the fully extended leaf number 3 (the third youngest leaf in the shoot, without conspicuous epiphytes in the medium and basal parts) of five shoots, and the first rhizome centimetre, of the same five shoots, constitute one replica of leaf and rhizome material, respectively. They were dried, finely grinded and analysed together.
b. ICP: Inductively Coupled Plasma; IRMS: Isotope Ratio Mass Spectrometry. List of the metrics used with their expected responses to changes in environmental quality, and the outline of the pertinent sampling/analytical methods.

Level	**Metric (and units)**	**Expected response to increasing anthropogenic disturbances**	**Standard measured method**
Water turbidity (Driver descriptors)	Light availability at canopy level (mmol m^2/s)	Decrease	Light sensors PAR QSO-Sun 2.5v connected to HOBO u12-013 data logger placed just above canopy level one month during the disturbance (may 2010) and one month after the disturbance (may 2011)
Sediment deposition (Driver descriptors)	Sediment deposited in traps (g)	Increase	Six sediment cylindrical traps of (16cm* 4.5 cm diameter) attached in groups of three in two independent tripods in each site were installed
Sediment composition (Driver descriptors)		Change	Three 50ml containers were filled with superficial sediment during each sampling in all sites.
	Organic matter %		%O.M. was determined by the difference of weight after burning sediment at 500ºc five hours in the muffle
	Sediment grain composition	Change	Grain composition was analysed by optical particle analyser Mastersizer 2000
Physiological level (Plant descriptors)			Five replicates of 2 shoots each one were randomly sampled in all sites
	Nitrogen and phosphorus content in rhizomes and leaves (%DW)	Increase	[a]Analysed using IRMS[b] (for N), and optic ICP[b] analysis after acid digestion in an HNO_3 and H_2O_2 solution at 180ºC 20 minutes in microwave (P)
	Soluble carbohydrate reserves in rhizomes (%DW)	Decrease	Extracted from 0.05 g DW[a] in hot EtOH (80 ºC) centrifuged at 4500 rpm (4 times) EtOH was evaporated to dryness under a stream of N_2, extracts were redissolved in distilled water and analysed spectrophotometrically (k = 626 nm) using anthrone assay standardized to sucrose (Alcoverro et al., 1999, 2001b
	(δC^{13}) in rhizomes and leaves	Increase (fish farm or urban effluents)	Sample (0.7–0.8 mg DW [a]) analysed using isotope ratio mass spectrometry using atmospheric nitrogen as standard (Romero et al 2007)
	Nitrogen isotopic ratio (δN^{15}) in rhizomes and leaves (% DW)	Decrease (fertilisers) depending on the N source	
	Sulphur isotopic ratio ($\delta^{34}S$) in rhizomes (‰)	Decrease	6 mg DW[a]) analysed using isotope ratio mass spectrometry) using CDT (Canyon Diablo Troilite) as standard
Individual level (plant descriptors)			Five replicates for each site and time
	Shoot surface (cm^2/shoot)	Decrease	Leaves (length and width) were measured to obtained shoot surface from five different shoots per location
	Leaves with necrosis (%)	Increase	Frequency of leaves with necrosis (as a percentage) obtained from direct observation in the laboratory
Population level (Meadow descriptors)	Shoot density (shoots/ m^2)	Decrease	Shoots number was counted in 12 (40 * 40 cm) quadrats, 3 fixed and 9 randomly placed over a ca. 400 m^2 area, excluding zones with zero cover (Renom and Romero, 2001)
Community level	Epiphytic biomass	Increase	Five replicates for each site and time Epiphytes were extracted from the leaves using glass slides
Pollution	Trace metals in plant tissues (µg/gDW)	Increase	Five replicates for each site and time. Analysed by optic ICP[b] (for Zn) and mass ICP[b] (for Cu, and Pb) from 0.1 g DW[a] after digestion in an HNO_3 and H_2O_2 solution at 180ºC 20 minutes in microwave. The analytical procedure was checked using standard reference material (Ulva lactuca, CRM 279) (Romero et al., 2007)

(BACI analysis, P<0.05). Eight months after the impact, seagrass density was significantly reduced (BACI analysis, P<0.05) (see fig.2).

The rest of the parameters analyzed (%N, %C, $\delta^{13}C$, $\delta^{15}N$, either leaves or rhizomes, $\delta^{34}S$, Ni, Cd, Cu, Ni, Zn, P, Epiphyte biomass) were not able to detect any significant changes after the disturbance.

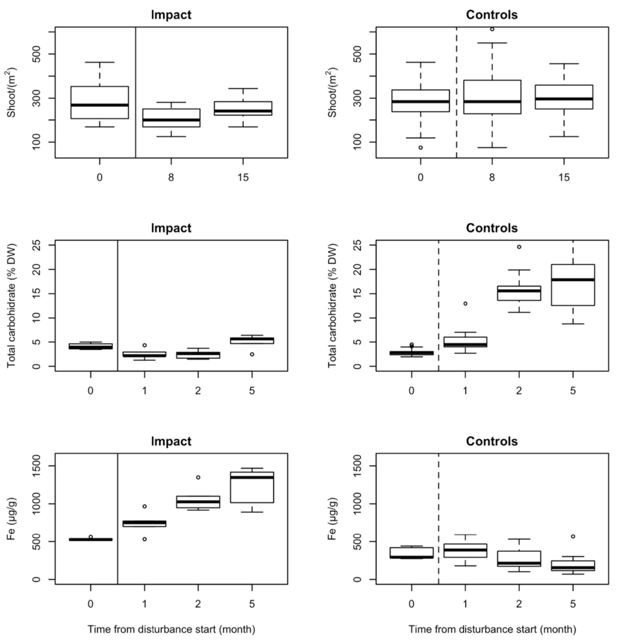

Figure 2: Boxplots of the impacted site and the three controls sites including means of Density, Total carbohydrate and Fe before (time=0) and at different times after the disturbance.

One year after the disturbance, the water was clear again, light availability recovered to its normal values (Fig. 1)(BACI analysis, NS), and fine sediment that had accumulated over the meadow was washed out (personal observation). Nevertheless, once the water conditions had recovered, indicators that reported the degradation of the ecosystem quality did not recover after one year, although certain trends of improvement could be detected with physiological indicators. Hence, our study shows that most indicators have important time lags responses to environmental improvements that need to be taken into account. Longer time series will be needed in this type of ecosystem to be able to detect any recovery.

When management protocols are established, it is not only important to make sure that indicators are able to respond fast enough to ecosystem degradation, it is also crucial to employ adequate indicators that reflect improvements when they occur. Determining the response time of indicators to ecosystem quality changes (degradation and improvement) would enable us to evaluate if management actions are giving desired results. Physiological indicators are the first ones responding to disturbances and are likely to be the first indicators responding to improvement, and are therefore recommended to be included in monitoring programs.

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Evaluation of macroinvertebrate indices for the assessment of lake and river acidification

Ann Kristin Schartau[1]**, Zlatko Petrin**[2]

[1] Norwegian Institute of Nature Research, Gaustadalleèn 21, NO-0349 Oslo, Norway
[2] Norwegian Institute of Nature Research, P.O. Box 5685 Sluppen, NO-7485 Trondheim, Norway Department of Biosciences, University of Aarhus, Denmark

·Email address of corresponding author: ann.k.schartau@nina.no

Key words: classification, diversity, low alkalinity, humic substances, Water Framework Directive, monitoring

Abstract

Assessment of ecological status of lakes and rivers according to the EC Water Framework Directive (WFD) requires high quality monitoring data for phytoplankton, macrophytes, invertebrates and fish in addition to hydromorphological and hydrochemical data. Among the biological quality elements, littoral macroinvertebrates and fish have been shown to be highly sensitive to acidification. In Norway, macroinvertebrates have been included in the monitoring programme on long-range transported air pollutants since the beginning of 1970s (rivers) and 1996 (lakes); hence, these datasets represent some of the longest time-series for macroinvertebrates in Europe. Based on the acid-tolerance of specific macroinvertebrate taxa, Raddum and Fjellheim (1984) developed a simple acidification index (Index 1) for the assessment of Norwegian rivers. This index has also been used in other countries in Northern Europe (e.g. Ireland). Later Index 1 was adjusted to also take into account sublethal effects of acidification (Index 2; Raddum and Fjellheim, 1994). Both indices were developed for the assessment of river acidification, but Index 1 has also been used for the assessment of lake's acidification (indicated by macroinvertebrate communities of lake's outlet).

However, the intercalibration process undertaken at the European level has shown that above indices do not meet all the requirements of the WFD. For instance, for Index 1 no reference value is defined, and neither Index 1 nor Index 2 includes all metrics that would be indicative of the function and structure of macroinvertebrate communities. Furthermore, none of the above indices appear suiTab. for the assessment of humic water bodies, or for sites that are characterized by very low alkalinity.

As a part of the Norwegian research project BIOCLASS-FRESH, we are now testing various macroinvertebrate indices for the assessment of lake and river acidification. The indices tested were developed by the Northern Intercalibration Group WG Macroinvertebrates (McFarland, unpublished) and include both simple indices based on the presence/absence of acid-sensitive taxa as well as multi-metric indices, which include all metrics that are required by the WFD: taxonomical composition, abundance, diversity and tolerant/sensitive taxa. We tested the dose-response relationships between the indices and the acidification gradient using (generalized) linear fixed and mixed models and analysis of covariance. pH, Acid Neutralizing Capacity (ANC) and labile aluminium were used as indicators of acidification, and the concentration of calcium and the content of humic substances were included as co-variables. Our analyses show that both calcium concentration (alkalinity) and the content of humic substances need to be taken into account in the assessment of acidification. The results were insignificant for a few acidification indices and complex for most others, i.e. the effects of certain measures of acidification, e.g. pH, depended on the specific levels of other variables suggesting complex interacting effects. Many of the indices were suiTab. for the assessment of acidification at sites that were characterized by low alkalinity and clear water as suggested by goodness-of-fit measures indicating that > 50 % of the variation in the data was explained by the independent variables and highly significant ($p<0.001$) effects of the measures of acidification. However, no significant differences were found between reference sites and acidified sites for humic water bodies. Therefore, a new approach is probably required to be able to establish a robust assessment system for acidification of humic rivers and lakes. This may include for instance on site-specific reference conditions, other indicator taxa and new metrics (e.g. functional traits). For sites with very

low alkalinity (Ca $<$ 1 mg/L, alk $<$ 0.05 meq/L), none of the tested indices were suiTab. indicators of acidification. Lakes and rivers that are characterized by very low alkalinity are highly sensitive to acidification. Such water bodies are very common in Norway, but rare elsewhere in Europe. Consequently, assessment systems for lakes and rivers with very low alkalinity are urgently needed, and such systems are likely to be based on new acidification indices yet to be developed.

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Data about data – the WISER metadatabase

Astrid Schmidt-Kloiber[1], Bernard Dudley[2], Jannicke Moe[3], Jörg Strackbein[4], Robert Vogl[1]

[1] University of Natural Resources and Life Sciences (BOKU), Institute of Hydrobiology and Aquatic Ecosystem Management, Max-Emanuel-Straße 17, 1180 Wien, Austria
[2] Centre for Ecology & Hydrology (CEH), Bush Estate, Penicuik, Midlothian, EH26 0QB, UK
[3] Norwegian Institute for Water Research (NIVA), Gaustadalléen 21, 0349 Oslo, Norway
[4] University of Duisburg-Essen (UDE), Faculty of Biology, Universitätsstrasse 5, D-45141 Essen, Germany

Email address of corresponding author: astrid.schmidt-kloiber@boku.ac.at

Key words: metadata, metadata discovery, database structure, accessibility, intellectual property rights (IPR)

Introduction – Aim and approach of the metadatabase

Metadata is loosely defined as "data about data". A metadatabase therefore should gather information on datasets in order to allow data visibility and assessment. For the data producer/provider metadata are meant to document data to inform prospective users of their characteristics, while for the data consumer/user metadata are used to both discover data and assess their appropriateness for particular needs – their so-called 'fitness for purpose'.

Integrated in WISER Module 2 (Data and guidelines) and Workpackage 2.1 (Data service) the aim of the WISER metadatabase was to summarise available data for the project's purposes. This included the compilation of information on both existing data as well as new data from the field exercises and from ongoing monitoring programmes. The basis for the WISER metadatabase was a list of ca. 100 datasets/databases, which WISER partners had announced as available for the project in the project proposal.

More specifically the metadatabase aimed to give an overview on

– general data availability per water category, biological quality element (BQE), geographical intercalibration group (GIG)
– availability of environmental data
– data comparability and data precision (e.g. regarding the taxonomic resolution)
– usability/accessibility of data (i.e. the intellectual property rights (IPR) of the datasets)

Methods – Structure of the metadatabase

The construction of the WISER metadatabase started with the compilation of information that workpackage (WP) members might need for their work. On this base a questionnaire was developed and further evaluated and amended by Module and WP leaders as well as project partners. In the next step this questionnaire was made available online. The content was divided into several information blocks (see below), which were then filled for each of the datasets by the project partners. The final step was the development of a query tool.

Generally, the metadatabase therefore consists of two main parts

– an online questionnaire to fill in metadata
– an online query page to find data

While the questionnaire is only accessible for WISER partners and metadata providers, the query tool is available through the website and can be used by all scientists and the interested public (http://www.wiser.eu/results/meta-database/).

Metadata questionnaire

The final structure of the WISER metadatabase and the accompanying questionnaire consists of ten main information blocks containing a variety of data characterising fields (see Table 1). To facilitate data entry for data providers most of the fields were designed as check boxes, radio buttons or selection lists. For additional information several comments-fields are available. Further, a handbook for providing supplementary information was compiled to give help while entering data.

The metadatabase query tool

The metadata query tool is available via a web interface and should help WISER scientists to find appropriate datasets for their analysis and method development. Further it should serve to gain information about these datasets, especially about the intellectual property rights and the accessibility of the data through the central database (CDB).

Table 1: Information blocks of the metadatabase and their specific content.

information block	content
general information	*database ID* *database name* *aim of the database* *short description of the database*
technical information	*operating system* *database format* *access level* *update level* *filling of gaps* *documentation* *technical contact person* *scientific contact person*
intellectual property rights	*ownership of the dataset* *availibility of the dataset* *criteria for using the dataset (during and after WISER)* *data management after the termination of WISER*
site specifications	*countries* *water category (ecosystem type)*
site specifications per water category	*number of sites* *number of waterbodies/lakes* *coordinate system* *coordinate format* *number of sites with coordinates* *site coding* *number of sites per GIG* *ecoregions* *number of sites/lakes per WFD System A criterion* *other site characteristic parameters*
environmental data	*number of sites per stressor type (eutrophication, hydromorphological degradation, acidification, organic pollution, toxic stress, general degradation)* *available data on hydromorphology* *site protocols*
biological data - overview	*number of samples per sample category*
physico-chemistry	*percentage of covered samples per parameter (total P, ortho P, total dissolved P, nitrate, nitrite, total N, ammonium, BOD, oxygen, water temperature, chlorophyll, hardness, pH, conductivity, alkalinity, Ca, colour, Secci depth, euphotic depth, termocline depth, mean depth, current velocity, substrate composition, other physico-chemical parameters)*
sample specification per sample category	*taxonomic level* *taxonomic coding* *sample type* *station/habitat* *replicate samples* *season* *covered timeframe* *data origin*
other specifications	*availability of predefined queries* *availability of GIS layers, shapes* *availability of photos* *availability of maps* *general comments*

The tool consists of ten main query blocks for specifying the search: water category (ecosystem type), GIG, typological criteria, ecoregions, country, stressor type, restored sites per stressor type, biological quality element, sample type, season.

The query design immediately displays a result table of appropriate datasets after choosing a selection criterion. IPR issues and the availability of datasets in the CDB are indicated with traffic light systems.

Results – Content of the metadatabase

The WISER metadatabase currently describes 21 river, 71 lake and 20 coastal/transitional datasets making a total of 112 datasets. While eight datasets were compiled based on the WISER field exercises (one for each BQE in lakes and coastal waters), the rest originates from previous EU-funded and national projects, as well as finalised and ongoing monitoring initiatives. The numbers of datasets as well as the numbers of sampling locations per water type are displayed in Figure 1. Views of the metadatabase query tool can be seen in Figure 2.

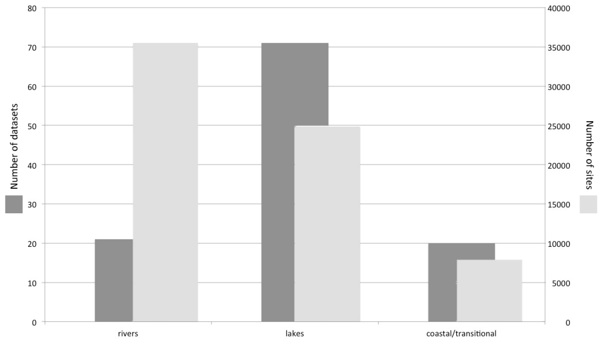

Figure 1: Number of datasets (dark grey) and sampling stations (light grey) per water type

The CDB contains 57 of the datasets listed in the metadatabase. For more information on access to these data see separate presentation of the CDB nearby (Moe et al. 2011).

Discussion – Usability of the WISER metadatabase

Collecting metadata has become a major task in environmental sciences (Costello 2009, Whitlock 2011), both for storing information on datasets, databases and data repositories, as well as for detecting appropriate data for scientific purposes. Most of these metadata collections use a standard set of information fields such as the "Dublin Core" or "Darwin Core" (including biological species information) metadata. There are also dedicated metadata harvesting, search and retrieval tools like Mercury (Devarakonda et al. 2009), which should facilitate the scientist's life.

Initially the WISER metadatabase was solely meant for the scientific work within the project. Together with other tools developed in the data service workpackage (Dudley et al 2011, Moe et al. 2011) it should help WP members with their analyses. Therefore the fields offered in the metadatabase questionnaire were designed for finding appropriate data. In the first instance they were not matched with standard metadata fields as proposed by e.g. the Federal Geographic Data Committee (FGDC) and data were also not recorded in a standard format as e.g. Dublin-Core, Darwin-Core, EML (ecological metadata language; http://knb.ecoinformatics.org/software/eml/) or ISO standards (e.g. ISO-19115).

However, with the development of the metadatabase query tool and the online availability of it, the WISER metadatabase has the potential to become a widely used

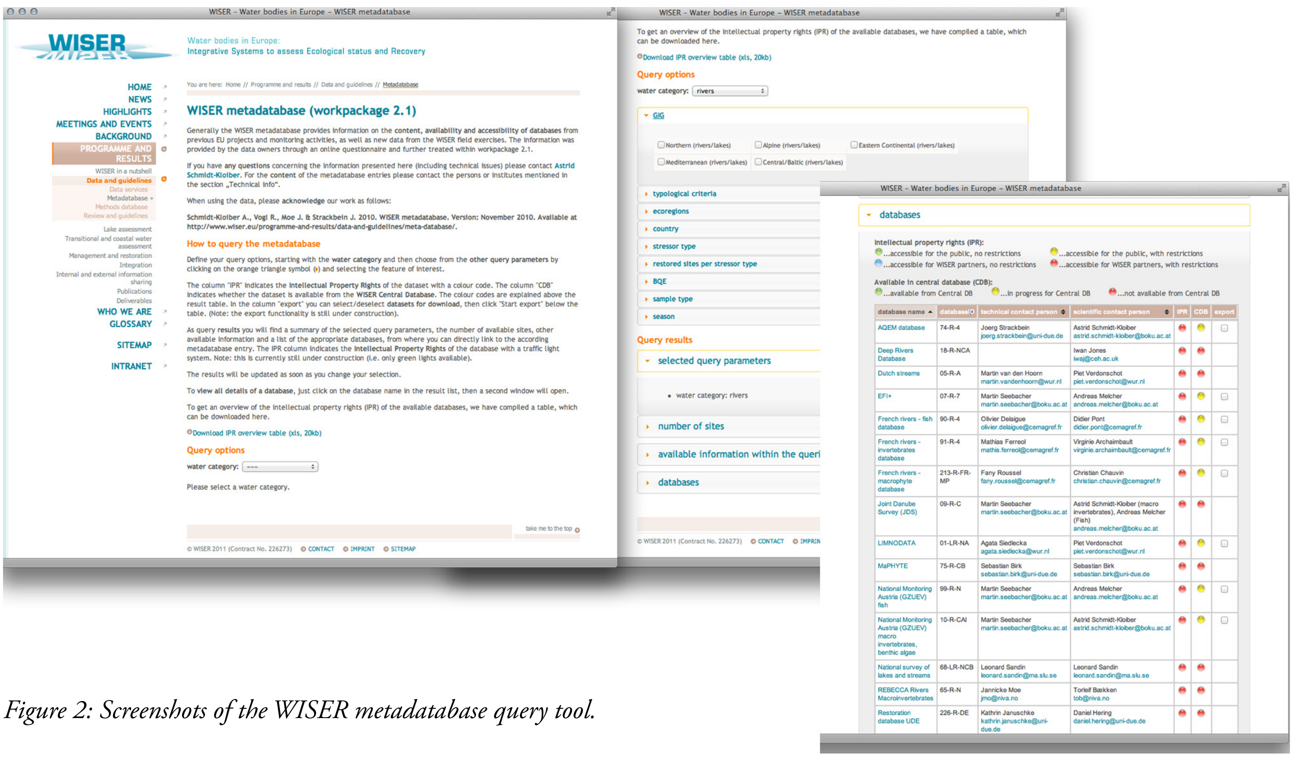

Figure 2: Screenshots of the WISER metadatabase query tool.

tool in applied water issues. This is particularly true as it also holds information on datasets of the Geographical Intercalibration Groups (GIG) as well as other publically funded databases. Especially regarding the accessibility and the Intellectual Property Rights (IPR) the metadatabase is a valuable resource for gaining appropriate information in terms of where to get data and the regulations for using this data.

Acknowledgements

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Lessons Learned from Large-Scale Coastal Ecosystem Restoration: More Synthetic and Strategic Planning Needed

Charles ("Si") Simenstad[1], Denise Reed[2], William C. Dennison[3]

[1] School of Aquatic and Fishery Sciences, University of Washington, 1122 N.E. Boat Street, Box 355020, Seattle, WA 98195-5020 USA; Voice: (1) 206 543 7185, FAX: (1) 206 685 7471; simenstd@u.washington.edu, fish.washington.edu/simenstad

[2] Department of Earth and Environmental Sciences, University of New Orleans, New Orleans LA 70148; Voice: (1) 504 280 7395, FAX (1) 504 280-7396; djreed@uno.edu, http://www.pies.uno.edu/

[3] University of Maryland Center for Environmental Science, Cambridge, MD 21613 USA; Voice: (1) 410 221 2004; dennison@umces.edu, www.ian.umces.edu

Email address of corresponding author: simenstd@u.washington.edu

Introduction

Restoration of coastal ecosystems has been an increasing focus of both governmental and non-governmental institutions throughout coastal North America, driven in part by the magnitude of their decline in quantity and quality but also the implications of the erosion in inherent ecosystem goods and services they provide. Coastal emergent wetlands alone[1] have declined by ~3,000 km^2 since the 1950's, and even between 2004 and 2009, during a period of increasing governmental regulation and protection, vegetated estuarine emergent wetlands have still declined by 2.4% in the United States.

In addition to regulations to limit further loss of wetlands due to development and other land use practices (i.e., Clean Water Act), proactive restoration in the United States is funded and organized through a variety of governmental agencies and other institutions. These range from the Estuary Restoration Act of 2000, which authorizes up to $12.5 million per year for various coastal restoration actions that promote an ecosystem or watershed approach to establish the self-sustaining structure and function necessary to support interrelated physical, biological, and chemical components of healthy estuarine habitats; to community-based restoration initiatives such as the National Oceanic and Atmospheric Administration's (NOAA) funding to target regional restoration priorities and leverage NOAA contributions with local resources. However, public resources and energies for such non-regulatory (i.e., not associated with a legal requirement for compensation) restoration are becoming limiting.

[1] A very conservative, restrictive class of tidal wetland "marsh" that does not include freshwater tidal scrub-shrub and swamps that are integral components of coastal ecosystems mosaics.

While individual, "opportunistic" restoration actions definitely contribute, often significantly, to recovery of damaged patches of landscape, recovery of many ecosystem goods and services ultimate depends on synthetic and strategic restoration planned at the landscape to watershed scale. Integrated restoration of ecosystems and the goods and services they provide is not functionally equivalent to sum of the individual parts. The scientific and planning rationale for this assertion is:

1. ecosystem processes that are required for sustainable restoration are seldom confined to small spatial scales;
2. emerging, and typically accelerating, stressors–climate change, development, water demand–operate at large spatial scales; and,
3. opportunities for sustainable restoration often lay outside the historic template over which restoration is often focused.

Thus, we are likely to run out of resources and knowledge before we randomly attain "critical landscape"! We argue that there is ever greater, more urgent need for restoration planning that is synthetic and strategic. In applying synthetic to restoration, we are referring to the need to combine and integrate our knowledge of the dynamic processes and factors that will determine the sustainability of restoration at the system scale. For the purposes of this argument, we define strategic as an approach to comprehensively guide the allocation of resources toward large-scale and effective restoration. To be successful, strategic restoration needs to be undertaken within a watershed perspective and incorporate large scale concepts and theory, in both geomorphology and ecology, that have been developed over the last 40 years, but also give due consideration to local knowledge and stakeholder opinion (Skinner and Bruce-Burgess 2005).

Despite this analytical perspective, economic, social and institutional constraints often conflict with the comprehensive restoration planning required to achieve more synthetic and strategic restoration. Future restoration will be confronted with difficult challenges, but also some potential…perhaps requisite…opportunities:

– How do we scale up beyond opportunistic, ad hoc, individual, community-based restoration to landscapes?
– How do we balance restoration approaches that are scientifically strategic with those that are institutionally implementable?
– How do stakeholders become engaged and invested in restoration planning that will require extensive intervention in a human accommodated landscape?

Case Studies

The three authors bring different perspectives in three different coastal systems with some common "lessons learned" derived from several decades of experience with restoration planning at multiple scales. We briefly describe a synthesis of our experiences from comprehensive restoration efforts in Chesapeake Bay (W. Dennison), Louisiana coast (D. Reed) and Puget Sound (C. Simenstad) that while representing different symptoms, drivers, approaches and futures (Table 1) convey some comparable lessons. An earlier, comprehensive summary and comparison of these and other comprehensive coastal restoration programs appeared in VanCleve et al. (2006).

Table 1: Comparison of characteristics, problems, comprehensive restoration approaches and future challenges in three coastal regions of the United States.

System	Symptoms	Drivers	Comprehensive Restoration Programs, Duration and Investment	Goals and Objectives	Stakeholders and Challenges	Prognosis
Chesapeake Bay *11,600 km^2 estuarine bay*	*eutrophication, submerged aquatic vegetation loss, benthic community degradation*	*watershed nutrient inputs*	*Chesapeake Bay Program, 14 yr, US$ 27 million [http://www.chesapeakebay.net/index.aspx?menuitem=13853]*	*reduce pollutants from multiple sources and restore water quality, and restore wildlife habitat for fish, birds, crabs and mammals*	*full participation of most of the surrounding states, with considerable public investment*	*improvement in submerged aquatic vegetation, but water quality continues to be degraded and resiliency compromised*
Louisiana Coast *21,500 km^2 coastal zone of emergent marsh, forested swamps and bayous*	*wetland loss, salinity intrusion, coastal community flooding; lost over 4,800 km^2 of coastal wetlands since 1932*	*natural geologic subsidence, hurricanes, oil & gas canals*	*Coastal Wetlands Planning, Protection and Restoration Act, 11 yr, US$ 16 million? [http://lacoast.gov/new/About/Default.aspx]; Louisiana Coastal Area, 14 yr, US$ 2.0 billion [http://www.lca.gov/]; Louisiana Coastal Master Plan, 4 yr,US$ 36.6 million? [http://www.coastalmasterplan.la.gov/]*	*propose a series of projects that can reduce flooding risks and rebuild wetlands on a large scale, while also considering the diverse needs of communities*	*extensive community input; public resistance to shifting resources, such as oysters*	*highly vulnerable to accelerated sea level rise; exceedingly high price in current economic climate*
Puget Sound *4,000 km of inland sea - estuarine shoreline*	*degraded shoreline processes, wetland loss, depressed populations of nearshore-dependent species; nearshore ecosystems less diverse and simplified*	*beach shoreline and river delta development,*	*Puget Sound Nearshore Ecosystem Restoration Project, 9 yr, US$ 17.2 million [http://www.pugetsoundnearshore.org/]*	*assemble a portfolio of potential solutions to restore, protect and preserve nearshore ecosystems*	*adopted community-based restoration project proposals as initial project population;*	*initially implemented project unlikely to meet Project objectives*

Chesapeake Bay

Formed in 1983, the Chesapeake Bay Program (CBP) involves an agreement among most of the surrounding states and the District of Columbia to restore and protect Chesapeake Bay and its tidal tributaries. The initial goal was to reduce nutrients in the bay by 40% by the year 2000. Perhaps epitomizing the "return to Neverland conundrum (Duarte et al. 2009), despite substantial progress toward this goal, subsequent analysis has identified a need for even greater reductions to affect meaningful restoration of the system (Williams et al. 2010). The CBP has since expanded to include reducing excess sediments and toxics, and restoring important habitat areas and populations of valued organisms.

Louisiana Coast

In 1990, as a response to the alarming rate of 60 km^2yr^{-1} coastal wetland loss in Louisiana – a combined result of the natural subsidence and the interruption of natural deltaic sedimentation processes from diking and

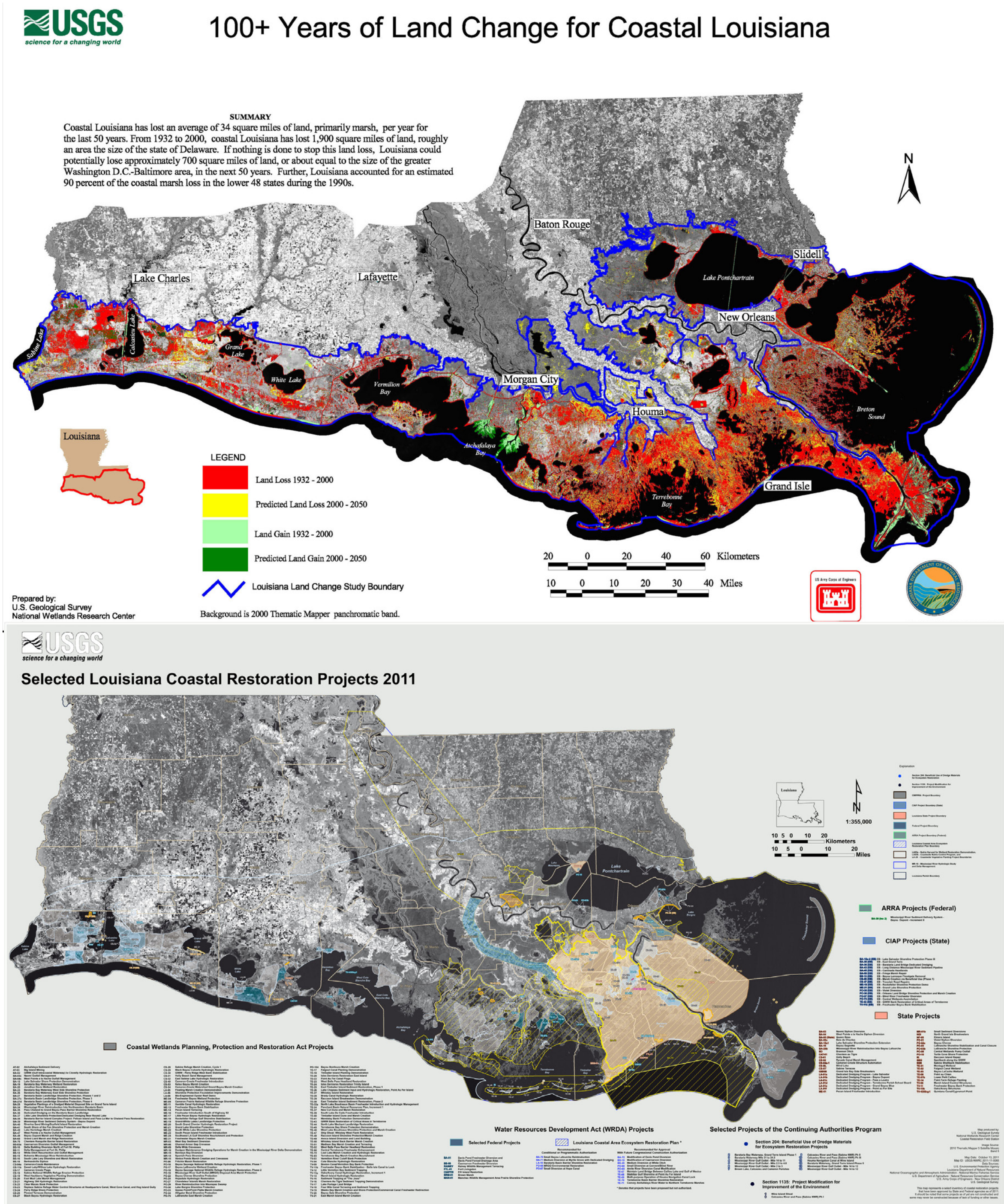

Figure 1: Maps of historic coastal land loss, projected land loss through 2050, and proposed Louisiana Coastal Master Plan restoration projects in the Louisiana coastal zone. Figures courtesy of U.S. Geological Survey, http://lacoast.gov/new/default.aspx).

channelization of the Mississippi River – Congress enacted the Coastal Wetlands, Planning, Protection and Restoration Act (CWPPRA), which funds wetlands enhancement projects and has contributed substantially to planning for large-scale restoration along the Louisiana coast. Since the initiation of CWPPRA, the Louisiana Coastal Area (LCA) and Louisiana Coastal Master Plan have emerged, and somewhat integrated, toward a comprehensive restoration goal. Recent planning of restoration projects illustrate the challenge of restoring wetlands and protecting coastal infrastructure in a dynamically changing landscape (Fig. 1).

Puget Sound: The Puget Sound Nearshore Ecosystem Restoration Project (PSNERP) is a General Investigation (GI) Feasibility Study managed by the U.S. Army Corps of Engineers and the State of Washington, represented by the Washington Department of Fish and Wildlife. PSNERP is completing a feasibility study to evaluate significant ecosystem degradation in the Puget Sound Basin; formulate, evaluate, and screen potential solutions to these problems; and to recommend a series of actions and projects. PSNERP has conducted comprehensive, spatially explicit analyses of nearshore change, the implications for degradation of nearshore ecosystem processes, and restoration and protection strategies to be implemented in the future (e.g., Fig. 2).

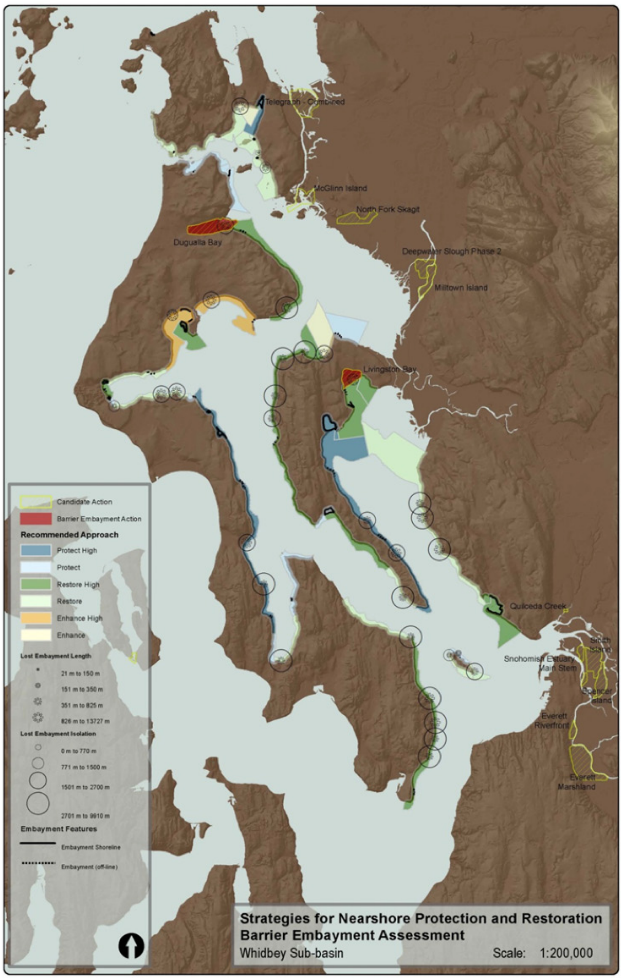

Figure 2: Whidbey Basin segment of Puget Sound indicating spatially-explicit restoration and protection strategies and proposed PSNERP restoration projects. Figure courtesy of PSNERP.

Lessons Learned

In synthesizing our collective lessons learned from involvement and exposure to these and other large-scale restoration programs in North America, we suggest that the following "lessons learned" may be valuable to carry forward into future initiatives in other regions.

1. Guidance principles of restoration: Compare/contrast nature of principles and how they are used
2. Ensure the "best available science" defines the difference between restoration success and failure: Distinguish what is 'knowable' and 'unknowable' expected of science and scientists in the process
3. Understand human dimensions affecting restoration: Assess stakeholder trust/acceptance of restoration science, and provide recognizable scenarios of the trade-offs
4. Restore ecosystem processes, not just ecosystem structure: Preserve natural disturbance regimes that account for complex and dynamic ecosystem structure and function, and recognize how restoration engages at the extremes
5. Incorporate landscape setting: Recognize that landscape change – the shifting mosaic – is characteristic of dynamically functioning ecosystems
6. Integrate restoration with protection: Consider how the rate and ultimate level of restoring functions often depends on integrity of adjacent ecosystems at local to watershed scales
7. Project future ecosystem change: Identify opportunities, in addition to constraints
8. Design meaningful measurements of restoration performance and "success": Try to integrate scientific, technical and social criteria, or at least elucidate trade-offs among them

Summary

Opportunistic, small-scale restoration is alluring and socially achievable, but cannot address recovery of impaired ecosystems at the landscape scale. A more synthetic, strategic approach is required for restoration of dynamic, interconnected ecosystems that will face future shifts in drivers across large, landscape scales. Comprehensive, science-based analysis can often achieve more than the narrow project objectives. However, the future of such comprehensive restoration programs may rest in how we evaluate their "success" and how invested stakeholders are in acknowledging the trade-offs across those scales. Promoting strategic restoration is not easy, especially when addressing ecosystem processes that involve landscapes across multiple ownership and jurisdictions. Incorporating science guidance and independent review from the beginning of restoration planning, that is transparent and open, will engage stakeholders. However, social investment is not likely to follow without better understanding of relationships of restoration to natural capitol and ecosystem goods and services. In the long term, institutional inertia can always marginalize even the strongest science-based restoration.

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Lakes assessment of ecological status: sensitivity and uncertainty of four biological quality elements along gradients of eutrophication and hydromorphological pressures

Anne Lyche Solheim[1], Laurence Carvalho[2], Geoff Phillips[3], Ute Mischke[4], Giuseppe Morabito[5], Gábor Borics[6], Birger Skjelbred[1], Marko Järvinen[7], Stina Drakare[8], Tiina Noges[9], Peeter Noges[9], Stephen Thackeray[2], Claire McDonald[2], Mike Dunbar[2], Agnieszka Kolada[10], Martin Söndergaard[11], Nigel Wilby[12], Seppo Hellsten[7], Bernard Dudley[2], Fraucke Ecke[8], Marit Mjelde[1], Martin Pusch[4], Ralph Clarke[13], Ken Irvine[14, 15], Angelo Solimini[16], Jukka Aroviita[7], Oliver Miler[4], Elaine McGoff[8], Jürgen Böhmer[17], Christine Argillier[18], Stephanie Pedron[18,20], Simon Caussé[18], Murielle Gevrey[18], Sandra Brucet[19], Kerstin Holmgren[8], Matthias Emmrich[4], Thomas Mehner[4], Erik Jeppesen[11], Torben L. Lauridsen[11], Julien De Bortoli[18], Ian Winfield[2], Pietro Volta[5], Atle Rustadbakken[1], Martti Rask[21], Jannicke Moe[1], Sandra Poikane[19], Christian Feld[22], Daniel Hering[22]

[1] Norwegian Institute for Water Research (NIVA), Oslo, Norway
[2] Centre for Ecology & Hydrology (CEH), UK.
[3] Environment Agency. Reading, UK
[4] Leibniz-Institute of Freshwater Ecology and Inland Fisheries (IGB), Berlin, Germany
[5] CNR – Institute for Ecosystem Study, Verbania-Pallanza, Italy
[6] Hungarian Academy of Sciences, BLRI, Department of Tisza River Research, Debrecen, Hungary
[7] Finnish Environment Institute (SYKE), Finland
[8] Swedish University of Agricultural Sciences, Sweden
[9] Centre for Limnology, Estonian University of Life Sciences, Estonia
[10] Institute for Environmental Protection, Warsaw, Poland
[11] Aarhus University, Department of Bioscience, Silkeborg, Denmark
[12] School of Natural Sciences, University of Stirling, Stirling, UK
[13] Conservation Ecology and Environmental Sciences (CEES), Bournemouth University, Dorset, U.K.
[14] Trinity College, Dublin, Ireland
[15] UNESCO-IHE Institute of Water Education, Delft, Netherlands
[16] Department of Public Health and Infectious Diseases, Sapienza University of Rome, Italy
[17] Bioforum Gmbh, Kirchheim/Teck, Germany
[18] Cemagref, now Irstea - National Research Institute of Science and Technology for Environment and Agriculture , France
[19] Joint Research Centre, EU Commission, Institute for Environment and Sustainability, Ispra, Italy
[20] Seine Normandy Water Agency, France
[21] Finnish Game and Fisheries Research Institute, FI-16970 Evo, Finland
[22] University of Duisburg and Essen, Germany

Email address of corresponding author: als@niva.no

Key words: dose-response, uncertainty, metrics, phytoplankton, macrophytes, benthic fauna, fish, eutrophication, hydro-morphological alterations

Objectives

The main objectives for the WISER work on lakes have been to identify the best metrics for assessing the ecological status of European lakes exposed to eutrophication and hydro-morphological pressures, according to the requirements of the Water Framework Directive. Candidate metrics and new indices for the biological quality elements (BQE) phytoplankton, macrophytes, benthic invertebrates and fish were tested to find those that are most sensitive to pressure and those that have the least within lake variability at sites with the same level of pressure. The best metrics were requested to be used as common metrics for the intercalibration exercise of lake assessment tools. Common metrics used for intercalibration have to be well correlated with pressure and with the national assessment systems, as well as to cover all relevant parameters indicative of the BQE. Common metrics should also show only minor bias due to biogeographical differences. Common metrics can also be used as national metrics in cases where national methods are missing or are of poor quality.

Data and methods

Sensitivity of various BQE metrics to pressure has been assessed from regression analyses of dose-response curves along pressure gradients using large scale pan-European datasets with taxonomic and/or abundance data, as well as pressure related data and lake-type data from > 7000 lakes from 21 countries (table 1). Sensitivity of macroinvertebrate metrics to hydro-morphological modification of lake margins was assessed on newly collected WISER data by comparing samples from unmodified (U), soft (S, e.g. riparian clear-cutting, recreational beaches) and hard (H, e.g. retaining walls, rip-rap) modified margins within each of 51 lakes, which were sampled on 9 sites per lake representing a full pressure range. In-lake variability of the other BQE metrics has been assessed from new WISER data sampled from 20-30 lakes in 2009-2010 (table 1a and figure 1).

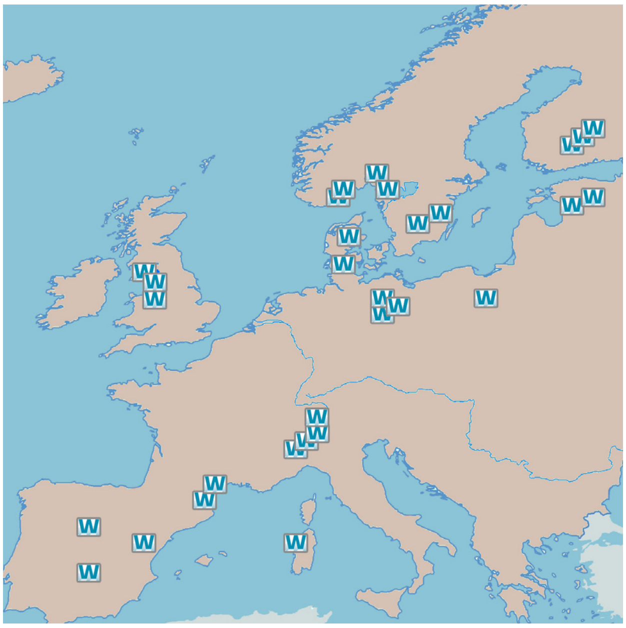

Figure 1. Location of lakes sampled for all four BQEs in 2009-2010. See table 1 for more info

Results on BQE sensitivity to dominant pressures

To assess ecological effects of eutrophication, phytoplankton is clearly the most sensitive biological quality element (table 2). The best metrics for this BQE are the Phytoplankton Trophic Index (r^2 = 0.67), as well as chlorophyll a (r^2 = 0.63, for lakes with TP<100µg/l). These two metrics have been combined to a common metric for intercalibration of phytoplankton methods with successful results in both the Northern GIG (Geographical Intercalibration Group) and the

Table 1. Overview of number of lakes (water bodies) per WISER work-package (WPs) or biological quality element (BQE).
a) WISER foreground data showing lakes sampled in 2009 or 2010

WP	BQE	Countries	# Water-bodies
3.1	*Phytoplankton*	*DE, DK, EE, ES, FI, FR, IT, NO, PL, SE, UK*	*32*
3.2	*Macrophytes*	*DE, DK, EE, FI, FR, IT, NO, PL, SE, UK*	*28*
3.3	*Macro-invertebrates*	*DE, DK, EE, FI, IE, IT, SE, UK*	*51*
3.4	*Fish*	*DK, FI, FR, DE, IE, IT, NO, SE, UK**	*26 (14)**

** 26 lakes sampled, but only 14 were compiled for uncertainty analyses*

b) WISER background data including all existing data compiled from many different datasets.

WP	BQE	Countries	# Water-bodies
3.1	*Phytoplankton*	*BE, CY, DE, DK, EE, ES, FI, FR, GR, HU, IE, IT, LT, LV, NL, NO, PL, PT, RO, SE, UK*	*6 927**
3.2	*Macrophytes*	*BE, EE, FI, IE, LT, LV, NL, NO, PL, RO, SE, UK*	*1 575*
3.3	*Macro-invertebrates*	*BE, DE, EE, GB, LT, LV, NL, PL*	*227*
3.4	*Fish*	*DE, DK, EE, ES, FI, FR, IE, IT, LV, LT, NO, PT, RO, SI, SE, UK*	*2 175*

**taxonomic data available from ca. 1300 water bodies*

Central-Baltic GIG. For macrophytes, the best metric for eutrophication pressure is the intercalibration common metric for taxonomic composition (ICM) (r^2 = 0.52), which is based on empirical data and has been used for intercalibrating macrophyte methods in the Northern GIG. The expert-based Ellenberg index also performs well but, in contrast to the ICM, is almost insensitive to TP when >80ug/L. In terms of hydromorphological pressure, we have tested the impacts of water level fluctuations on macrophyte taxonomic composition in regulated lakes in the Northern countries. The macrophytes water level fluctuation index (Wlc) has high r^2 (0.77) and clear threshold response for indicator taxa e.g. Isoetes at a Wlc value of -20, corresponding to ca. 3.5 m water level fluctuations. Thus, this metric is a very promising tool to set true biological boundaries for good ecological potential for heavily modified water bodies.

For littoral benthic invertebrates the best metric for eutrophication assessment is a multimetric index for

Central European lakes consisting of several single metrics, including the number of taxa of mayflies, stoneflies, caddisflies, water beetles, mussels, dragon-flies (EPTCBO), average score per taxon (ASPT), % abundance of mayflies, caddisflies and dragon-flies (%ETO), % abundance of taxa on stony substrates (all % in relation to abundance classes). This multimetric has a correlation with Total P (r^2 = 0.40), when applied to whole lakes, which is less good than the best metrics found for phytoplankton and macrophytes (table 2) response to eutrophication. However, this multimetric index has a better correlation with combined pressures including morphological shore-line modifications, land-use and TP (r^2 = 0,53) and has been used for intercalibration of national macroinvertebrate methods for the Central-Baltic GIG. Evenness in macroinvertebrate communities also responds to eutrophication, declining with TP overall and within countries, although the correlation is not very strong (r^2 = 0.17). Another multimetric index has also been developed to assess macroinvertebrates specific response to morphological alterations in the form of shore-line modifications. This multimetric also includes several single metrics, such as the number of taxa of mayflies, stoneflies, caddisflies, water beetles, mussels, dragon-flies (EPTCBO), % abundance classes of gatherer/collectors, % abundance classes of chironomids, and Margalef diversity. This multimetric index has an r^2 = 0.49 against a pressure index representing the degree of morphological lakeshore modification. Macroinvertebrate taxonomic richness (all families or EPTCBO) and percentage individuals preferring particulate organic matter (%POM) were lower at both soft and hard modified lake margins than at unmodified margins in 64% of 44 lakes. Further improvement of multimetrics for benthic fauna response to shore-line modifications is needed through assessment based on

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.

BQE	Metric	Metric description	Pressure	r^2	GIG or country	p	N
Phytoplankton	*Chl-a*	*Chlorophyll a (µg/l)*	*Eutrophication (Total-P)*	*0.63*	*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*
	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*
	Cyano bloom intensity	*Cyanobacteria biovolume (mg/l)*	*Eutrophication (Total-P)*	*0.34 (GAM)*	*All, but mainly NGIG & CBGIG*	*<0.001*	*1710 (1010 NGIG, 602 CBGIG)*
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)*	* 0.10 0.29 0.52*	*All, but mainly NGIG & CBGIG*	* <0.001 <0.001 <0.001*	* 640 908 652*

BQE	Metric	Metric description	Pressure	r^2	GIG or country	p	N
	WIc	*Water level Taxonomic comp index*	*Hydro-morphological 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*	*?*	*161*
	MMI	*Multimetric Index*	*Morphological alterations and Eutrophication (shore line modifications, landuse in lake surroundings and TP)*	*0.53*	*CB-GIG*	*<0. 001*	*161*
	MMI	*Multimetric Index*	*Morphological alterations (shore line modifications)*	*0.49*	*All, mainly CBGIG*	*??*	*44*
	MMI	*Multimetric Index*	*Morphological changes (shore line modifications)*	*-0.70** *-0.49** *-0,37** *-0.50**	*DE+DK* *Italy* *SE+FI* *IE+UK*		
	Evenness	*Evenness of taxa abundances*	*Eutrophication (Total-P)*	*0.17*	*All, but mainly NGIG & CBGIG*	*0.005*	*51*
	NTaxa	*Number of taxa*	*Morphological changes (shore line modifications)*	*0.12*	*All, but mainly NGIG & CBGIG*	*<0.001*	*44*
	EPT CBO	*Number of EPTCBO taxa*	*Morphological changes (shore line modifications)*	*0.11*	*All, but mainly NGIG & CBGIG*	*<0.001*	*44*
	%POM	*% individuals preferring particulate organic matter 'habitat'*	*Morphological changes (shore line modifications)*	*0.08*	*All, but mainly NGIG & CBGIG*	*<0.001*	*44*
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)* *(Total-P)*	*0.16* *0.18*	*All* *All*	*<0.001* *<0.001*	*445* *445*

**Based on Spearman correlation Rho from composite samples; different metrics correlated best with the stressor index in four biogeographically distinct regions.*

habitat-specific sampling, and by approaches to account for existing biogeographical differences.

For fish, the best correlation with pressure (non-natural land cover) was obtained with a multimetric index (r^2=0.25) composed of three metrics: biomass per unit effort (BPUE), catch per unit effort (CPUE) and relative number of omnivorous individuals (OMNI). This multimetric index has not been used for intercalibration of national methods. Further work is needed to develop a common multimetric index for fish response to shore-line modifications, as the current regression analyses are not satisfactory.

Results on BQE metric variability at the same level of pressure

Within-lake variability caused by natural spatial variation, as well as variability related to sampling and sample processing was low for phytoplankton (table 3), although this BQE probably has much higher temporal variability related to sampling frequency (tbc). If excluding temporal variability, the most precise phytoplankton metrics having the lowest within-lake variance are chlorophyll, Cyanobacteria biomass and the taxonomic composition index PTI. The most important variance component for these metrics is the sub-sampling. Still, the error caused by sub-sampling is small, as the total within-lake variance is so low for these metrics (ca. 5-10%). For phytoplankton, the most important variance component is probably the seasonal variability. The uncertainty in the growing season mean value of the metrics which are used for assessing ecological status can be reduced by increasing the sampling frequency, or by applying a more standardised sampling period (May - September).

For lake macrophytes, metric variability averaged 25-30% with station as the major variance component. Thus, for macrophyte metrics it is necessary to sample several stations or increase station area to reduce uncertainty in ecological status assessment. 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. For fish the major variance components were depth stratum (numbers), referring to benthic gill-nets set in successive 3-m depth zones, and variability between individual nets (biomass), reflecting substantial spatial heterogeneity of fish distribution, caused by abiotic factors such as oxygen, temperature and light. This result may explain why dose-response curves of many fish metrics with pressure variables predicted a relatively low proportion of variance when compared with other BQEs.

Conclusions

Table 3. 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.

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. Further results on temporal variation in WISER lakes with time-series and/or different sampling frequencies will be included as soon as they become available.*
*** includes within-lake variance of 8-12% due to margin modification type (U,S,H)*

In conclusion, the botanical BQEs (phytoplankton and macrophytes) respond more clearly to eutrophication pressure than the zoological BQEs (benthic invertebrates and fish), and are thus the most sensitive BQEs for assessing lake eutrophication impacts. These botanical BQEs also exhibit less intra-lake metric variability and thus together provide the best BQEs to assess ecological status of lakes subject to eutrophication. Seasonal variation for phytoplankton is however important and requires regular sampling to allow precise assessment of ecological status. For the zoological BQEs other confounding factors cause a noisier dose-response relationship, e.g. mixture of response to increased productivity and oxygen-depletion, more intensive top-down control of macroinvertebrates (fish predation), human impacts of fishing and stocking of fish.

For hydromorphological pressure, the macrophytes seem promising in terms of response to water level fluctuations in ice-covered lakes used for hydropower production, and may thus be used as a tool to set boundaries for good ecological potential in heavily modified lake/reservoir water bodies, although this needs further evidence from other biogeographic regions. Recent results for macroinvertebrates show clear responses to morphological alterations of lakes and changes in habitat composition (submerged habitats as well as shore vegetation structure) and may thus enable assessment of the ecological effects of the second most important human pressure to European lakes. For fish, the impact of hydromorphological pressures has not been demonstrated with the datasets available in WISER. Fish may still be sensitive to this pressure, for example littoral fish species in regulated lakes, but other data are needed to test this.

Sampling and sample processing as sources of uncertainty in lake phytoplankton community metrics

Stephen J. Thackeray[1], Peeter Nõges[2], Michael Dunbar[3], Bernard J. Dudley[4], Birger Skjelbred[5], Giuseppe Morabito[6], Laurence Carvalho[4], Geoff Phillips[7], Ute Mischke[8], Jordi Catalan[9], Caridad de Hoyos[10], Christophe Laplace[11], Martina Austoni[6], Tomasa Virdis[12], Kairi Maileht[2], Agnieszka Pasztaleniec[13], Marko Jarvinen[14], Stina Drakare[15] & Anne Lyche Solheim[5]

[1] Centre for Ecology & Hydrology, Lancaster Environment Centre, Library Avenue, Bailrigg, Lancaster, LA1 4AP, UK
[2] Centre for Limnology, Institute of Agricultural and Environmental Sciences, Estonian University of Life Sciences, 61117 Rannu, Tartumaa, Estonia
[3] Centre for Ecology & Hydrology, Maclean Building, Benson Lane, Crowmarsh Gifford, Wallingford, Oxfordshire, OX10 8BB, UK
[4] Centre for Ecology & Hydrology, Bush Estate, Penicuik, Midlothian, EH26 0QB, UK
[5] Norwegian Institute for Water Research (NIVA), Oslo, Norway
[6] CNR Istituto per lo Studio degli Ecosistemi, Largo V. Tonolli 50, 28922 Verbania Pallanza, Italy
[7] Environment Agency, Cobham Road, Ipswich, Suffolk, IP3 9JD, UK
[8] Leibniz Institute of Freshwater Ecology and Inland Fisheries, Justus-von-Liebig-Straße 2, 12489, Berlin, Germany
[9] Center for Advanced Studies of Blanes (CEAB-CSIC), Accés Cala St. Francesc 14, Blanes 17300, Spain
[10] Centro de Estudios Hidrográficos del CEDEX, PO Bajo de la Virgen del Puerto, 3 28005, Madrid, Spain
[11] Research Institute for Agricultural and Environmental Engineering, CEMAGREF, av de Verdun 50, 33612 Cestas-Gazinet, France
[12] University of Sassari, Piazza d'Armi, 17, 07100 Sassari, Italy
[13] Institute of Environmental Protection, 01-692 Warszawa, Kolektorska 4, Poland
[14] Finnish Environment Institute (SYKE), Jyväskyla Office, P.O. Box 35, 40014 Jyväskyla, Finland
[15] Institutionen för vatten och miljö, Box 7050, Lennart Hjelms väg 9, 750 07 Uppsala, Sweden

Email address of corresponding author: sjtr@ceh.ac.uk

Key words: ecological quality assessment, eutrophication, chlorophyll, species composition

Abstract

The EU Water Framework Directive (WFD) states that attributes of biological communities should be used to assess the ecological status of fresh- and coastal/transitional waters. For lakes, the phytoplankton is a key biological community to be used for this purpose. It is therefore necessary to develop metrics that describe high-level properties of phytoplankton communities and that are sensitive to environmental pressures, such as nutrient enrichment.

Assessment of the utility of such metrics demands a knowledge of the extent to which they can be affected by sampling and sample processing procedures e.g. where samples are collected from and who processes the samples. If metrics vary more with differences in sampling method and sample processing within a lake than they do among lakes of varying pressure, then they are unlikely to provide a sensitive means of describing differences in the biological impacts of an environmental pressure among lakes.

We have analysed the results of a multi-scale field campaign of 32 European lakes, to resolve the extent to which seven proposed phytoplankton metrics vary among lakes and with sampling/sample processing. For all seven metrics, between 65% and 96% of the variance in metric scores was due to variability between lakes. Differences in locations around a lake, or sampling and analytical variability, only accounted for a small proportion of the variability in metric scores. These results are especially true for three candidate phytoplankton metrics being considered for Intercalibration: chlorophyll (abundance metric), PTI (composition metric), and cyanobacteria abundance (bloom metric). For these three metrics, >85% of the variability in metric scores was attributed between lakes and total phosphorus concentration was the best single predictor of this between lake variation. Although much between-lake metric variation still remained unexplained by the

available environmental data, we conclude that these three proposed metrics are sufficiently robust metrics for ecological status assessment and are suitable for adoption by in the Intercalibration process or as metrics by Member States. The relatively small contributions of analyst and sub-sample level variation to the total indicates that standardisation of sample mixing and sedimentation protocols, as well as of taxonomic procedures, can help minimise sampling and analytical variability and help make more meaningful comparisons of ecological status among different lakes.

Combination of Biological Quality Elements towards complete water body assessment

Wouter van de Bund, Rossana Caroni

European Commission, Joint Research Centre, Institute for Environment and Sustainability, Ispra, Italy

Email address of corresponding author: wouter.van-de-bund@jrc.ec.europa.eu

Introduction

For the assessment of the ecological status of a water body the WFD requires that several biological quality elements (BQEs) are taken into account. Since the ecological status classes are set and intercalibrated at BQE level it is very important for the comparability of the final classification results that all countries apply similar approaches to combine BQE results into a complete water body assessment. According to the CIS guidance document on classification, ecological the lowest class of all relevant BQEs determines the ecological status of a water body (the "one out – all out" principle). There are no specific requirements on how to combine different metrics used within a BQE; this can be done using "one out – all out", but averaging or other approaches are also acceptable.

Differences in approaches between countries

In spite clarity of the guidance, a variety of approaches for combining BQE results is applied in different European countries, varying from a strict application of the "one out – all out" principle to more pragmatic approaches often involving the application of expert judgment, with potentially serious consequences for the comparability of the final classification results. Also, the number of BQEs used varies between countries.

Important factors that influence the final classification

Obviously, the "one out – all out" approach always gives a lower classification than averaging. Using both simulated data and real monitoring data, it can be clearly demonstrated that the following factors have a strong influence on the final classification:

Even a single BQE having a high level of uncertainty will strongly affect the reliability of the final classifications using the "one out – all out" approach. This can be remedied by improving the accuracy of the methods, or to exclude methods with high uncertainty.

The higher the number of BQEs, the larger the negative bias in the classification outcome for the "one out – all out" approach. This can especially be problematic if all BQEs address the same pressure and the effect is proportional to the level of uncertainty associated with the individual BQEs. Averaging may give a more reliable result in such a case

If the different BQEs address different pressures, averaging results in a positive bias in the classification; here the "one out – all out" approach gives more correct results, provided that the uncertainty associated with the individual BQEs is not too high

If within the different BQEs metrics are included that are sensitive to different pressures (e.g eutrophication and acidification) it is better to group them by pressure than by BQE. This will improve the reliability of the final assessment (even if this is not consistent with the recommendations of the classification guidance).

Conclusions and recommendations

The "one out – all out" approach only gives acceptable and comparable results if the different BQEs are complementary (showing the effects of different pressures, showing effects on different temporal and/or spatial scales, showing effects on different aspects of ecosystem functioning). Also, the level of uncertainty in the classification should not be too high and not too different between BQEs.

One has to avoid the blind application of the "one out – al out " rule if those conditions are not met. It is recommended to avoid redundancy between BQEs with regard to the pressures they are responding to, and to use the possibility given by the WFD to exclude BQEs that are too variable wherever necessary. There are good examples where expert judgment is used to avoid the pitfalls mentioned, to ensure that classification is based on the most reliable information available – but the disadvantage of such approaches is that there is always a certain level of subjectivity in the assessment. Such subjectivity can be reduced by decreasing the redundancy and increasing the accuracy of the methods for the BQEs.

Comparison of recovery processes in rivers, lakes and estuarine and coastal waters

Piet F.M. Verdonschot[1], Hanneke E. Keizer-Vlek, Bryan Spears, Sandra Brucet, Christian K. Feld

[1] Alterra Wageningen UR, Department of Freshwater Ecology, P.O. Box 47, 6700 AA Wageningen; phone: +31 (0)317486552

Email address of corresponding author: piet.verdonschotr@wur.nl

Key words: Rivers, streams, lakes, coastal and estuarine waters, recovery, restoration, biological quality elements, biological interactions, climate change

Introduction

Catchment wide integrated basin management requires knowledge on cause-effect and recovery chains within water bodies as well as on the interactions between water bodies and categories. In the WISER WP6.4 recovery processes in rivers, lakes and estuarine and coastal waters were evaluated. The major objectives were:

– To analyse and compare (cause-effect and) recovery chains within water categories based on processes and structural and functional features.

– To detect commonalities among different chains in different water categories. Thus, to compare recovery chains between water categories.

– To link recovery chains to over-arching biological processes and global change.

– To develop a method to combine recovery effects in a summarising 'catchment' metric.

The main stressors studied to reach these objectives were acidification, eutrophication and hydromorphological changes.

Methods

To compare recovery-chains within water bodies and between water categories information was extracted from published reports and peer-reviewed papers. Apart from a variety of about 20 major reviews, three major sources of information were included. For rivers 370 papers were reviewed and 168 papers were analysed by Feld et al. (2010). For lakes 302 lake-equivalent recovery case studies for which eutrophication was the major stressor were analysed in detail (Spears et al. 2010). For estuarine and coastal waters the review of 51 studies by Borja et al. (2010) was the major information source.

Results on Recovery

Degradation

Rivers integrate the adverse effects of various activities on land and are, therefore, often simultaneously affected by multiple stressors arising from agriculture, deforestation, urbanization, storm water treatment, flow regulation and water abstraction (Palmer et al. 2010). Globally, lake ecosystems are mainly being affected by eutrophication (intensive agricultural land use) and physical habitat modification of their shoreline, while estuaries and wetlands constitute the ultimate sink for nutrients and other sources of pollution and contaminants originating from entire river basins. Furthermore, many estuarine and coastal waters are being physically modified, for instance, for flood protection purposes and navigation. The conceptual models (DPSIR-chains) of the different water categories are hard to compare. Striking is the difference in the level of detail between rivers and lakes (high) on the one hand and the marine ecosystems (low) on the other. This difference probably has to do with the scale of degradation in rivers and lakes, where it is easier to find/deduct pathways of ecosystem response.

Recovery Concepts

The Driver-Pressure-State-Impact-Response-Recovery (DPSIRR) scheme provides a framework to link socio-economy with ecology. Literature was searched for existing DPSIRR-chains for the three water categories. Such conceptual models on the recovery of river, lake and estuarine and coastal ecosystems were scarce and fragmented. Such models lacked for the marine systems were quite one-sided, focusing on eutrophication, for lakes and quite specific for certain measures in rivers. Comparison and integration of DPSIRR-chains is up date impossible.

Recovery Measures

In rivers most measures target the morphology of the stream stretch or the instream habitats. Few only are related to reduction of nutrient input. On the contrary, in lakes all measures target to reduce nutrient levels, especially phosphate. Others mainly focus on acidification. Measures are not often taken directly in estuarine and coastal waters, these much more relate to measures taken inland through legislation on nutrient reduction. These observations supported our initial hypothesis that "at a catchment scale, nutrient stress affecting functional (production/decomposition) processes will be more important in lakes and marine systems, while hydromorphological stress affecting habitat availability will be more important in rivers".

Recovery: Data availability and processing

In rivers and lakes quite an amount of monitoring data are available. In estuarine and coastal waters such data are scarce. Despite the number of monitored recovery cases, each one seems to stand alone as monitoring schemes were set-up for local situations and to answer partial questions. Furthermore, in many, many cases data on recovery just lack and this is quite alarming! Not only is the amount of available data surprisingly low, the composition of the available data is often very limited and does not allow the evaluation and generalisations of improvements and eventually of successes. The huge investments in recovery of surface waters require control of the ecological effects. Therefore, restoration monitoring should become mandatory. Only by frequent monitoring of biological and abiotic changes after restoration will restoration practitioners and scientist be able to evaluate the success of the restoration measure and eventually of the investment done.

Recovery: Organism groups

The majority of restoration studies in rivers and in estuarine and coastal ecosystems have focused on macroinvertebrates. In rivers also fish are important indicators. In lakes phytoplankton is the BQE studied most extensively. The difference in indicator groups used goes back to the causes of degradation. In lakes eutrophication is most important and phytoplankton best reflects the nutrient status of the lake over time. In rivers most degradation goes with hydromorphological change. Macroinvertebrates and fish respond strongly tot these types of changes. The choice of macroinvertebrates as indicators of degradation in estuarine and coastal waters is less obvious as eutrophication and organic load are most common causes of degradation along with bottom disturbances. The latter would best be reflected in macroinvertebrate responses the first less. The confounding factor in estuarine and coastal waters for phytoplankton is water movement. Water movement reduces the indicative value of phytoplankton.

Recovery: Time-scale

Although, analyses in the different reviews do not address full recovery', authors do give indications on 'full recovery' based on estimates. Marine ecosystems may take between 35 and 50 years to recover. Recovery after weir removal may take as long as 80 years. Recovery after riparian buffer installment may take at least 30-40 years. Despite the fact that they do not indicate ,'full recovery' we compared recovery times between the three water categories as mentioned in the different reviews. In marine ecosystems benthic invertebrates and macrophytes have the potential to recover within months (in two studies on recovery of sediment disposal) and fish within one year. When only marine studies that recover from eutrophication are included, recovery times for macroinvertebrates varied between >3 years and >6 years. Although in some cases recovery can take <5 years, especially for the short-lived and high-turnover biological components, full recovery of estuarine and coastal ecosystems from over a century of degradation can take a minimum of 15–25 years for attainment of the original biotic composition, diversity and complete functioning may lag far beyond that period. In lakes recovery time from eutrophication for macroinvertebrates varied between 10 and 20 years. As in marine ecosystems recovery of macrophytes (2 to >40 years) and fish in lakes (2 to >10 years) be relatively fast. Response times for organism groups in rivers are lacking, because the literature rarely includes post hoc monitoring of more than 5 years. Also, the fact if biological response in rivers occurs within short term is undecided. The potential benefits of most in-stream structures will be short-lived (<10 years) unless coupled with riparian planting or other process-based restoration activities supporting long-term recovery of key ecological and physical processes.

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. Since eutrophication is considered to be the most important pressure in rivers and lakes, only this is not addressed in rivers, this might be a major cause. Especially, the response of macroinvertebrates in rivers is questionable, some studies mention recovery times of others question recovery of macroinvertebrates completely. In lakes internal nutrient loading often delays recovery.

Recovery: Failure or delay in response

Several major reason return in many publications on recovery failure or delay:
– spatial scale: must be large enough (catchment),
– temporal scale: there is time needed for recovery,
– multistressors: mostly only one or a few stressor were tackled, others forgotten,
– confounding abiotic processes affect recovery, such as internal P loading, and biological interactions, like the early arrival of non-native species,
– distance from source populations and lack of connectivity results in dispersal limitations and colonisation barriers.

Recovery: Shifting baselines

It is difficult to judge whether the concept of shifting baselines is part of the reality of ecosystems developments as proof is hard to find. Even in the coastal and estuarine examples it is questionable whether the responses are due to alternative states or due to overlooked other stressors. Often in many lake examples the latter is the case.

Recovery: Effects of biological interactions

Restoring the appropriate habitat is still the main component of aquatic ecosystem restoration efforts. Although the importance of establishing the suitable abiotic conditions is stressed by a multitude of studies, the awareness that other factors should be considered as well is apparent in recent recommendations on freshwater restoration. There are several, more or less connected issues that are repeatedly stressed in a multitude of studies:
– Incorporating the spatial and temporal scale (i.e. maximum and minimum) of the habitat and the connectivity between the various habitat patches, including both abiotic and biotic components;
– Incorporating the knowledge of source populations and dispersal ability or constraints in predicting restoration outcome. However few studies attempt to match this ecological background with empirical data.
– Incorporating mitigating measures to prevent non-native species to colonise and set priority effects.

Recovery: Impacts of climate and global change

A range of biological management practices (especially fishery management) and extreme weather events were identified as key factors that were responsible for slowing down or contradicting recovery processes. Alterations in nutrient concentrations and biogeochemical cycling at the sediment-water interface, following nutrient management, can influence the magnitude and timing of nutrient delivery to downstream ecosystems. This phenomenon is likely to be highly sensitive to changes in local weather conditions associated with climate change.

Research gaps

In summary, there is need for the following research efforts;
– Need for statistical understanding of ecological responses.
– Need for more comprehensive and long-term monitoring to underpin quantitative assessment of management measures.
– Need to quantitatively assess cause-effect relationships during the recovery process.
– Need for case studies relevant to WFD targets.
– Need for specific knowledge on certain BQEs in certain water categories.
– Need for knowledge on maintenance, and recurring management.
– Need for knowledge on the most important factor(s) for recovery and their interactions.
– Need for knowledge on shifting baselines and thresholds.

Conclusions

Restoration ecology is just in its infancy. The huge amount of literature evaluated brings up one major conclusion. Restoration is a site, time and organism group specific activity. Generalisations on recovery processes are up to date hard to make. Despite the multitude of studies that provided theoretical frameworks, guidelines, research needs and issues that are important for freshwater restoration, only few studies provide evidence of how this ecological knowledge might enhance restoration success.

Goals of restoration projects typically encompass a multitude of objectives (species groups, ecological, cultural and landscape values) and a multitude of measures. Thus, evaluation of the response of a single factor to a single measure tends to be difficult.

Another major bottleneck is the lack of sufficient monitoring allowing for insufficient learning from both successful and unsuccessful restorations. However, the frequently occurring general recommendation in proposed guidelines for restoration projects, including appropriate monitoring and publishing of the results, could help to gain insight into the processes important to successful restoration.

Another problem is related to the many detected effects that occur only in the short-term and at the local (site) scale, which raises the question of appropriate scaling for restoration. There is not yet evidence for the most appropriate spatial nor temporal scale, but several extended review studies supported the hypothesis that the local scale is inappropriate to achieve long-term measurable improvements.

Guidelines for standardisation of hydroacoustic methods

Ian J. Winfield[1], Matthias Emmrich[2], Jean Guillard[3], Thomas Mehner[2] and Atle Rustadbakken[4]

[1] Centre for Ecology & Hydrology, Lancaster Environment Centre, Library Avenue, Bailrigg, Lancaster, Lancashire LA1 4AP, U.K.
[2] Department of Biology and Ecology of Fishes, Leibniz-Institute of Freshwater Ecology and Inland Fisheries, Műggelseedamm 310, 12587 Berlin, Germany
[3] Institut National de la Recherche Agronomique, UMR Carrtel, BP 511, 74203 Thonon-les-Bains, France
[4] Norwegian Institute for Water Research, Gaustadalléen 21, NO-0349 Oslo, Norway
Email address of corresponding author: ijw@ceh.ac.uk

Key words: lake, fish, population, survey, sampling, hydroacoustics, echo sounding, sonar, standardisation

Introduction

The use of transmitted underwater sound to survey fish populations (known effectively interchangeably as hydroacoustics, echo sounding or sonar) has a long and extensive record of successful applications in the marine environment where most of its major developments have historically taken place. The very efficient transmission of sound in water, particularly when compared with that of light, makes this remote-sensing technique highly effective in most aquatic ecosystems and under many environmental conditions. As a result, it provides a valuable complement to capture-based and frequently destructive fish sampling techniques. In recent decades, technological developments, including the miniaturisation of electronic components and rapidly increasing computing power, have facilitated the production of hydroacoustic systems which can be readily deployed from small vessels on fresh waters including lakes and reservoirs.

WISER Deliverable 3.4-3 'Guidelines for standardisation of hydroacoustic methods'

WISER Deliverable 3.4-3 'Guidelines for standardisation of hydroacoustic methods' has been written for an audience with no or little previous knowledge of hydroacoustics. It gives an introduction to this still developing field, together with a set of guidelines specifically for the application of hydroacoustics to the investigation of fish populations in European standing freshwater bodies. In addition to explaining the basic principles of hydroacoustics and reviewing appropriate hardware and software currently available, detailed guidance is given on conducting hydroacoustic surveys on lakes and reservoirs.

Basic principles of hydroacoustics

The basic principles behind the use of hydroacoustics for the investigation of fish populations are relatively simple. Essentially, an instrument called an echo sounder is used to transmit a pulse of sound into the water column where it spreads much like the pattern of light spreading from a hand torch. The sound travels at a speed of approximately 1500 m s-1, with its exact speed in fresh waters depending primarily on temperature. The sound may be directed vertically or horizontally and effectively insonifies a cone of water with each pulse.

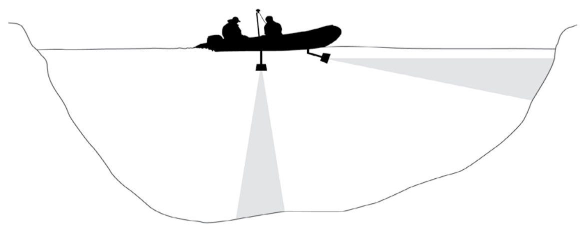

Figure 1: Vertically and horizontally orientated transducers as deployed in a hydroacoustic survey of a lake or reservoir. Sound beams are shown as shaded triangles, although in reality they each approximate to a cone.

When this wave meets an object (usually referred to as a target) of density different to that of the water, it is reflected, again spreading like light from a hand torch, and a component of this reflected sound reaches the echo sounder where it is recorded as an echo. For each pulse of sound, the echo sounder records the time taken for the echo to return (which using the speed of sound can be readily converted to target distance), its strength and, in most systems currently in use, its direction relative to the echo sounder.

A modern echo sounder is usually composed of three basic components, i.e. a surface unit which essentially generates electrical instructions for the production of sound, a laptop computer which controls the surface unit, records data, and provides real-time information in the form of an echogram and display of other data, and an underwater component called a transducer which converts the electrical instructions into a sound wave and then detects returning echoes and converts them back into an electrical signal which is then further processed by the system. In addition, a Global Positioning System (GPS) unit is usually connected to the

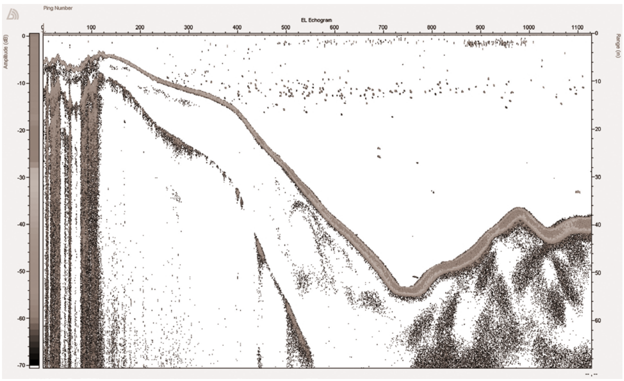

Figure 2: An example echogram produced along part of one transect during a night-time vertical survey of a deep lake. The horizontal axis represents time elapsed (equivalent to horizontal displacement in a mobile survey) while the vertical axis, labelled on the right of the Fig., represents range from the transducer (effectively water depth in a vertical survey) running here from 0 m at the top of the Fig. to 70 m at its bottom.]

echo sounder to record location information. On lakes and reservoirs, hydroacoustic data are usually collected effectively continuously from a boat moving along a number of contiguous or spaced pre-planned transects. The data then require substantial post-survey processing in order to produce information on fish abundance, distribution and other features. There are three main methods for such analysis, i.e. echo counting, trace counting and echo integration, each with relative strengths and weaknesses. These basic principles encompass a large degree of technical complexity at all steps of the hydroacoustic process. Fortunately, knowledge of all these details is not essential for the successful application of the technique.

Equipment hardware, software and training

In terms of equipment suitable for deployment on lakes and reservoirs, the European market is dominated by BioSonics (BioSonics Inc., U.S.A., www.biosonicsinc.com), HTI (Hydroacoustic Technology Inc., U.S.A., www.htisonar.com) and Simrad (Simrad Kongsberg Maritime AS, Norway, www.simrad.com). All three companies have been active in the hydroacoustic manufacturing and research fields for many years, during which time equipment has evolved to a current generation of split-beam systems which significantly out-perform earlier generations and so is strongly recommended for use in all fish investigations in lakes and reservoirs. All three companies also offer options for sound frequencies over the range typically used in freshwater applications, transducers suited for vertical and horizontal applications, provision for direct inputs of location data (essential for some types of hydroacoustic data analysis and highly desirable for all) from a GPS unit, are of similar physical size and weight, and can be powered from a 12 volt battery or from the inboard power systems of larger survey vessels. Transducer orientation sensors, transducer rotators and other miscellaneous hardware components are also available but are not fundamentally essential for most applications.

BioSonics, HTI and Simrad all provide proprietary software for use on a laptop computer to control the echo sounder during surveys and for other associated tasks in the field. Specialised software is also essential for post-survey data analysis. Again, all three manufacturers provide analysis software of varying complexity as part of their systems and data certainly can be and are analysed using only such software, but many researchers also use analysis software produced by third parties. In addition to more sophisticated and often faster analytical capabilities, such third party software can read the propriety data files produced by hardware from the different manufacturers. This is a major advantage for collaboration between researchers using different hardware systems and is an approach long-adopted by the marine hydroacoustics research community, where international collaborations have been commonplace for many years. Many members of the marine community have standardised on Echoview produced by Myriax (Myriax Software Pty Ltd, Australia, www.myriax.com), which was initially developed for large-scale marine applications but is now also used by some members of the freshwater community. Alternatively, Sonar5-Pro produced by Lindem Data Acquisition (Lindem Data Acquisition, Norway, www.fys.uio.no/~hbalk/sonar4_5/index.htm) was originally developed for use with data collected from lakes, reservoirs and rivers and in Europe at least is becoming established as the de facto standard for data analysis.

It is highly desirable and arguably essential that researchers intending to lead hydroacoustic surveys are provided with some level of training. BioSonics, HTI and Simrad all periodically offer multi-day training courses covering the general principles of hydroacoustics and the specific hardware and software components of their systems. Similarly, Lindem Data Acquisition and Myriax also periodically run training events including courses and workshops.

Guidelines

Although space limitations preclude the presentation of specific details here, WISER Deliverable 3.4-3 'Guidelines for standardisation of hydroacoustic methods' gives detailed guidance on pre-survey planning (general considerations, design of survey route, sound transmission and recording parameters), survey and data acquisition (immediate pre-survey activities, survey itself, immediate post-survey activities), post-survey data analysis (general considerations, choice of analysis method, echo counting, trace counting, echo integration, further processing of analysis results), and finally reporting and data archiving. Where appropriate, reference is made to a developing and more general, but also more technical, CEN standard 'Water Quality - Guidance on the estimation of fish abundance with mobile hydroacoustic methods'.

Conclusions

The technique of hydroacoustics has undergone remarkable advances in recent years such that it now constitutes a powerful tool for the survey and assessment of fish populations in lakes, reservoirs and other freshwater bodies. Increased appreciation of the opportunities afforded by hydroacoustics and the adoption of the guidelines developed within the present work will help to produce hydroacoustic surveys which are compatible with current best practice and as a consequence will facilitate the future valid comparison of hydroacoustic datasets for lakes and reservoirs from across Europe.