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Review

Assessing the ecological status in the context of the European Water Framework Directive: Where do we go now?

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HIGHLIGHTS

- A Science-Policy Interface activity was led by DG Research & Innovation and Onema.
- The aim was to establish a list of research needs for enhancing WFD implementation.
- For ecological status, 10 research issues were identified.
- The outcomes of SPI are likely to feed into the revision of the WFD.

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ABSTRACT

The Water Framework Directive (WFD) is now well established as the key management imperative in river basins across Europe. However, there remain significant concerns with the way WFD is implemented and there is now a need for water managers and scientists to communicate better in order to find solutions to these concerns. To address this, a Science-Policy Interface (SPI) activity was launched in 2010 led by Directorate-General for Research and Innovation and Onema (the French national agency for water and aquatic ecosystems), which provided an interactive forum to connect scientists and WFD end-users. One major aim of the SPI activity was to establish a list of the most crucial research and development needs for enhancing WFD implementation. This paper synthesises the recommendations from this event highlighting 10 priority issues relating to ecological status. For lakes, temporary streams and transitional and coastal waters, WFD implementation still suffers from a lack of WFD-compliant bioassessment methods. For rivers, special attention is required to assess the ecological impacts of hydromorphological alterations on biological communities, notably those affecting river continuity and riparian covering. Spatial extrapolation tools are needed in order to evaluate ecological status for water bodies for which no data are available. The need for more functional bioassessment tools as complements to usual WFD-compliant tools, and to connect clearly good ecological state, biodiversity and ecosystem services when implementing WFD were also identified as crucial issues.

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Contents

1. European context

The European Water Framework Directive (WFD), which was adopted in 2000, changed the way European Union (EU) Member States (MS) considered water management by putting ecosystem integrity at the base of management decisions (European Commission, 2000). Since then, all MS expended considerable time and resources to collect appropriate biological, environmental and pressure data and to develop operative tools in order to elaborate river basin management plans (Birk et al., 2012). As the magnitude and difficulties of this large-scale endeavour became evident, both the European community and individual MS have funded a large number of research projects, particularly in the areas of ecological assessment and catchment modelling (e.g. Hering et al., 2013).

The WFD was welcomed by many for its innovativeness and the radical shift towards measuring the status of all surface waters using a range of biological communities rather than the more limited aspects of chemical quality or targeted biological components. A WFD-compliant method necessarily complies with the requirement to include all the biological parameters listed in the normative definitions (Annex V), relates to reference conditions *i.e.* pristine conditions and expresses results as an Ecological Quality Ratio, a relative and comparable measure of quality. Recent years have been pivotal for ecological assessment of water quality in Europe (Nõges et al., 2009; Hering et al., 2010). After several years of scientific and technical work as well as important financial contributions from MS, the second round of intercalibration for biological methods was achieved, greatly improving homogeneity in the assessment of ecological status throughout Europe (Birk et al., 2013), for all surface water body categories (rivers, lakes, transitional and coastal waters). A major step was achieved at the end of 2012 with the intercalibration of 230 methods from 28 countries (European Commission, 2013a). Nevertheless, further effort is still required as around 100 methods (30% of the total) are not developed and/or not intercalibrated (Poikane and van de Bund, personal communication).

A key stage in the implementation of the WFD was therefore reached, whereby a state-of-the-art evaluation can be undertaken and the scientific outputs required to move forward in water management identified. More and more tools are becoming available for integrative management of hydrographic basins (Hering et al., 2013). Several recently completed European research projects specifically designed to support water body/ catchment management: WISER: *"Water bodies in Europe—Integrative Systems to assess Ecological Status and Recovery"* (http://www.wiser.eu/), MIRAGE: *"Mediterranean Intermittent River ManAGEment"* (http://www.igb-berlin.de/mirage.html), REFRESH *"Adaptive Strategies to Mitigate the Impacts of Climate Change on European Freshwater Ecosystems"* (http://www.refresh.ucl.ac.uk/), etc., as well as the contributions of national scientific programs were important to this aim. An efficient strategy for communication and transfer of knowledge has been established, in order to make water managers aware of the availability of these tools and how to use them. If this knowledge and these tools are not taken up, it will constitute an enormous loss of scientific profit for the water community (scientists, managers, end-users, and consumers in a broad sense) as a whole, and will inevitably cause major delays in WFD implementation. Likewise, water managers have identified bottlenecks and barriers to implementation that need to be tackled as a matter of priority.

In this context, a Science-Policy Interface (SPI) activity was launched in 2010 led by Directorate-General for Research and Innovation and Onema (the French national agency for water and aquatic ecosystems), for the period 2010–2012. The Directorate-General for Research and Innovation's mission is to develop and implement the European research and innovation policy with a view to achieving the goals of Europe 2020 and the Innovation Union. In this context, the Common Implementation Strategy (CIS)-SPI activity aimed to support the river basin management planning process by linking the research needs of end users with scientific research outputs and promoting partnership relations between researchers and policy makers. The first 'water science meets policy' event (SPI event) was held on 30 September 2010 and was attended by 150 participants from 15 MS and Switzerland as well as Non-Governmental Organisations, stakeholder groups, the Joint Research Centre, DG Research and Innovation and DG Environment. The present work represents a summary of the discussion, outcomes and recommendations from the event by the working group A (WG A, also called ECOSTAT) of the SPI activity, devoted to ecological status evaluation, and is intended to identify the way forward for ecological assessment over the next decade. To do so, a state-of-the-art of the available scientific knowledge for each issue raised during the SPI activity was achieved, as well as identification of research needs and means of making progress. This study complements the work of Hering et al. (2010) which sought to make recommendations for enhancing WFD implementation 10 years after its adoption. Three issues were particularly addressed by Hering et al. (2010): (i) the development of assessment methods (including reference conditions, typologies and intercalibration); (ii) the implementation of assessment systems in monitoring programmes; and (iii) the consequences for river basin management plans (such as the design, monitoring and success of restoration measures). As with Hering et al. (2010), this contribution will be of particular interest for scientists working in the field of aquatic ecology and assessment of anthropogenic pressures impacts, as well as for water managers directly concerned by WFD implementation.

2. Outline of the SPI process

The first SPI event took place on 30th September 2010 in Brussels (Belgium). During this event, there was a relatively balanced representation between researchers and WFD end-users (35% of representatives were from the scientific community and 65% were end-users). End-users category covered all the three main levels of implementers or decision makers in relation to WFD, from the European level to national and river basin levels. They represented different points of view but were all invested in WFD implementation, even if in different ways, *i.e.* from general issues to specific local ones such as the definition of restoration measures. Research needs were discussed at parallel round tables. The round table themes were aligned with the CIS groups (ecological status, chemical aspects, groundwater, floods, water scarcity and droughts, WFD and agriculture, hydromorphology), together with a number of cross-cutting issues (socio-economics, integrated river basin management plans/ management and dissemination). In total, 59 research areas (encompassing about 180 specific research issues) were discussed (the full "Water Science meets Policy" report can be downloaded at: http://www.onema.fr/IMG/EV/cat1a-13.html).

For working group A (ecological status), 16 research issues were prioritised and discussed. After the first SPI event was completed, a formal consultation of the ECOSTAT working group was undertaken. ECOSTAT led the intercalibration exercise in relation with the European Commission's Joint Research Centre (JRC) in Ispra (Italy), and brings together stakeholders from every MS (water managers, scientific experts and representatives of ministries). The SPI process was presented to ECOSTAT members and comments/identification of additional issues were invited. This resulted in a consolidated shortlist of 10 priority research issues which was finalised at the beginning of 2012. Some discussion points on the SPI process were then made at each ECOSTAT meeting.

From April to June 2012, scientific experts working in different research fields identified were solicited and asked to contribute a state-of-the-art summary, including knowledge gaps for each research issue. These elements were compiled into a synthesis document in late 2012 and transmitted to the Strategic Coordination Group at the end of 2012, as part of the CIS-SPI progress report for transmission to the Water Directors group. Discussion among the different working groups of SPI (chemical aspects, groundwater, etc.) was facilitated during the whole process (notably, the ways to validate the different steps of the process in each group were compared), and the CIS-SPI final activity report was then published (European Commission, 2013b; http://bookshop.europa.eu/en/science-policy-interface-in-support-of-the-water-framework-directive-pbKI3112744/), bringing together the information collected from the SPI activity.

3. Top-10 issues

Following the first SPI event and formal consultation with CIS-ECOSTAT members, the following research issues were identified as of primary concern for enhancing WFD implementation.

3.1. To overcome knowledge gaps for transitional waters

Transitional waters (*i.e.* estuaries and lagoons) are usually viewed as complex ecosystems as they constitute ecotones between freshwater and saltwater (they therefore exhibit characteristics from these environments as well as specific ones). Moreover, they are of particular concern for management and conservation issues as they usually shelter very abundant and diversified biological communities (algae, phanerogams, invertebrates, fish, birds, etc.). Within this context, we suggest that the following issues are given consideration in the future to enhance ecological status evaluation.

3.1.1. Multi-pressure context

Among transitional waters, estuaries are the water bodies most affected by pressures resulting from activities such as dredging, land reclamation, harbor and industrial development, as well as recreational and tourism development which have induced major alterations to the original hydromorphological characteristics (European Environment Agency, 2012; Fehér et al., 2012). The water quality of these environments is also affected by important of pollutants from domestic and industrial effluents (Borja et al., 2011), and is strongly dependent on the chemical fluxes coming from the upstream areas of the drainage basins (Masson et al., 2006). At last, the ecology of estuaries has been subjected to intense human influence for many years, from intensive commercial harvesting and aquaculture (Sousa Leitao and Gaspar, 2007). One of the challenges in transitional waters, and given the difficulty of distinguish between the effects of natural and anthropogenic stressors in these environments (the so called "Estuarine Quality Paradox"; Elliott and Quintino, 2007), is therefore to underpin decision making, risk assessment and management of these systems under complex multiple stress conditions. Research should enhance the understanding of multiple stressor interactions (Thrush et al., 2008; Ban et al., 2010). In this context, some biological quality elements (BQEs) may be more sensitive to some of the pressures than to others (e.g. phytoplankton and macroalgae to eutrophication; seagrasses and fish to habitat loss or hydromorphological changes).

3.1.2. Reference conditions

Estuaries are very dynamic water bodies because of their location at the interface between freshwater and saltwater. They are therefore prone to considerable physical and chemical variation, notably water depth and velocity and associated chemical parameters (salinity, conductivity, etc.). Besides, the transitional waters fish component comprises marine, estuarine and freshwater species, and estuarine phytoplankton and nutrients are influenced by catchment run-off and marine/tidal flushing (Elliott and Whitfield, 2011). Hence, it is difficult to detect the anthropogenic influence against a background of natural variation in these very dynamic environments (Elliott and Whitfield, 2011). These natural variations involve some difficulties to define reference conditions, all the more that pristine or minimally disturbed estuaries are rare at the European scale. This leaves 'best professional judgment' as the most practical tool for setting reference conditions in transitional waters (Basset et al., 2013). Stoddard et al. (2006) defined best professional judgment as judgment based on *"experienced aquatic biologists, with perhaps decades of experience sampling and examining physical, chemical, and biological attributes across wide ranges of severity and types of human disturbance"* who developed *"an empirical understanding of condition in the absence of significant human disturbance"*. It has proven to be very useful in assessing the status of areas across United States and Europe, with a common set of criteria among different experts (Borja et al., 2012). Hence, exploring the use of these alternative options in setting reference conditions should be encouraged.

3.1.3. Estuaries as a part of the river basin management

We know that climate, oceanic, riverine and catchment factors control a hierarchy of processes and broadly determine the physical and biological characteristics of estuaries (Hume et al., 2007). Hence, when taking measures to lessen or remove human pressures, these factors must be taken into account in order to ensure successful recovery (Borja et al., 2010). A better knowledge of the environmental and biological relationships across the river-estuary-coastal continuum is therefore needed for a better understanding of the response of estuarine systems to the measures and the recovery processes. Truly integrated coastal management, where terrestrial and marine managements are considered together should be a priority for understanding the impacts and the effects of mitigation strategies (Meiner, 2010).

3.2. To overcome knowledge gaps for lakes

Over the last decade, WFD-compliant assessment methods for lakes have been developed for most BQEs, supported by major European projects, e.g. REBECCA and WISER (Lyche-Solheim et al., 2008, 2013). There are now close to 100 different national assessment methods for lakes across Europe (Brucet et al., 2013). Despite this indubitable progress, the following recommendations can be made to enhance WFD implementation.

3.2.1. Improving indicators

Different BQEs in lake assessment are needed to address different anthropogenic pressures (e.g. phytoplankton usually addresses eutrophication while benthic invertebrates are good indicators of acidification and morphological degradation), different habitats (e.g. benthic invertebrates and macrophytes are usually used to assess littoral habitat while phytoplankton is used for pelagic areas), different timescales (e.g. phytoplankton may react immediately to the modifications of the environment while macrophytes respond more slowly to the changes in pressures) and different ecosystem services (e.g. harmful algal blooms are crucial indicators for recreation and provision of drinking water while condition of alimentary resources is mainly related to fish). Most methods available to date mainly address eutrophication (or general degradation), whereas lakes are generally affected by multiple stressors, climate change including browning of the water, biological manipulation (e.g. fish management and macrophytes removal) and hydromorphological alterations (European Environment Agency, 2012). Lake assessment tools should now be developed to address pressures other than nutrient enrichment and to disentangle the effects of multi-stressors. Special attention should be paid to cyanobacterial blooms as they threaten ecosystem services and ultimately human health (Carvalho et al., 2013), as well as to macrophytes and macroinvertebrates growing and living in the littoral areas as they are sensitive to water level fluctuations and alteration of the littoral habitats (Mjelde et al., 2013).

Some of the pressure–impact relationships established to develop the national BQE methods are not yet sufficiently robust, and often defined using small datasets. Moreover, the large diversity of methods and strategies used (Brucet et al., 2013) and the large number of national lake types (*i.e.* 673) where the majority is not clearly linked to the common intercalibration types, restricts comparability of class boundaries as well as performance of methods in different situations. Thus, better and more harmonised metrics for all BQEs must be developed allowing more robust pressure–impact relationships to be defined. This is vitally important in the context of revising the current nutrient standards and to ensure that these are correctly linked to the intercalibrated good/moderate class boundaries, as nutrient standards are the benchmark used to plan the programmes of measures.

Management practices for fish have not been fully considered in the development of bioassessment methods based on fish communities, despite the possibility that this could be a major source of bias and/or variability in the pressures–impacts models (Argillier et al., 2002). It is necessary to distinguish the impacts of these activities from those of other human pressures on fish-based indicators to improve the predictive power of the indices for both natural lakes and reservoirs.

3.2.2. Revisiting European lake typology

There is an urgent need to harmonise lake typologies across Europe, to ensure more comparable reference conditions and to establish environmental target values. The difficulty of achieving 'type-specific' assessment in some cases was already emphasised by Hering et al. (2010) in their critical review of the WFD. A first step in this process could be to identify a small number of broad lake types across Europe to allow grouping of existing national types, while a second step could be to harmonise the ranges of the most commonly factors used to establish the typologies (e.g. alkalinity, colour, mean depth, altitude, surface area).

3.2.3. Ecological assessment methods in the Eastern Continental and Mediterranean regions

There have been additional difficulties with the development and intercalibration of ecological assessment methods for the Eastern Continental and Mediterranean lakes, notably naturally eutrophic lakes in the continental lowlands (Borics et al., 2013). As a result, the nutrient content of these lakes, even in a relatively natural state, can be high ($TP > 100\ \mu g\ L^{-1}$) but no significant relationships may be found between phosphorus, chlorophyll, and macrophyte coverage (Krasznai et al., 2010). As well as different targets, these lakes may require different management measures as a large enough reduction of nutrient loading to result in an expected response may not be feasible within a realistic time scale (Borics et al., 2013). Identifying functionally-different lake types and setting eutrophication targets for these lakes is an important area that requires further research.

3.3. To analyse the links between ecotoxicology and bioassessment tools

The implementation of several European new regulations (e.g. WFD, REACH) still requires important efforts to develop ecotoxicological risk and biological assessment tools for water bodies. Ecotoxicological Risk Assessment (ERA) is primarily a predictive discipline that attempts to predict the future consequences of water or sediment contamination by chemicals from agriculture, industry, or any other uses. On the other hand, bioassessment in a WFD context mainly focuses on patterns and processes that occurred in the past or are occurring in the present, which resulted in the current ecological status of the water bodies. However, there are predictive approaches in bioassessment as well as retrospective methods in ERA where these two research-and-practice disciplines merge (Verberk et al., 2013). An international consensus now supports the need to move from a fundamentally toxicology-based to a more ecology-based risk assessment of chemicals (Artigas et al., 2012). In this context, the following recommendations can be made.

3.3.1. Pushing forward the development of trait-based bioassessment tools

Biological traits are well defined, measurable properties of organisms that strongly influence organismal performance, and constitute the causal mechanisms underlying the relationships between the species and their environment. As a result, trait-based approaches have clear potential in environmental diagnoses and prognoses (Dolédec and Statzner, 2008; Mondy et al., 2012), notably by (i) providing mechanistic understanding allowing cause–effect relationships between stressors and biological impairments to be inferred; (ii) enabling the evaluation and prediction of anthropogenic impacts at large spatial and temporal scales, and (iii) establishing links between community organisation and ecosystem goods and services. Biological traits (e.g. lifeforms and cell-sizes for diatoms, relative abundance of invertebrates from “plurivoltine” species or species with “aquatic passive dispersal” or using “ovoviviparity” as reproduction technique) offer the potential to resolve the effects of multiple stressors (Mondy and Usseglio-Polatera, 2013), given the diversity and specificity of trait responses (e.g. Schäfer et al., 2011). However, they have been rarely included in WFD compliant methods, except feeding habits or guilds based on food preferences for invertebrates and fish (e.g. Böhmer et al., 2004), or longevity, age structure and reproductive or migration behaviour for fish (e.g. Pont et al., 2006). Besides, multi-metric tools facilitate the diagnosis of the most probable causes of degradation at a given site or water body, and therefore relevant selection of appropriate restoration measures (Hering et al., 2010). For instance, if a biological metric sensitive to toxics is impacted on a given water body, it can be assessed with the knowledge of pressures likely to explain failure of the environmental targets in mind, and restoration measures selected accordingly. Multi-metric tools based on biological traits are therefore a promising research avenue for the ecological evaluation of water bodies in a multi-pressure context.

3.3.2. Reconnecting chemical and ecological evaluations. There is a disconnection between the evaluation of ecological status on the one hand and chemical status on the other hand. Chemical status does not provide water managers with a straightforward means to efficiently reduce the discharges of harmful substances. This is especially true regarding sediment contamination, for which BQEs such as macroinvertebrates which are directly in contact with the sediment would be useful. The reason for this disconnection is that the procedure for chemical status evaluation is often based on an ERA which cannot be compared with the observed values of ecological indices calculated on the basis of data gathered from field surveys. Thus, one drawback in the WFD as it stands is the difficulty with defining relevant environmental quality standards for chemical substances that are truly related to the ecological assessment results. To overcome this, a potential approach could be to compare the reaction of both biological indicators and biomarkers with a common set of toxics at different levels, from large-scale statistical data analyses to *in situ* comparative studies and mesocosm experiments. This would provide important fundamental and applied knowledge that could facilitate the diagnosis of water status.

The ecological vulnerability of ecosystems may be viewed as the combination of potential exposure risk, trait-based sensitivity and recovery capacity, and be predicted using auto-ecological information on internal metabolism, regulation capacity, toxicological sensitivity of individuals and traits (De Lange et al., 2010). The development of integrated methods enabling the assessment of ecosystem vulnerability and resilience to toxic substances, and increasing understanding on ways to reduce vulnerability is challenging, but offers considerable potential for surface water management. Within this context, it is important to consider life-history theory issues and associated specific traits (fecundity, egg size, triglyceride contents, etc.), which determine both resource allocation and energy pathways in a dynamic evolutionary context, as these govern individual fitness and, ultimately, demographic issues (see Stearns, 1992). It would be also important to consider the concept of Pollution Induced Community Tolerance (the PICT approach), taking into account the potential replacement of sensitive species by more tolerant ones (especially in photoautotrophic communities) following chronic contamination for providing ecologically relevant predictions of toxic effects in the environment (e.g. Blanck, 2002).

3.4. To overcome difficulties in assessing ecological status in temporary streams

Temporary streams comprise half the global river network and this proportion is predicted to increase due to global change (Carlisle et al., 2010). The recurrent cessation of water flow of temporary rivers influences biotic communities as well as nutrient and organic matter processing (Larned et al., 2010). Several studies have focused on the highly adapted biological communities that live in these streams (reviewed in Lake, 2011). However, until now they have not been fully integrated into water regulations because most water managers apply perennial river management principles when making decisions related to temporary streams. Based on this, we recommend the following issues are given further consideration.

3.4.1. Developing specific bioassessment methods

The definition of six aquatic states (hyperrheic (floods), eurheic (continuous flow with riffles), oligorheic (connected pools), arheic-(disconnected pools), hyporheic (no surface water, alluvium saturated) and edaphic (alluvium not saturated)) by Gallart et al. (2012) summarised the set of aquatic mesohabitats which occurs on a given stream reach at a particular moment depending on the hydrological conditions. In this context, while a myriad of methods are available to establish the ecological status of permanent streams (Birk et al., 2012), no methods are defined for ephemeral streams and very few for intermittent streams. In certain cases, the same methodologies used for permanent streams could be of use, notably when the stream has been in a eurheic or oligorheic state for a sufficiently long period (García-Roger et al., 2011). For other cases there is an urgent need to develop robust and specific bioassessment tools, as well as to incorporate expert knowledge in the diagnosis (European Environment Agency, 2012). The work conducted within the collaborative EU-funded project Mediterranean Intermittent River ManAGEment (MIRAGE) has addressed most of the difficulties associated with ecological status evaluation in temporary streams and has used diverse approaches to solve them. These approaches have been brought together in the so-called MIRAGE Toolbox (Prat et al., 2014). Other promising approaches would consider hyporheic communities in the bioassessment tools developed for this type of streams (notably phytobenthos and invertebrates) as it is well-known that these organisms may survive in the substratum even during long dry events, as well as the use of molecular methods (Biomonitoring 2.0; Baird and Hajibabei, 2012). Given the particularly high natural variability of temporary streams, it is of particular importance to ensure biological data are adequately collected and reliable information about pressures is available before making any development, whatever the method being favoured.

3.4.2. Optimizing ecological assessment relevance

The innate variability that characterises temporary streams provides challenges for the assessment of their ecological status because reference conditions may vary between seasons, dry and wet periods, and after hydrological events in the same river type (Munné and Prat, 2011; Prat et al., 2014). Ecological status assessment in temporary streams requires preliminary analysis of the hydrological regime (permanent, intermittent with pools, intermittent dry and ephemeral) as well as knowledge of the aquatic state of the stream at least three months before sampling (Gallart et al., 2012). Once this analysis has been done, the sampling calendar is crucial as temporary streams should be sampled during the flowing phase but at least one month after the spring floods. In this case the standard ecological assessment methods for permanent streams may be used. Under other circumstances (e.g. floods or droughts), the aquatic organisms are subjected to pressures and comparisons with ecological status in permanent streams are less reliable. It is important therefore to determine whether or not the intermittence of a given stream is natural or not, as subsequent interpretations and recommendations would differ from one case to another one. In this context, the hydrological status concept has been introduced (Cazemier et al., 2011) allowing reconstruction of the natural hydrological regime of temporary streams and predictions about changes in the future using rainfall-runoff parameters. Another key issue is the requirement for improved knowledge of ecological status during the arheic (only isolated pools) or dry states and there are currently methods in development for both cases (Steward et al., 2011). In this context, the use of functional measures has some promise but in all three cases we are still far from implementation and more research is needed.

3.4.3. Defining a temporary stream typology for Europe

The presence of temporary streams in the hydrographical network of drainage basins is a characteristic shared by numerous MS across Europe, not only in Mediterranean areas. Even if temporary rivers are very frequent and abundant in Southern Europe (e.g. in SE Spain even some large catchments >50 km length are intermittent or ephemeral), there are also many temporary streams in non-Mediterranean or mountain areas across Europe. For instance, it has been shown by Datry et al. (2011) that the density of temporary streams in the Mediterranean areas of France (a dry climatic zone) is comparable with Brittany (a climatic zone with high levels of precipitations). Within this context, it would be very useful to map the temporary streams at the European scale and to propose a European typology for this type of streams (The RM5 river type envisaged in the intercalibration exercise is a mixture of river types and was not intercalibrated for such reason; Feio et al., 2014). This would greatly facilitate the definition of reference conditions for temporary streams and subsequently ecological status thresholds.

3.5. To take into account uncertainties in ecological evaluation

The importance to deal with uncertainties was already pointed out by Hering et al. (2010) in their review of the difficulties met when implementing WFD. Uncertainties in ecological state assessment may have many sources. As an illustration, guidance 13 of the Common Implementation Strategy for the WFD listed several potential sources of variation for a given water body (European Commission, 2005): daily and seasonal patterns, longer temporal trends e.g. climate warming, variation with sampling location, bias from equipment, etc. These different types of variation can be clustered into three different categories, *i.e.* natural temporal variability, natural spatial variability and human bias. The following ways to deal with these sources of variability could be explored.

3.5.1. Improving skills

The best way to limit human bias is to improve skills in applying both the sampling protocols and evaluation tools. This can be achieved by standardizing methods, both at the European level (*i.e.* European Committee for Standardization) and at national levels (national standardisation committees), and by producing best practice documents for several key steps, from selection of sampling sites to protocol application and taxonomic determination (in laboratories or directly onto the field depending on the BQE). Once the protocols have been standardised, training people in the application of field or laboratory protocols as well as quality assurance procedures are crucial to enhance confidence in ecological evaluations.

3.5.2. Developing "stable" bioassessment tools

Once protocols have been standardised, people adequately trained and quality assurance procedures adopted, it is clearly needed that water managers rely on "stable" bioassessment tools, *i.e.* tools which take into account the natural variability of communities and do not assign bad status for wrong reasons. Typically for rivers, a high flow event can drastically affect communities but be considered as natural as long as it is fits the natural hydrological cycle observed at the (sub-)basin scale (*i.e.* the so-called "Natural flow regime", Poff et al., 1997). It should therefore not result in a water body being declassified for bad reasons (some species might be favoured by high flow events while other might collapsed). These natural abiotic events (e.g. high flow events, "hot" years) are likely to have direct and indirect effects, the latter by creating conditions which modify biotic interactions, which may in turn affect population dynamics and ultimately ecological quality class. It is therefore important that bioassessment tools integrate natural variability in their development (e.g. Marzin et al., 2014), so that water managers can rely confidently on the ecological status evaluation being performed. In this context, it is also becoming increasingly important to consider uncertainties associated with global warming and their influence in the shift of reference conditions (Logez and Pont, 2013).

3.5.3. Developing "probabilistic" tools

The final output of WFD biological assessment is a 'quality class' (from high to bad) for each individual BQE. Within this context, and because of the different sources of uncertainty in ecological evaluation, it is now very important to provide water managers with tools providing probabilities for the different status classes rather than single 'simplistic' assignments. For instance, for a water body the outcome of a tool providing the probabilities (from high to bad) of: 30–50–10–10–0% would be very different from another water body where the same tool gives probabilities of: 20–30–20–20–10%. In both cases the water body would be classified in good status, but while in the first case the diagnosis (80% chance of at least good status) will be very reliable, the second provides only a 50–50% chance to be either in at least good status or less than good status. The use of probabilistic tools for one (Marzin et al., 2014) or, ideally, all BQEs of a given water body (Marzin et al., 2012) provides a promising approach as it would give much more confidence to water managers when designing their programmes of measures, therefore limiting some of the drawbacks associated by the one-out, all-out principle (Hering et al., 2010).

3.6. To develop models for the spatial extrapolation of ecological status

During the past 10 years, most of the WFD oriented research has focused on the development of bioassessment tools for the different BQEs using data from national monitoring networks. However, a large number of water bodies are not directly monitored across Europe. Within this context, it is of crucial importance to develop efficient extrapolation methods which enable a diagnosis to be made without directly measured biological and chemical data, for all water body type categories. The following approaches can be recommended.

3.6.1. Developing models relating biota to pressures at different spatial scales

There is a clear lack of predictive tools linking pressures to ecological status at large or regional scales. Recently developed tools offer both scientists and water managers a means to do this. For instance, Donohue et al. (2006) showed that urbanisation, arable farming and extent of pasturelands are the main factors affecting ecological status of streams in Ireland, and that the probability of a river site achieving good status can be predicted using simple models requiring widely available landcover data or chemical data. In Denmark, Kristensen et al. (2012) developed models capable of predicting the presence of different stream fish based on land use data at different scales, showing that the presence of assemblages can be predicted with high accuracy. In France, Villeneuve (2010) developed models predicting the ecological status of rivers based on biological indices calculated using macroinvertebrates, phytobenthos and fish data using variables related to land use, hydromorphological features and physico-chemical conditions. These examples illustrate the types of predictive models that should be pushed forward as they could be particularly useful for implementing risk assessment procedures and identify water bodies likely to fail environmental objectives (Article 4 and annex II 1.5 of the WFD).

3.6.2. Integrating expert judgment in diagnosis tools

Another approach to assess the ecological status of non-surveyed water bodies involves using expert judgment in the diagnosis (European Environment Agency, 2012). Models may provide water managers with objective predicted data/outputs for a given sampling site or water body, but these results may be very variable in terms of usefulness depending on the resolution of data used to build the model (the “calibration” dataset). For example, some models may provide insights for nutrient contents or degree of river chenalisation, but if they were developed using data collected at large spatial scale (*i.e.* at catchment scale or even national scale) or with relatively low resolution, they may be inappropriate for delivering appropriate information at the local scale where diagnosis needs to be made. Taking into account expert judgment could reduce this bias, by directly injecting knowledge on local context/pressure in the diagnosis parallel to model outputs. However, this option may lead to a heterogeneous assessment across MS as practices may differ among water managers, so that specifying the way expertise applies should be addressed through normative documents or technical European guidances. Bayesian modelling, which enables to directly inject expertise in the model outputs, would also be a promising research avenue (Marzin et al., 2014).

3.6.3. Improving (sub)basin-scale nutrient models

Physico-chemical data are often measured at the monitoring network site level. Yet, physico-chemical parameters, as well as hydromorphology, must be considered in the WFD as supporting elements governing the development of the biological communities. In this context, models capable of simulating physico-chemical variables at the river basin scale (notably nutrients) based on basin-specific hydrological features as well as land use and population density, are now required (e.g. INCA model; Whitehead et al., 2002, MONERIS model; Behrendt et al., 2003). This is a prerequisite for implementation of real basin-integrated approaches in the MS, greatly facilitating comparisons from one country to another. Furthermore, these models should be reliably linked to the biological elements of status, to generate efficient programmes of measures.

3.7. To better understand hydromorphological impacts

River hydrology and habitats have been substantially altered during the last century, while eutrophication, toxic substances and emerging stressors contribute to the complex set of pressures affecting our rivers. The benefits of wastewater treatment have been thoroughly documented over the last 30 years. In contrast, the response to hydromorphological restoration was shown to be more complex and less predictable (Vaughan et al., 2009). Thus, there is a great need to better understand and predict the costs and benefits of future river hydromorphological restoration projects (Buijse et al., 2005; Lake et al., 2007), as there is a lack of knowledge in several key areas including: (i) the characterisation of hydromorphological status focuses on patterns not processes; (ii) data collected represent small spatial scales, with larger spatial scales or long term impacts are often neglected; and (iii) the way hydromorphological change affects BQE and ecological functioning is inadequately understood. The Heavily Modified Water Bodies (HMWB), which represent a significant proportion of water bodies across Europe, are particularly concerned by these issues (European Environment Agency, 2012). For HMWB, a *'good ecological potential'* must be reached in the same way as good ecological status for natural water bodies, but only few member states actually do have approaches to achieve it. Based on these concerns, the following recommendations can be made.

3.7.1. Restoring continuity: sediment transport, water flows and fish migration

There is a need to understand more thoroughly the processes and dynamics of sediment transport, hydraulic connectivity and flow regimes within river systems (Gurnell, 2014). The use of demonstration projects to improve or restore sediment transport would provide the framework to improve existing knowledge. This should be fully evaluated and shared among MS. There is also a need to assess the impacts and effectiveness of the removal of in-stream structures in terms of fish migration, especially in large rivers (e.g. for sturgeon, eels and salmon smolts), which remain poorly studied. While the ecohydraulics of fish passes for large migratory fish and salmonids are well known, these remain unclear for smaller species (Santos et al., 2012). Rivers present a multitude of small obstacles and the effect of these on the ecohydraulics of migration upstream remains poorly assessed for every kind of species (salmonids and non-salmonids). Moreover, most migratory species are strongly threatened at the European scale, and restoring continuity therefore constitutes an urgent and imperative need to keep these species from becoming extinct.

3.7.2. Linking geomorphology, habitats and biology

There is an urgent need to gather scientific evidence illustrating how geomorphology supports biota, and to improve the understanding of the links between morphology, habitats and biology. It is important to examine closely the geomorphological functioning of systems beyond a simple description of the geomorphological conditions. Many of the existing tools only give a description of condition rather than an understanding of functioning. There is a crucial need to understand the hydromorphological and biological responses to new modifications of water environment and future environmental change. Examples of such recent and upcoming modifications encompass modern interventions for flood protection (e.g. ‘room for the rivers’), land use changes affecting sediment budgets in streams and rivers (e.g. reforestation of mountain regions resulting from depopulation) and rehabilitation measures to regain dynamic processes (e.g. dam removal, remeandering and naturalizing riparian zones). Again, demonstration projects provide an excellent way to fully understand these links (Verdonschot et al., 2013).

3.7.3. Terrestrial and aquatic interface and catchment management

There is a need to understand the linkage between the terrestrial catchment and the aquatic ecosystems, including the impact of different land management practices on the water environment, which could give insights into the restoration of the different parts of a catchment (e.g. the headwaters, floodplains and riparian corridors). Some water bodies may depend upon the chemical or ecological, as well as hydromorphological status of other water bodies situated upstream (and possibly downstream, e.g. regressive erosion). There is a need to fully understand the three dimensions of connectivity of river systems (*i.e.* upstream–downstream connectivity, floodplain connectivity and links to groundwater) to better inform WFD assessments and select mitigation measures.

3.7.4. Rehabilitation in degraded ecosystems

In heavily degraded and multi-stressed systems, there is a need for a specific restoration approach, a 'reasonable' objective being more an 'ecological improvement' rather than a 'complete restoration' of the water body (e.g. in systems exhibiting heavy flow regulation due to several dams) (Buijse et al., 2002; 2005). In particular, there is a research gap in our technical understanding of how to rehabilitate very large rivers, especially heavily incised river systems. This has relevance to the definition of *'good ecological potential'* for HMWB as the different practices across MS need to be compared and discussed.

3.8. To develop functional assessment tools

Many bioassessment tools based on metrics reflecting community attributes (taxonomical composition, traits occurrence, etc.) have been developed under the WFD (Birk et al., 2012). These methods generally reveal organism stress along gradients of environmental quality and deviation from reference conditions. Even if these tools are widely recognised as useful in a management context, they only partially highlight specific changes in ecosystem functioning and processes, even if some promising multimetric tools have sometimes been developed taking into account biological traits related to functions (e.g. reproductive strategies, feeding, migratory capacities; Dolédec and Statzner, 2008; Mondy et al., 2012). However, when trying to restore ecosystem integrity, understanding the mechanisms that drive ecosystem functioning and stability/resilience can certainly help to identify the specific pressures responsible for the observed patterns. Here we suggest different ways to evaluate and quantify ecosystem functioning.

3.8.1. Focusing on ecological fluxes and trophic networks

One of the most conventional ways to assess ecosystem functioning is to focus on trophic networks and fluxes of matter and energy through the system. These fluxes are generally characterised by the levels of primary and secondary production, the efficiency of matter and energy transfers from one trophic level to another, and the biomass of the organisms belonging to the different trophic levels. For instance, the carbon flux can be characterised by the Lindeman efficiency α, which is the ratio of total metabolic energy fluxes at trophic level 1 to those at level 0 (Lindeman, 1942). The α ratio is generally calculated between the biomass of adjacent trophic levels. A more sophisticated way to proceed is to analyse global photosynthesis and metabolism of the ecosystem via the evaluation of gross and net primary production, ecosystem respiration, and CH_4 efflux (Yvon-Durocher et al., 2010). These variables are known to be sensitive to anthropogenic pressures and particularly to organic pollution and global warming (Young and Collier, 2009; Yvon-Durocher et al., 2011), showing potential as functional indicators. During the last 10 years, there has been a growing interest in using stable isotope analysis to characterise the general structure of trophic networks as it may provide insights regarding ecosystem stability and resilience (Arreguín-Sánchez, 2014). Ecological stoichiometry approach, which considers how the balance of energy and elements affects and is affected by organisms in the environment, has also been identified as a promising research avenue (e.g. grazer–periphyton interactions, consumer-driven nutrient recycling, multi-element nutrient spiraling in streams; Frost et al., 2005). At last, the rate of leaf litter decomposition, as it reflects processes acting at a very small spatial and temporal scales (*i.e.* microbial activity), may bring very early warning signals of increasing anthropogenic pressures (Gessner and Chauvet, 2002).

3.8.2. Integrating size as a functional metric

Size is a key biological characteristic in ecology, and is required as a normative condition in WFD (*i.e.* for fish-based bioassessment tools). Many ecological properties are governed by body size or body mass, from individuals (e.g. fecundity) to populations (e.g. population growth rate) and communities (e.g. inter-specific competition). Consequently, size structures have been used to evaluate ecosystem functioning (San Martin, 2005), and may be particularly useful for situations with low taxonomic richness (e.g. fish in some Alpine rivers). Evaluation using size may be based on the relationship between size classes and normalised abundances, with the slope of the linear relationship being considered to account for the efficiency of biomass transfer across the different trophic levels, while the intercept is believed to give an idea of the total abundance of organisms (San Martin, 2005). With increasing productivity, the determination coefficient usually tends to decrease (indicating that the system is far from a stable state) as well as the slope, and conversely for the intercept. Methods based on allometric relationships (*i.e.* changes in organisms in relation to proportional changes in body size) and size analysis also provide an interesting framework to assess ecosystem functioning, as they offer opportunities to estimate the ecological impacts of various anthropogenic drivers (Basset et al., 2012). The conjunction of allometric approaches and size structure analyses offers a potential way to develop functional bioassessment tools related to ecosystem resilience and stability.

3.8.3. Going deeper into thermodynamic concepts

Another way to assess ecosystem functioning which has been given some consideration recently is the use of thermodynamic concepts. Ecosystems can be considered as open and dissipative systems as defined by Prigogine (1969). As a consequence, there is a conceptual framework where thermodynamics and ecology are linked. Most of the research has focused on entropy (Margalef, 1996) and more recently on exergy (Jørgensen, 1992). For instance, it has been shown that the eco-exergy tends to decrease with increasing chemical pollution (Xu et al., 1999, 2011). However, if approaches based on thermodynamic principles are attractive, they suffer from a complicated theoretical background and from the difficulty to associate a decrease in eco-exergy to a specific pollution or disturbance. Nevertheless, it is important to develop these research approaches as physical and biological systems are intrinsically connected and thermodynamic concepts can be related to restoration efforts and programmes of measures.

3.9. To clarify the links between global change and ecosystem functioning

Unprecedented global changes resulting from societal use of natural resources will profoundly impact on aquatic ecosystems over the next century with major consequences for ecosystem structure and function (Sala et al., 2000). Climate change, water resource requirements, land use change, connectivity losses, alterations in global circulation of nitrogen and carbon, breaching of biogeographical barriers and biotic exchanges will result in major stress affecting biodiversity (including freshwater, estuarine and marine components) and ecological status at a global scale, despite policy-driven efforts to minimise these. These changes are likely to affect the implementation of the WFD quality assessments (Wilby et al., 2006; Hering et al., 2010) and the maintenance of good status. To tackle this issue, the following key research areas should be investigated in the near future.

3.9.1. Clarifying the nature and combination of effects from global stressors interactions

The nature of interactions between global stressors is still mostly speculative (Kernan et al., 2010). Some global changes may even be beneficial and compensatory, while in other cases primary global changes will trigger secondary stresses, as human society adapts. A major response to the predicted changes in seasonal variability and decreases in precipitation in Southern Europe will be water storage in existing and newly built reservoirs with subsequent alteration of flow regimes. The success of biotic introductions will vary according to environmental conditions and will be related to the degree of human activity but also to secondary pressures resulting from global change (Jeppesen et al., 2011). Global changes may trigger a series of stressor feedbacks and additive, synergistic or antagonistic interactions propagating across ecosystem compartments. Increased understanding of such dynamic stressor networks is required in order to increase our predictive capabilities at a global scale (Feld et al., 2011).

3.9.2. Identifying best available conditions and developing indicators for global pressures

WFD Planning depends on the use of reliable indicators of quality, measured against reference conditions (Hughes, 1995) and likely to evolve following global changes. Thus, the decoupling of effects due to natural temporal variability (e.g. demographic issues related to high flow events or "hot" years) from those resulting for anthropogenic changes on a longer time period (e.g. inter-annual trend due to climate warming) requires urgent attention (Nestler et al., 2010). We should be able to predict the ways and degree by which reference communities are affected by global stressors, how these alterations affect the ecological assessment of status, and how robust our assessment systems are. Long-time series of biological change in near-natural ecosystems used with modelling forecasts are invaluable tools to understand how global changes interact with natural variability. Developments with palaeolimnological methods mean this approach can now be used to assess alternative recovery targets, notably for lakes (Bennion et al., 2011).

3.9.3. Decoupling global non-biological and biological disturbances

Anthropogenic introduction is the main cause of non-native fish species invasions (Leprieur et al., 2008). Worldwide, barrier removal has resulted in a general homogenisation of freshwater faunas (Rahel, 2007). We need to understand cause–effect relationships between alien invasive species and global changes, potential conflicts (for example, barrier breaching as opposed to damming) and interactions with local scale disturbances (Strayer, 2010). In many European bioassessment methods, non-natives are usually included in community metrics and not considered as aliens likely to degrade ecological status (Birk et al., 2012), despite Kennard et al. (2005) demonstrated they can be seen as a reliable 'first cut' indicator of river health. We need to understand when biotic exchanges are a primary symptom of global change, or rather resulting from ecosystem cascading effects. Finally, we need to allow for the potential for species migration as a result of climate change, by encouraging habitat protection and connectivity and to understand and accommodate the difference between benign migration and real invasions.

3.9.4. Incorporating global change uncertainties into restoration planning

Some global change mechanisms remain difficult to forecast, notably variations in precipitation, runoff and phreatic levels (Johnson and Weaver, 2009). The impacts will depend on interactions among drivers, and there is a need to quantify the uncertainties inherent to global change (Adger, 2006). Ecological assessments result in uncertainties related to the assessment method, sampling strategy, and seasonal variability (Johnson et al., 2006). Nonetheless, uncertainty will determine the comparability band for the intercalibration of quality assessments (Bennett et al., 2011). Therefore, we need to disentangle uncertainties resulting from current monitoring approaches and those related to global change (Lempert et al., 2004), and to investigate the usefulness of top-down prediction-based assessments for local decision-making (*i.e.* large-scale and remotely obtained indicators and patterns, enabling us to be less dependent from local monitoring data which usually incorporate a large ecological noise resulting from sampling and local biological variability).

3.10. To reinforce the knowledge on relationships between good ecological status, biodiversity and ecosystem's services

The concept of ecosystem services (ES) is widely recognised as having great potential for examining the interaction between ecosystems and human well-being (National Research Council, 2005; Cardinale et al., 2012). Following the Millennium Ecosystem Assessment (MA, 2005) and The Economics of Ecosystems and Biodiversity (TEEB, 2010) report, the ES concept has been taken up more widely in environmental planning and in national and international policy obligations (*i.e.* the MAES initiative, Maes et al., 2012).

The European Biodiversity Strategy to 2020 is clearly a key driver for using the ES approach but other key policy areas such as WFD also provides opportunities. The Blueprint to Safeguard Europe's Water Resources (European Commission, 2012a) describes actions to improve the implementation of Europe's water policies, including the need for cross-cutting problem solving. In this context, the integration of an ecosystem services approach (ESA) in WFD implementation is envisaged to improve the effectiveness of its implementation, as the ESA could help frame the WFD objective of "good status" under a broader social and economic context. Additionally, there are strong links between the concept of ES and several aspects of the WFD, namely the economic analysis, exemptions, concept of water services, pricing and cost-effectiveness of the programmes of measures.

Based on this, the following recommendations can be made.

3.10.1. Enhancing understanding of ecological processes

Biodiversity is considered a key component of ecosystem structure which is essential to maintaining basic ecosystem processes and supporting ecosystem functions. There is a need to develop further research in particular on the links between good ecological status, biodiversity and ecosystem functioning, from both preservation and restoration perspectives. Within this context, developing functional indicators based on fluxes of matters and energy, trophic networks and taking into account biological metrics related to size and other life-history traits are promising paths (see issue 8 related to functional assessment tools). It is increasingly obvious that functional ecology is becoming a fundamental step between the analysis and understanding of biodiversity patterns and ESAs, as it is believed to allow more straightforward inferences on ecosystems' resilience and stability compared to classical taxonomical approaches. Research is also needed to provide knowledge on the impact of multiple drivers on the functional capacity of aquatic systems (such as biodiversity and tipping points) and on the impacts on quality and provision of different ES.

3.10.2. Raising water managers awareness

The concept of ES is still relatively new particularly amongst water policy makers and managers. Simple and concrete guidelines for operational use by water managers are needed to facilitate the application of ESA. In the context of the Biodiversity Strategy 2020 implementation, the MAES (Mapping and Assessment of Ecosystem Services) working group developed an analytical framework for ecosystem assessments (Maes et al., 2013). This work sets out a conceptual framework for mapping and assessment of ES by linking biodiversity to human well-being. The framework builds on the important and commonly accepted conclusions of the Millenium Ecosystem Assessment (2005) and the TEEB (2010) but refocuses the debate on biodiversity as pivotal to delivery of ecosystem services and benefits. The MAES conceptual framework links ecosystems to socio-economic systems *via* the flow of ES, and through the drivers of change that affect ecosystems either as consequence of using the services or as indirect impacts due to human activities. In this context, it must be kept in mind that scale issues are important components when performing an ESA, as any ecosystem assessment should be bounded by spatio-temporal scales that are appropriate to the objectives of local policy makers and natural resource managers. Different types of ecosystem services are valued differently as the spatial and/or temporal scale of the analysis varies, and the spatial fit between the geographical extent of the water resource and the territorial scope of its management institutions is frequently mismatched (Moss, 2012).

4. Synthesis and recommendations

The outcomes of this SPI activity are timely in that they can feed into the revision of the WFD (which is likely to occur in 2019) and highlight the main issues which need to be addressed during future management cycles (Fig. 1). Some urgent concerns were already identified, in particular the need to evaluate uncertainties and to better quantify the good ecological potential of HMWB (Hering et al., 2010). In this context, the European Commission working group ECOSTAT has recently launched an activity focused on the evaluation of good ecological potential, the objective of which is to compare and harmonise the various approaches to the evaluation of HMWB which currently co-exist at the European scale. ECOSTAT also plans to prioritise uncertainty in classification, as this is a recurrent topic raised by water managers. A key aspect of this is the need to take into account the uncertainties associated with the natural variability as this is pivotal for encouraging water managers to adopt ecological rather than chemical evaluation, which is often perceived as more straightforward and therefore more reliable.

An important step in WFD implementation in terms of ecological status evaluation has been taken recently with the intercalibration of 230 methods from 28 countries (European Commission, 2013a). However, further efforts are still required as a significant proportion of methods are still not intercalibrated or remain under development (Birk et al., 2013), and it is therefore essential to complete intercalibration of the remaining methods as soon as possible (the new round of the intercalibration exercise runs from 2014 to 2016). This gap is particularly important for transitional waters, which suffer from multiple stresses, indefinite reference conditions and lack of historical data (e.g. Ban et al., 2010). These limitations also apply for lakes, although historical data may often be more available for these water bodies, facilitating the definition of reference conditions. Significant gaps also exist for countries in Southern and Eastern Europe and specific ecosystem types such as temporary streams or very large rivers (Birk et al., 2012).

Multi-pressure conditions, which are more often the rule than the exception at the European scale, constitute another major issue that is faced by both scientists and water managers. This is especially true for very large rivers, transitional waters and lakes as they usually constitute the focal point of catchment processes (either by their location

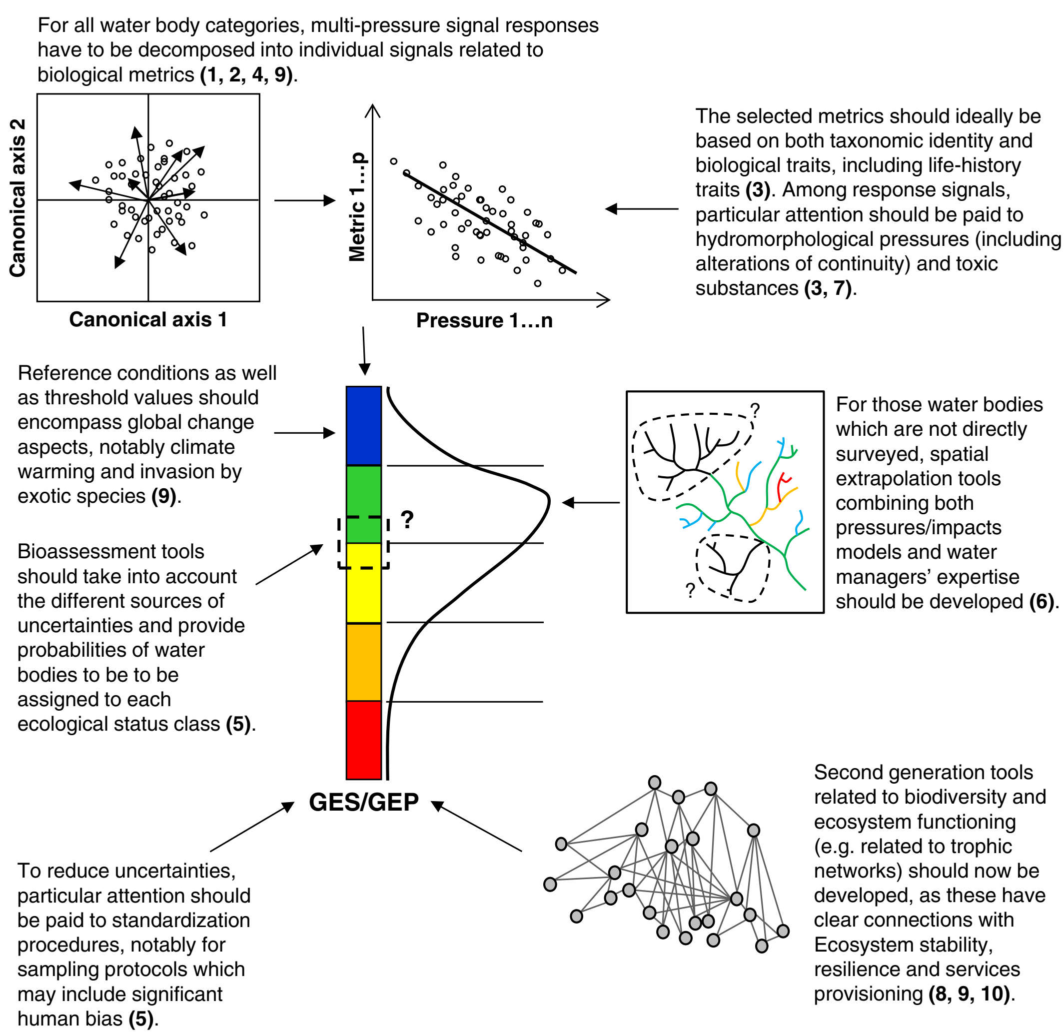

Fig. 1. Conceptual scheme illustrating a proposed way forward with WFD ecological status/potential assessment, following the SPI activity. Numbers in brackets refer to the sections in this paper where the corresponding issues are addressed. GES: good ecological status; GEP: good ecological potential.

downstream of hydrographical basins or because they constitute closed or semi-closed environmental systems in the case of lakes) and are affected by intense hydromorphological alterations (bank alteration by channelisation and/or urbanisation, impoundments, etc.). In this context, specific assessment tools are needed to disentangle the effects of the different pressures acting, and to help water managers prioritizing programmes of measures. This constitutes a major scientific challenge as the effects of these pressures are often difficult to differentiate, and although classical statistical methodologies such as variation partitioning may be useful (Marzin et al., 2012), innovative statistical and experimental approaches may be pushed forward (e.g. Mondy and Usseglio-Polatera, 2013). Developing 'hybrid' Bayesian tools combining classical statistical modelling and expert judgment based on water managers' knowledge offers a useful approach to this aim. Future integrative tools will also have to be developed as part of an integrated management framework and take into account the fluxes of nutrients (N, P) at the hydrographical basin scale, rather than observed concentrations at an individual site at a given time, increasing the value of the tools for water managers. The outputs of the ongoing MARS European research project (Managing Aquatic ecosystems and water Resources under multiple Stress; http://www.mars-project.eu) are expected to be useful for these purposes (Hering et al., 2014).

During the last 10 years and parallel with WFD implementation, '*ecosystem functioning*' has become a priority research avenue (e.g. Naeem et al., 2009). In this context, the development of trait-based multi-metric tools constitutes a promising research area. These tools consider the functional attributes of the flora/fauna, and are specifically designed to assess multi-pressure conditions (Dolédec and Statzner, 2008; Mondy et al., 2012). They will allow ecological status to be related more easily to ecosystem functioning and not only to biological community structures which has often been the case to date, despite the WFD explicitly mentioning both terms (see page 6, definition 21 of the directive: *"Ecological status is an expression of the quality of the structure and functioning of aquatic ecosystems associated with surface waters"*). Integrating size as a functional metric in future bioassessment tools may be a useful first step in this way (San Martin, 2005), especially as it is already required by the WFD for fish, but seldom put into practice. In parallel with trait-based approaches, it is now increasingly recognised that ecosystem functioning, stability and resilience in the face of external pressures can notably be assessed through the study of trophic networks (Arreguín-Sánchez, 2014). This may give birth to a second generation of bioassessment tools likely to provide information about ecosystem functioning as a whole (and not BQE by BQE), with a more integrated perspective, complementing classical taxonomical tools currently used and already intercalibrated. It may be difficult to prescribe these kinds of second generation tools on a regulatory basis in the forthcoming revision of the directive, but MS should be encouraged to use them. Functional indicators such as leaf litter decomposition rate (Gessner and Chauvet, 2002) may also been pushed forward as they may help detecting early signals of anthropogenic degradation, or, conversely, restoration following measures.

Most bioassessment tools developed to date by MS enable water managers to focus on general degradation or organic pollution (e.g. tools based on macroinvertebrates in streams and rivers) or nutrient issues (e.g. tools based on phytoplankton in lakes). Regardless of the water body category under study, more emphasis on the study of the impacts of hydromorphological alterations (bank degradation, density of riparian areas, alteration of natural continuity both in its lateral and longitudinal dimensions, etc.) is urgently needed. In particular, special attention should be paid by all MS to the alteration of longitudinal continuity by the succession of weirs and dams along river courses, as it is acknowledged that these have significant impacts on natural hydrological and thermal regimes, cause deficit of coarse sediments in downstream areas, and participate to the decline of migratory species already at great risk of extinction at the European scale. An important limit of the WFD as it stands is that a river water body may be classified in good ecological status even if the populations of migratory species historically present in the water body have been drastically impacted. The lateral connectivity of fluvial systems, which are usually multi-impacted and whose ecological functioning is known to be complex (Buijse et al., 2005) should also be taken into account in future, particularly as the intercalibration exercise for very large rivers currently being carried focuses only on main channel. Contamination by toxic substances is another pressure category which now urgently needs to be addressed as it is now acknowledged that all aquatic systems are contaminated to some degree at the continental scale (Malaj et al., 2014). The development of related tools will clearly necessitate greater collaborative work as this constitutes the boundary between ecological and chemical status evaluation in the sense of WFD, and ecologists and ecotoxicologists remain unintegrated (Artigas et al., 2012). The requirement to collect robust data regarding sediment contamination by toxic substances, for all water body categories, is a crucial element to support this aim. This is a major challenge as these substances are often present at very low concentrations, but are likely to interact in an additive way. In this context, the SOLUTIONS project (http://www.solutions-project.eu/) aims to deliver a conceptual framework to support the evidence-based development of environmental policies with regard to water quality, and will develop the tools for the identification, prioritisation and assessment of those water contaminants that may pose a risk to ecosystems and human health (Brack et al., 2014).

Of equal importance in terms of river basin management is the adoption of the Ecosystem Services Approach (ESA). This provides a unique opportunity to bring together scientists working across different disciplines together (e.g. ecologists and economists), and provides a very practical and useful common language that can be used independently by politicians, scientists, water managers and citizens, in a sustainable development perspective. In this context, the new research framework for Europe (Horizon 2020—The Framework Programme for Research and Innovation) clearly places emphasis on the necessity to protect biodiversity and ecosystem services, from a global change perspective (European Commission, 2012b). To date, ESA is not clearly used by water managers when designing their programmes of measures, even if some examples do exist (see Wallis et al., 2012 for details). This is especially relevant for wetlands, which are only marginally taken into account in the WFD despite their major importance in purifying water and for biodiversity conservation. More generally, it is important that water managers become able to identify all the main ecosystem services provided by aquatic environments at the basin, sub-basin and water body scales, as this will encourage and facilitate dialogue and compromises among end-users and allow a more integrated management of resources, with both ecological and economical perspectives (Wallis et al., 2012). This offers the opportunity to overcome one of the key barriers to implementation of the WFD and thus also merits special attention in the upcoming revision of the directive.

References

Adger WN. Vulnerability. Glob Environ Chang 2006;16:268–81.

Argillier C, Pronier O, Changeux T. Fishery management practices in French lakes. In: Cowx IG, editor. Management and ecology of lake and reservoir fisheries. Oxford: Blackwell Science; 2002. p. 312–21.

Arreguín-Sánchez F. Measuring resilience in aquatic trophic networks from supply–demand-of-energy relationships. Ecol Model 2014;272:271–6.

Artigas J, Arts G, Babut M, Caracciolo AB, Charles S, Chaumot A, et al. Towards a renewed research agenda in ecotoxicology. Environ Pollut 2012;160:201–6.

Baird DJ, Hajibabei M. Biomonitoring 2.0: a new paradigm in ecosystem assessment made possible by the next generation DNA sequencing. Mol Ecol 2012;21:2039–44.

Ban NC, Alidina HM, Ardron JA. Cumulative impact mapping: advances, relevance and limitations to marine management and conservation, using Canada's Pacific waters as a case study. Mar Policy 2010;34:876–86.

Basset A, Barbone E, Borja A, Brucet S, Pinna M, Quintana XD, et al. A benthic macroinvertebrate size spectra index for implementing the Water Framework Directive in coastal lagoons in Mediterranean and Black Sea ecoregions. Ecol Indic 2012;12:72–83.

Basset A, Barbone E, Borja A, Elliott M, Jona-Lasinio G, Marques JC, et al. Natural variability and reference conditions: setting type-specific classification boundaries for lagoon macroinvertebrates in the Mediterranean and Black Seas. Hydrobiologia 2013;704:325–45.

Behrendt H, Bach M, Kunkel R, Opitz D, Pagenkopf W-G, Scholz GG, et al. Nutrient Emissions into River Basins of Germany on the Basis of a Harmonized Procedure. Umweltbundesamt: Federal Environmental Agency; 2003.
Bennett C, Owen R, Birk S, Buffagni A, Erba S, Mengin N, et al. Bringing European river quality into line: an exercise to intercalibrate macro-invertebrate classification methods. Hydrobiologia 2011;667:31–48.
Bennion H, Battarbee RW, Sayer CD, Simpson GL, Davidson TA. Defining reference conditions and restoration targets for lake ecosystems using palaeolimnology: a synthesis. J Paleolimnol 2011;45:533–44.
Birk S, Bonne W, Borja A, Brucet S, Courrat A, Poikane S, et al. Three hundred ways to assess Europe's surface waters: an almost complete overview of biological methods to implement the Water Framework Directive. Ecol Indic 2012;18:31–41.
Birk S, Willby NJ, Kelly MG, Bonne W, Borjà A, Poikane S, et al. Intercalibrating classifications of ecological status: Europe's quest for common management objectives for aquatic ecosystems. Sci Total Environ 2013(454–455):490–9.
Blanck H. Critical review of procedures and approaches used for assessing pollution-induced community tolerance (PICT) in biotic communities. Hum Ecol Risk Assess 2002;8:1003–34.
Böhmer J, Rawer-Jost C, Zenker A. Multimetric assessment of data provided by water managers from Germany: assessment of several different types of stressors with macrozoobenthos communities. Hydrobiologia 2004;516:215–28.
Borics G, Nagy L, Miron S, Grigorszky I, László-Nagy Z, Lukács BA, et al. Which factors affect phytoplankton biomass in shallow eutrophic lakes? Hydrobiologia 2013;73:93–104.
Borja Á, Basset A, Bricker S, Dauvin J-C, Elliott M, Harrison T, et al. Classifying ecological quality and integrity of estuaries. In: Wolanski E, McLusky DS, editors. Treatise Estuar Coast Sci, 1. Waltham: Academic Press; 2011. p. 125–62.
Borja Á, Dauer D, Elliott M, Simenstad CA. Medium- and long-term recovery of estuarine and coastal ecosystems: patterns, rates and restoration effectiveness. Estuar Coasts 2010;33:1249–60.
Borja Á, Dauer DM, Grémare A. The importance of setting targets and reference conditions in assessing marine ecosystem quality. Ecol Indic 2012;12:1–7.
Brack W, Altenburger R, Schüürmann G, Krauss M, Herráeza DL, van Gilsb J, et al. The SOLUTIONS project: challenges and responses for present and future emerging pollutants in land and water resources management. Sci Total Environ 2014. http://dx.doi.org/10.1016/j.scitotenv.2014.05.143.
Brucet S, Poikane S, Lyche-Solheim A, Birk S. National biological assessment methods for European lakes: ecological rationale and their reflection of human impacts. Freshw Biol 2013;58:1106–15.
Buijse AD, Coops H, Staras M, Jans LH, van Geest GJ, Grift RE, et al. Restoration strategies for river floodplains along large lowland rivers in Europe. Freshw Biol 2002;47:889–907.
Buijse AD, Klijn F, Leuven RSEW, Middelkoop H, Schiemer F, Thorp JH, et al. Rehabilitation of large rivers: references, achievements and integration into river management. Arch Hydrobiol 2005;Suppl. 155:715–38. [Large, Rivers 15].
Cardinale BJ, Emmett Duffy J, Gonzalez A, Hooper DU, Perrings C, Venail P, et al. Biodiversity loss and its impact on humanity. Nature 2012;486:59–68.
Carlisle DM, Wolock DM, Meador MR. Alteration of streamflow magnitudes and potential ecological consequences: a multiregional assessment. Front Ecol Environ 2010;28:1369–77.
Carvalho L, McDonald C, de Hoyos C, Mitschke U, Phillips G, Borics G, et al. Sustaining recreational quality of European lakes: minimizing the health risks from algal blooms through phosphorus control. J Appl Ecol 2013;50:315–23.
Cazemier MM, Querner EP, van Lanen HAJ, Gallart F, Prat N, Tzoraki O, et al. Hydrological analysis of the Evrotas basin, Greece. Alterra Report; 2011. p. 2249.
Datry T, Snelder T, Sauquet E, Pella H, Catalogne C, Lamouroux N. Hydrologie des étiages : typologie des cours d'eau temporaires et cartographie nationale. Rapport technique Onema-Irstea; 2011.
De Lange HJ, Sala S, Vighi M, Faber JH. Ecological vulnerability in risk assessment – A review and perspectives. Sci Total Environ 2010;408:3871–9.
Dolédec S, Statzner B. Invertebrate traits for the biomonitoring of large European rivers: an assessment of specific types of human impact. Freshw Biol 2008;53:617–34.
Donohue I, McGarrigle ML, Mills P. Linking catchment characteristics and water chemistry with the ecological status of Irish rivers. Water Res 2006;40:91–8.
European Commission. Directive 2000/60/EC of the European Parliament and of the Council of 23 October 2000 establishing a framework for Community action in the field of water policy. Off J 2000;L 327:1–72.
European Commission. Common Implementation Strategy for the Water Framework Directive (2000/60/EC). Guidance document n° 13: Overall approach to the classification of ecological status and ecological potential; 2005.
European Commission. A Blueprint to Safeguard Europe's Water Resources. Communication from the Commission to the European Parliament, the Council, the European Economic and Social Committee and the Committee of the Regions. COM, 673. ; 2012a. [final].
European Commission. European Parliament resolution of 20 April 2012 on our life insurance, our natural capital: an EU biodiversity strategy to 2020 (2011/2307(INI)); 2012b [http://ec.europa.eu/environment/nature/biodiversity/comm2006/pdf/EP_resolution_april2012.pdf].
European Commission. Commission Decision of 20 September 2013 establishing, pursuant to Directive 2000/60/EC of the European Parliament and of the Council, the values of the Member State monitoring system classifications as a result of the intercalibration exercise and repealing Decision 2008/915/EC (notified under document C(2013) 5915) (Text with EEA relevance) (2013/480/EU). Off J Eur Union 2013a. [L 266/1–47].
European Commission. Science-policy interface in support of the water framework directive. CIS-SPI Activity report 2010–12; 2013b [http://bookshop.europa.eu/en/science-policy-interface-in-support-of-the-water-framework-directive-pbKI3112744/].
European Environment Agency. European waters—assessment of status and pressures. EEA Report n°8; 2012.
Elliott M, Quintino V. The Estuarine Quality Paradox, Environmental Homeostasis and the difficulty of detecting anthropogenic stress in naturally stressed areas. Mar Pollut Bull 2007;54:640–5.
Elliott M, Whitfield A. Challenging paradigms in estuarine ecology and management. Estuar Coast Shelf Sci 2011;94:306–14.
Fehér J, Gáspár J, Veres KS, Kiss A, Kristensen P, Peterlin M, et al. Hydromorphological alterations and pressures in European rivers, lakes, transitional and coastal waters. ETC./ICM Technical Report n°2; 2012.
Feio MJ, Ferreira J, Buffagni A, Erba S, Dörflinger G, Ferréol M, et al. Comparability of ecological quality boundaries in the Mediterranean basin using freshwater benthic invertebrates. Statistical options and implications. Sci Total Environ 2014;476–477:777–84.
Feld CK, Birk S, Bradley D, Hering D, Kail J, Marzin A, et al. From natural to degraded rivers and back again: a test of restoration ecology theory and practice. Adv Ecol Res 2011;44:120–209.
Frost PC, Cross WF, Benstead JP. Ecological stoichiometry in freshwater benthic ecosystems: an introduction. Freshw Biol 2005;50:1781–5.
Gallart F, Prat N, García-Roger EM, Latron J, Rieradevall M, Llorens P, et al. A novel approach to analysing the regimes of temporary streams in relation to their controls on the composition and structure of aquatic biota. Hydrol Earth Syst Sci 2012;16:3165–82.
García-Roger EM, Sánchez-Montoya MM, Gómez R, Suárez Alonso ML, Vidal-Abarca MR. Do seasonal changes in habitat features influence aquatic macroinvertebrate assemblages in permanent vs temporary Mediterranean streams? Aquat Sci 2011;73:567–79.
Gessner MO, Chauvet E. A case for using litter breakdown to assess functional stream integrity. Ecol Appl 2002;12:498–510.
Gurnell A. Plants as river system engineers. Earth Surf Process Landf 2014;39:4–25.
Hering D, Borja A, Carstensen J, Carvalho L, Elliott M, Feld CK, et al. The European Water Framework Directive at the age of 10: a critical review of the achievements with recommendations for the future. Sci Total Environ 2010;408:4007–19.
Hering D, Borja A, Carvalho L, Feld CK. Assessment and recovery of European water bodies: key messages from the WISER project. Hydrobiologia 2013;704:1–9.
Hering D, Carvalho L, Argillier C, et al. Managing aquatic ecosystems and water resources under multiple stress—an introduction to the MARS project Science of the Total Environment; 2014. http://dx.doi.org/10.1016/j.scitotenv.2014.06.106.
Hughes R. Defining acceptable biological status by comparing with reference conditions. In: Davis W, Simon T, editors. Biological assessment and criteria: tools for water resource planning and decision making. Boca Raton, FL: CRC Press; 1995. p. 31–47.
Hume TM, Snelder T, Weatherhead M, Liefting R. A controlling factor approach to estuary classification. Ocean Coast Manag 2007;50:905–29.
Jeppesen EB, Kronvang JE, Olesen J, Audet J, Søndergaard M, Hoffmann CC, et al. Climate change effects on nitrogen loading from cultivated catchments in Europe: implications for nitrogen retention, ecological state of lakes and adaptation. Hydrobiologia 2011;663:1–21.
Johnson R, Hering D, Furse MT, Clarke RT. Detection of ecological change using multiple organism groups: metrics and uncertainty. Hydrobiologia 2006;566:115–37.
Johnson TE, Weaver CP. A framework for assessing climate change impacts on water and watershed systems. Environ Manag 2009;43:118–34.
Jørgensen SE. Exergy and ecology. Ecol Model 1992;63:185–214.
Kennard MJ, Arthington AH, Pusey BJ, Harch BD. Are alien fish a reliable indicator of river health? Freshw Biol 2005;50:174–93.
Kernan M, Moss B, Battarbee RW. Climate change impacts on freshwater ecosytems. Oxford: Wiley Blackwell; 2010.
Krasznai E, Borics G, Várbíro G, Abonyi A, Padisák J, Deák C, et al. Characteristics of pelagic phytoplankton in shallow oxbows. Hydrobiologia 2010;639:173–84.
Kristensen E, Baattrup-Pedersen A, Andersen H. Prediction of stream fish assemblages from land use characteristics: implications for cost-effective design of monitoring programmes. Environ Monit Assess 2012;184:1435–48.
Lake PS. Drought and aquatic ecosystems: effects and responses. Chichester, UK: Wiley-Blackwell; 2011.
Lake PS, Bond N, Reich P. Linking ecological theory with stream restoration. Freshw Biol 2007;52:597–615.
Larned ST, Datry T, Arscott DB, Tockner K. Emerging concepts in temporary-river ecology. Freshw Biol 2010;55:717–38.
Lempert R, Nakicenovic N, Sarewitz D, Schlesinger M. Characterizing climate-change uncertainties for decision-makers. Climate Change 2004;65:1–9.
Leprieur F, Beauchard O, Blanchet S, Oberdorff T, Brosse S. Fish invasions in the world's river systems: when natural processes are blurred by human activities. PLoS Biol 2008;6:e28. http://dx.doi.org/10.1371/journal.pbio.0060028.
Lindeman RL. The trophic–dynamic aspect of ecology. Ecology 1942;23:399–418.
Logez M, Pont D. Global warming and potential shift in reference conditions: the case of functional fish-based metrics. Hydrobiologia 2013;704:417–36.
Lyche-Solheim A, Feld CK, Birk S, Phillips G, Carvalho L, Morabito G, et al. Ecological status assessment of European lakes: a comparison of metrics for phytoplankton, macrophytes, benthic invertebrates and fish. Hydrobiologia 2013;704:57–74.
Lyche-Solheim A, Rekolainen S, Moe SJ, Carvalho L, Philipps G, Ptacnik R, et al. Ecological threshold responses in European lakes and their applicability for the Water Framework Directive implementation: synthesis of lakes results from the REBECCA project. Aquat Ecol 2008;42:317–34.
Maes J, Egoh B, Willemen L, Liquete C, Vihervaara P, Schägner JP, et al. Mapping ecosystem services for policy support and decision making in the European Union. Ecosyst Serv 2012;1:31–9.

Maes J, Teller A, Erhard M, Liquete C, Braat L, Berry P, et al. Mapping and Assessment of Ecosystems and their Services. An analytical framework for ecosystem assessments under action 5 of the EU biodiversity strategy to 2020. Luxembourg: Publications office of the European Union; 2013.

Malaj E, von der Ohe PC, Grote M, Kühne R, Mondy C, Usseglio-Polatera P, et al. Organic chemicals jeopardize the health of freshwater ecosystems on the continental scale. Proc Natl Acad Sci 2014. http://dx.doi.org/10.1073/pnas.1321082111.

Margalef R. Information and uncertainty in living systems, a view from ecology. Biosystems 1996;38:141–6.

Marzin A, Archaimbault V, Belliard J, Chauvin C, Delmas F, Pont D. Ecological assessment of running waters: do macrophytes, macroinvertebrates, diatoms and fish show similar responses to human pressures? Ecol Indic 2012;23:56–65.

Marzin A, Delaigue O, Logez M, Belliard J, Pont D. Uncertainty associated with river health assessment in a varying environment: the case of a predictive fish-based index in France. Ecol Indic 2014;43:195–204.

Masson M, Blanc G, Schafer J. Geochemical signals and source contributions to heavy metal (Cd, Zn, Pb, Cu) fluxes into the Gironde Estuary via its major tributaries. Sci Total Environ 2006;370:133–46.

Meiner A. Integrated maritime policy for the European Union—consolidating coastal and marine information to support maritime spatial planning. J Coast Conserv 2010;14:1–11.

Millenium Ecosystem Assessment. Ecosystems and human well-being: synthesis. Washington DC: Island Press; 2005.

Mjelde M, Hellsten S, Ecke F. A water level drawdown index for aquatic macrophytes in Nordic lakes. Hydrobiologia 2013;704:141–51.

Mondy CP, Usseglio-Polatera P. Using conditional tree forests and life history traits to assess specific risks of stream degradation under multiple pressure scenario. Sci Total Environ 2013;461:750–60.

Mondy CP, Villeneuve B, Archaimbault V, Usseglio-Polatera P. A new macroinvertebrate-based multimetric index (I2M2) to evaluate ecological quality of French wadeable streams fulfilling the WFD demands: a taxonomical and trait approach. Ecol Indic 2012;18:452–67.

Moss T. Spatial fit, from Panacea to practice: implementing the European Water Framework Directive. Ecol Soc 2012;17:2.

Munné A, Prat N. Effects of Mediterranean climate annual variability on stream biological quality assessment using macroinvertebrate communities. Ecol Indic 2011;11:651–62.

Biodiversity, ecosystem functioning, and human wellbeing: an ecological and economic perspective. In: Naeem S, Bunker DE, Hector A, Loreau M, Perrings C, editors. Oxford, United Kingdom: Oxford University Press; 2009.

Nestler JM, Theiling CH, Lubinski KS, Smith DL. Reference condition approach to restoration planning. River Res Appl 2010;26:1199–219.

Nõges P, van de Bund W, Cardoso AC, Solimini AG, Heiskanen A-S. Assessment of the ecological status of European waters: a work in progress. Hydrobiologia 2009;633:197–211.

National Research Council. Valuing ecosystem services: towards better environmental decision making. Washington DC: National Academies Press; 2005.

Poff NL, Allan JD, Bain MB, Karr JR, Prestegaard KL, Richter BD, et al. The natural flow regime. Bioscience 1997;47:769–84.

Pont D, Hugueny B, Beier B, Goffaux D, Melcher A, Noble R, et al. Assessing river biotic condition at a continental scale: a European approach using functional metrics and fish assemblages. J Appl Ecol 2006;73:70–80.

Prigogine I. Structure, dissipation and life. In: Marois M, editor. Theoretical physics and biology. Amsterdam: North-Holland; 1969. p. 23–52.

Prat N, Gallart F, Von Schiller D, Polesello S, García-Roger EM, Latron J, et al. The mirage toolbox: an integrated assessment tool for temporary streams. River Res Appl 2014. http://dx.doi.org/10.1002/rra.2757.

Rahel FJ. Biogeographic barriers, connectivity and homogenization of freshwater faunas: it's a small world after all. Freshw Biol 2007;52:696–710.

Sala O, Chapin III FS, Armesto JJ, Berlow E, Bloomfield J, Dirzo R, et al. Global biodiversity scenarios for the year 2100. Science 2000;287:1770–4.

San Martin E. Latitudinal variation in plankton size spectra along the Atlantic Ocean. Ph.D. thesis UK: University of Southampton; 2005.

Santos JM, Silva A, Katopodis C, Pinheiro P, Pinheiro A, Bochechas J, et al. Ecohydraulics of pool-type fishways: getting past the barriers. Ecol Eng 2012;48:38–50.

Schäfer RB, Kefford BJ, Metzeling L, Liess M, Burgert S, Marchant R. A trait database of stream invertebrates for the ecological risk assessment of single and combined effects of salinity and pesticides in South-East Australia. Sci Total Environ 2011;409:2055–63.

Sousa Leitao FM, Gaspar MB. Immediate effect of intertidal non-mechanised cockle harvesting on macrobenthic communities: a comparative study. Sci Mar 2007;71:723–33.

Stearns SC. The evolution of life histories. US, Oxford: Oxford University Press; 1992.

Steward AL, Marshall JC, Sheldon F, Harch B, Choy S, Bunn SE, et al. Terrestrial invertebrates of dry river beds are not simply subsets of riparian assemblages. Aquat Sci 2011;73:551–66.

Stoddard JL, Larsen DP, Hawkins CP, Johnson RK, Norris RH. Setting expectations for the ecological conditions of streams: the concept of reference condition. Ecol Appl 2006;16:1267–76.

Strayer DL. Alien species in fresh waters: ecological effects, interactions with other stressors, and prospects for the future. Freshw Biol 2010;55:152–74.

TEEB. The economics of ecosystems and biodiversity ecological and economic foundations. In: Kumar Pushpam, editor. London and Washington: Earthscan; 2010.

Thrush SF, Hewitt JE, Hickey CW, Kelly S. Multiple stressor effects identified from species abundance distributions: interactions between urban contaminants and species habitat relationships. J Exp Mar Biol Ecol 2008;366:160–8.

Vaughan IP, Diamond M, Gurnell AM, Hall KA, Jenkins A, Milner NJ, et al. Integrating ecology with hydromorphology: a priority for river science and management. Aquat Conserv Mar Freshwat Ecosyst 2009;19:113–25.

Verberk WCEP, van Noordwijk CGE, Hildrew AG. Delivering on a promise: integrating species traits to transform descriptive community ecology into a predictive science. Freshwater Sci 2013;32:531–47.

Verdonschot PFM, Spears BM, Feld CK, Brucet S, Keizer-Vlek H, Borja A, et al. A comparative review of recovery processes in rivers, lakes, estuarine and coastal waters. Hydrobiologia 2013;704:453–74.

Villeneuve B. Extrapolation spatiale de l'état écologique à partir de données d'occupation du sol, de physicochimie et d'hydromorphologie à l'échelle des réseaux de suivi nationaux. Rapport technique Onema-Cemagref; 2010.

Wallis C, Seon-Massin N, Martini F, Schouppe M. Implementation of the Water Framework Directive—when ecosystem services come into play. Onema Meetings Recap; 2012.

Whitehead PG, Johnes PJ, Butterfield D. Steady state and dynamic modelling of nitrogen in the River Kennet: impacts of land use change since the 1930s. Sci Total Environ 2002;282:417–34.

Wilby RL, Orr HG, Hedger M, Forrow D, Blackmore M. Risks posed by climate change to the delivery of Water Framework Directive objectives in the UK. Environ Int 2006;32:1043–55.

Xu FL, Jorgensen SE, Tao S. Ecological indicators for assessing freshwater ecosystem health. Ecol Model 1999;116:77–106.

Xu FL, Wang JJ, Chen B, Qin N, Wu W-J, He W, et al. The variations of exergies and structural exergies along eutrophication gradients in Chinese and Italian lakes. Ecol Model 2011;222:337–50.

Young RG, Collier KJ. Contrasting responses to catchment modification among a range of functional and structural indicators of river ecosystem health. Freshw Biol 2009;54:2155–70.

Yvon-Durocher G, Allen AP, Montoya JM, Trimmer M, Woodward G. The temperature dependence of the carbon cycle in aquatic ecosystems. Integrative Ecol 2010;43:267–313.

Yvon-Durocher G, Montoya JM, Trimmer M, Woodward G. Warming alters the size spectrum and shifts the distribution of biomass in freshwater ecosystems. Glob Chang Biol 2011;17:1681–94.