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A guide for assessing the hydromorphological quality of natural and artificial lakes

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Contents

Abstract

This report provides a guide for assessing the hydromorphological quality of natural and artificial lake water bodies in accordance with the requirement of the Water Framework Directive (WFD).

First, 60 metrics are described for characterizing the mandatory lake hydromorphological quality elements comprising those that are hydrological (i.e., quantity and dynamics of water flow, residence time, connection to groundwater body) and those that are morphological (i.e., lake depth variation, quantity, structure and substrate of the lake bed, structure of the lake shore). The data required for their calculation, as well as tools and techniques for acquiring them (e.g., field survey, models, remote sensing …), with examples of their application are also provided. Advice is also given on how to define reference conditions and combine metrics to develop a lake hydromorphological index that complies with the WFD.

In the final section, a brief review of the literature presents current knowledge on the links between hydromorphology and biology. The document then focuses on recommendations on how to develop WFD-compliant methods addressing hydromorphology in assessing the ecological status of lakes. Some existing methods are described to provide practical examples.

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1 Introduction

1.1 Aim of the guide

The long-term goal of the Water Framework Directive (WFD; European Parliament, Council of the European Union, 2000) is to ensure sustainable water resource use in the European Union. This involves establishing objectives which, if achieved, will create conditions that are consistent with sustainable water resources. The criteria (set out in Annex V of the WFD), against which the objectives are to be judged, are collectively termed “ecological status”, with “good ecological status” (GES: defined as no more than a slight change from that expected under unimpaired conditions) being the threshold for sustainable water resources. The criteria include biological quality elements (BQEs, e.g., invertebrates, phytoplankton, fish), physico-chemical and hydromorphological (HYMO) quality elements. This means that all countries have to develop hydromorphological assessment methods and use them in the classification of ecological status. However, many countries have not yet developed these for lakes, and those who have done use a wide range of different methods, most of which are not linked to BQEs (Poïkane et al., 2020b; Argillier et al., 2023). To overcome this problem, the working group ECOSTAT lake hydromorphological assessment has been established to exchange best practices and facilitate consistent and comparable HYMO assessment methods.

This report provides a guide for assessing the hydromorphological quality of lake water bodies, focusing on 1) how to describe hydromorphological alterations to lakes and develop HYMO indices, 2) which methods allow the collection of relevant hydromorphological information, and 3) how to develop WFD-compliant methods for using hydromorphology when assessing the ecological status of lakes. These questions are addressed not only for natural lakes, but also for non-natural lakes. It has been produced on the basis of the knowledge and expertise of the contributors, as well as information gathered during exchanges with experts commissioned by the EU Member States.

1.2. General trends

The first civilizations developed along rivers or near other water bodies, precisely because water has always represented an important resource for life and human activities (Grindlay et al., 2011). The development of civilizations subsequently led to active management of water through the creation of canals to improve the fertility of the land and to facilitate travel and trade, and to the construction of dams to have water available even in dry periods for irrigation, domestic water supply, for flood protection and subsequently for the production of hydropower (Gleick, 2003). However, management of water resources and territorial development did not proceed in harmony. The traditional hydraulic perspective of water resource use has caused a lack of spatial sustainability in water planning and management (Grindlay et al., 2011). Increasing demands for water over recent decades have led to excessive water use and a consequential degradation in water resources and adverse impacts on aquatic ecosystems that are exacerbated by pollution from human activities such as land use. Integrated Water Resource Management (IWRM) can be a new paradigm of water resource management. This is based on the proposition that only through recognizing the close interrelationships of biophysical, social, economic and political factors is it possible to define such management in a socially and ecologically sustainable way, using an interdisciplinary approach (Grindlay et al., 2011; Ciampittiello et al., 2024). In the light of IWRM as a new paradigm, the WFD can be considered as precursor legislation that aims to protect and improve aquatic ecosystems and to define sustainable management of water resources throughout Europe (Carvalho et al., 2019).

Hydromorphological pressures, i.e. the human activities that bring about alteration to the hydrology and morphology of water bodies are the second most prevalent human stressors influencing aquatic ecosystems, after eutrophication (Fehér et al., 2012). These pressures include flow regulation by dams, weirs, sluices, locks, water diversions and abstractions, landscape alterations caused by canalization and river channel straightening, and the disconnection of natural watercourses from their floodplains (Poïkane et al., 2020b). Hydromorphological pressures reduce the structural complexity and heterogeneity of river and lake habitats (riparian, shore and littoral), altering natural flow regimes, and consequently affect ecological integrity and biological communities. In addition, hydromorphological pressures can alter lake mixing and thermal stratification regimes. This may have impacts on the internal cycling of nutrients, often favouring the colonization and establishment of alien species, with the probability of causing widespread alterations to the entire lake ecosystem (Poïkane et al., 2020b). Such exemples of stressors and their impacts are given in Ostendorp et al., 2004 (Table 1).

1.3. The lakes in this context

Without reference to criteria of size or origin, lakes account for around 90 % of the world's freshwater surface area in liquid form (Duker & Borre, 2001). It is estimated that 130,000 km^3 of fresh water are stored in lentic environments, i.e., around 33 times more than in fluvial systems. These environments have multiple ecological, economic and social functions, which are often closely interlinked. In addition to their role as freshwater reservoirs for drinking water supply, irrigation and industry, they are important regulators of the carbon, nitrogen and phosphorus cycles. They help to regulate water flows, purify water and recharge aquifers, and transport people and goods. Lakes are home to an abundance of flora and fauna, and numerous endemic species in a variety of taxa have been recorded from lake water bodies. Lakes are inhabited by an array of aquatic organisms and may host extremely high biodiversity. They also serve as temporary habitats for numerous species, including insects, amphibians, and birds, thus contributing to the maintenance of biodiversity in terrestrial environments. In addition, certain species, especially fish, that inhabit lakes are heavily exploited commercially, providing essential nutrition to local communities or the region. Consequently, lakes play a vital role in the economic and social development of these areas. From this perspective, the tourist appeal of these sites – particularly for boating and fishing – is also significant. This diversity of uses highlights the complexity of the issues associated with protection and restoration of these systems. According to recent estimates, species populations in fresh waters are more sensitive to human pressures and imperilled more rapidly than those in terrestrial and marine biomes (e.g., Reid et al., 2019). Therefore, biodiversity losses and the risks of species extinctions in freshwater ecosystems are often disproportionately high compared with those in terrestrial and marine realms. In Europe, consideration of all the ecological, economic and social issues associated with water bodies is reflected in several texts and in particular in the European WFD, which sets water quality objectives for each Member State (MS) (Directive 2000/60/EC, European Parliament, Council of the European Union, 2000). The major changes brought about by this legislation lie in the fact that all dimensions of ecosystems are taken into account in the assessment of their status, including hydromorphology.

Table 1. Examples of human activities and driving forces, pressures related to them, and possible impacts taken.

Human activities	Pressure	Impact
Water use (water supply, navigation, flood protection, hydropower generation)	Dams, weirs, reservoirs and flow regulation, straightening of river channels, extraction of sand and gravel from stream beds.	Deterioration of the ecological quality of the littoral zone, increase in severity of flood waves (esp. barriers with no storage capcity) and higher water level with decrease in biodiversity (macrophytes, benthic macroinvertebrates, fish), sediment deficit in the shore zone, bank erosion, modification of natural water level.
Water infra-structures for agriculture	Drainage of hydric soils (nutrient mineralisation), abstraction of water (irrigation).	Eutrophication, lowering of water level, change in hydrological regime.
Agricultural	Soil erosion and solid matter inflow, inflow of dissolved and particulate nutrients, input of pesticides and other agro-chemicals, cattle grazing.	Excessive sedimentation (riparian wetlands, helophytes), eutrophication (riparian wetlands, macrophytes, benthic algae), toxic effects, trampling and grazing.
Road building	Direct destruction of habitats, land fill-up, shore enforcement, release of hazardous substances.	Loss in total area of habitats fragmentation and disintegration of lakeshore zonation, modification of wave characteristics, erosive forces and longshore transport of sediment matter, toxic effects and mechanical damage during decontamination.
Urbanization and industrialization	Land reclamation and erosion defense constructions, long-shore and cross-shore constructions, waste disposal, contaminated landfills marinas, landing places, harbours, loading bridges, housing and commercials estates, sealing of the ground, urban drainage, storm water and emergency overflows, inflow of untreated/treated sewage.	Direct destruction and isolation of habitats, sediment loss, erosion or excessive sedimentation of organic matter, hygienic problems (e.g., fecal bacteria). Deleterious effects on the biota, decomposable organic matter, oxygen deficits, nutrient inflow.
Sediment extraction	Onshore pits, bedload dredging in river delta channels, sublacustrine dredging of landing places and harbour channels.	Enhancement of wave energy to the shore, sediment matter deficits in the shorezone, sediment erosion from neighbouring areas and silting.
Recreation	Camping places, bathing beaches, boating, surfing, mooring, landing places, jetties and platforms.	Direct destruction of habitats, disturbance of breeding birds and waterfowl modification of habitat suitability.
Navigation	Shore development and infrastructure, maintenance, invasive alien species, contamination with hazardous chemicals.	Direct destruction of habitats, sediment erosion from neighbouring areas and silting, out-competing of native species, effects on the whole ecosystem functioning (food web).
Professional and recreational fisheries	Selective removal of species or age classes, artificial stocking of fish species, introduction of alien genotypes and fish species.	Modification of fish population dynamics and food web genetic mixing, extinction of local genotypes competition with native species.
Hunting of water fowl	Disturbance of resting waterfowl, contamination by lead from hail-shot.	Disturbance of resting waterfowl, reduction of body fat reserves of migrating bird, toxic effects to benthivore waterfowl.

Source: Ostendorp et al., 2004

1.4. Regulatory requirement and policy

The WFD describes water as a common resource that is not a commercial product like others, but rather a heritage that must be protected, defended and treated as such, so that future generations can benefit from an intact environmental heritage. It is on this last point that the objective of the WFD to protect the status of aquatic ecosystems, including rivers, lakes, transitional waters, coastal waters, and groundwater, is founded (European Parliament, Council of the European Union, 2000). To assess progress towards this objective, the WFD mandates the development of classification schemes for the ecological status or ecological potential of surface water bodies. This assessment considers specific elements such as biological, hydromorphological, physico-chemical and chemical quality. According to Annex V of the WFD the hydromorphological condition of lakes is evaluated by six parameters related to hydrological regime and morphological conditions:

- Quantity and dynamics of water flow
- Residence time
- Connection to groundwater body
- Lake depth variation
- Quantity, structure and substrate of the lake bed
- Structure of the lake shore.

At the same time, faced with continuing water-related crises, the European Union needs to become water resilient by preparation, adaptation, and by sustaining healthy freshwater ecosystems, moving away from managing crises towards proactively managing risks. In 2024, the European Commission launched a non-legislative initiative for Water Resilience (https://www.eesc.europa.eu/en/our-work/opinions-information-reports/opinions/initiative-water-resilience). It includes a number of immediate steps and opens a public debate about achieving water resilience. Furthermore, there is a growing need to integrate further the protection and sustainable management of water resources through other policies and areas such as energy, transport, agriculture, fisheries and tourism. A new integrated vision is needed of the management of natural resources that takes into account each regulated aspect not separately but together in which, like individual pieces of a jigsaw puzzle, each piece supports the others. It is precisely from this perspective that broader regulations such as the 2030 Agenda for Sustainable Development[1] and the European Green Deal[2] are also framed.

The 2030 Agenda for Sustainable Development[1] is a programme of action for people, planet and prosperity signed in September 2015 by the governments of the 193 UN member countries. It includes 17 Sustainable Development Goals (SDGs). These 17 goals are all interconnected but there are specific ones on water, its use and preserving its ecological quality and quantity: Goal 6. Ensure the availability and sustainable management of water and sanitation for all; goal 13. Take urgent action to combat climate change and its impacts.

[1] United Nations, Transforming our world: the 2030 Agenda for Sustainable Development.

[2] Communication from the Commission to the European Parliament, the European Council, the Council, the European Economic and Social Committee and the Committee of the Regions. 2019. The European Green Deal COM/2019/640 final.

The European Green Deal[2] is a package of strategic initiatives that aims to put the European Union on the path to a green transition, with the ultimate goal of achieving climate neutrality by 2050. It highlights the need for a holistic and cross-sectoral approach in which all relevant policy sectors contribute to the ultimate climate objective. The package includes initiatives regarding climate, environment, energy, transport, industry, agriculture and sustainable finance, all highly interconnected sectors. The European Green Deal builds on the United Nations 2030 Agenda, of which it is an integral part, but identifies additional, more ambitious objectives.

The EU Biodiversity Strategy for 2030[3], as a core part of the Green Deal, aims to stimulate biodiversity recovery throughout Europe. This emphasizes that protecting and restoring biodiversity and well-functioning ecosystems is key to boost resilience and prevent the emergence and spread of future diseases. Biodiversity action is directed, inter alia, with a view to restoring freshwater ecosystems. In particular, according to the EC Communication on Biodiversity Strategy[3], it is recognized that greater efforts are needed to restore freshwater ecosystems and the natural functions of rivers in order to achieve the objectives of the WFD. In addition, the Biodiversity Strategy states that MS authorities should review water abstraction and impoundment permits to implement ecological flows in order to achieve good ecological status or good ecological potential of all surface waters by 2027 at the latest, as required by the WFD. Overall, it is recognized that investing in large-scale river and floodplain restoration can improve water regulation, flood protection, and nursery habitats for fish, as well as helping to reduce nutrient pollution.

As foreseen by the EU Biodiversity Strategy for 2030, the Regulation (EU) 2024/1991 of the European Parliament and of the Council of 24 June 2024 on nature restoration[4] has been published and entered into force in August 2024. The Regulation sets out new rules to restore degraded habitats in the EU to enable long-term and sustainable recovery of nature. The restoration of freshwater ecosystems is considered a priority as they support the achievement of favourable conservation status for river, lake, alluvial and riparian habitats and species living in those habitats protected by the Habitats and Birds Directives (Directives 92/43/EEC[5] and 2009/147/EC respectively[6]). Some of those habitats and species of EU interest are interlinked with lake hydromophological conditions; they may be found in lake riparian and littoral zones and thus form part of the structure of lake shores, and/or their conservation status depends upon the status of hydromophological conditions (e.g., fish spawning habitats in relation to lake water level fluctuations). Moreover, the characterization of lake hydromorphological conditions may provide a benchmark, against which the effectiveness of nature restoration measures can be assessed.

Furthermore, the European Commission also recognizes water scarcity and drought as a priority in the European Green Deal and this is reflected in several major European strategies such as the 2021 EU Strategy on Adaptation to Climate Change (https://climate.ec.europa.eu/eu-action/adaptation-climate-change_en) and the Biodiversity Strategy for 2030. It is worth noting that

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[3] Communication from the Commission to the European Parliament, the Council, the European Economic and Social Committee and the Committee of the Regions. 2020. EU Biodiversity Strategy for 2030 Bringing nature back into our lives. COM/2020/380 final

[4] Regulation (EU) 2024/1991 of the European Parliament and of the Council of 24 June 2024 on nature restoration and amending Regulation (EU) 2022/869 (Text with EEA relevance). Official Journal L, 2024/1991, 29.7.2024, p. 1-93.

[5] Council Directive 92/43/EEC of 21 May 1992 on the conservation of natural habitats and of wild fauna and flora. Official Journal L 206, 22.7.1992, p. 7–50.

[6] Directive 2009/147/EC of the European Parliament and of the Council of 30 November 2009 on the conservation of wild birds (Codified version). Official Journal L 20, 26.1.2010, p. 7–25.

as WFD is the basic framework for water policy, it addresses the issue of water resilience in the face of climate change. Water scarcity and drought affect the hydrological regime of lakes (quantity and dynamics of water flow, level, residence time, connection to groundwaters). In the context of the WFD, the Working Group Water Scarcity and Drought provides related guidance to address this issue, including the update of the Guidance document No. 24 River Basin Management in a Changing Climate (European Commission: Directorate-General for Environment, 2009). The update makes recommendations such as to avoid structural changes of natural water bodies as much as possible, as the greatest resilience to face climate change is provided by natural systems, to stimulate the uptake of nature-based solutions in planning and implementation, and to enforce sustainable and cooperative water allocation within and across basins. These recommendations are linked to both the hydrological regime and morphological conditions of lakes.

Based on what is stated above it is clear that the management of water resources is not an isolated issue that solely concerns the quantity of water available for human uses and for economic activities such as agriculture, industry and hydroelectric energy. Water management represents a crossroads, a crucial point that requires choices and the implementation of actions. These actions should have large-scale, spatial and temporal repercussions such as on ecosystems and the services they offer human society free of charge: on human health and the safety of populations, on social and cultural values of a territory or a nation, on the ability or otherwise to be resilient (i.e., the ability of an ecosystem to resist and recover after stress or substantial change) and to be able to adapt to continuing climate changes. This had already been understood by the legislators of the WFD which still today represents an important point of reference especially with regard to the hydromorphological status of water bodies.

Another important element resulting from the requirements of the WFD, especially in reference to the status of water bodies, is the evaluation of ecohydrology, understood as a subdiscipline of hydrology that deals with the ecological processes present within the hydrological cycle and aims to use these processes to improve environmental sustainability. Greater efforts are needed to implement the WFD in a more economically sustainable way. This requires a deeper interdisciplinary undestanding of the regulatory mechanisms within ecosystems that will help to achieve sustainable development (Zalewski, 2011).

2. Metrics for characterizing hydromorphological alterations of lakes

Early attempts to characterize hydrology or morphology focused on single features such as lake shape, slope, shore substrate structure, habitat heterogeneity, bathymetry, depth, and hydrological regime (Håkanson, 1981). At the end of the 1990s, Baker et al. (1997) developed a protocol in the USA for collecting data on water quality, ecological variables and physical structure, including physical and chemical measurements within lakes, as well as for shore zone habitat survey, that was similar to the approach of the WFD. Field sampling and data collection were transformed into numerical equivalents of the quality of habitat and morphological modification.

Under the WFD, hydrological and morphological aspects became new entities for which studies, methods and models were developed (Ciampittiello et al., 2017). In practice, the WFD helped to elevate the importance of hydromorphology in water management practices by requiring MSs to assess and manage the HYMO conditions of water bodies as part of their overall river basin management plans. This includes assessing the following features: quantity and dynamics of water flow, level, residence time, connection to groundwaters, lake depth variation, substrate of the lake bed, and structure and condition of the lake shore to support the ecological objectives of the Directive.

Among the metrics listed in this chapter, most of them are directly related to one WFD feature whereas others can provide information on several features. Some of them relate to processes that affect different HYMO parameters and can be considered as a proxy of the HYMO features. For some metrics, in particular those describing hydrological regime, climate change must be considered as a factor likely to interact with other human stressors and influence their measurement, therefore it is important to distinguish the two impacts (i.e. those linked to climate and others).

In the following sections, the vocabulary and features described are in accordance with the two European standards related to lake hydromorphology (CEN, 2011; 2017; Boon et al., 2019).

2.1. Change in water flow

This feature is related to the WFD parameter 'quantity and dynamics of water flow'. Changes in water flow can result from inflow or outflow alterations. These changes can be measured directly or indirectly in a quantitative way or by a qualitative approach.

- Changes of the water balance

Data required: several models can be used to calculate this metric. In general, they include several parameters: inflow to and outflow from the water body, volume of storage, area of the lake, precipitation, evaporation, interaction of the lake with groundwater, etc. (see section 3.4)

- Changes of flux in the tributaries at the entrance to or at the outlet from the lake before and after human impacts.

Data required: to calculate this quantitative change in volume flow, it is necessary (1) to have hydrometric monitoring stations at the entrance to the lakes or their outlets, with time series covering the period before and after alterations, or (2) to use models powered by different information (e.g. precipitations, catchment size; see section 3.4). Changes can be calculated annually, seasonally, monthly or daily depending on the size of the lake and its catchment area and the extent of the impact.

An indirect quantitative assessment is also possible by taking into account changes in the flow entering the catchment owing to the presence of artificial structures on the tributary, or by water abstraction. Several metrics can be used in relation to this feature:

- Relative area of upstream catchment impounded by any dams/structures
- Number of hydrological structures and upstream impoundments.
- Rate of natural tributaries (as opposed to artificial channels) where there are alterations to stream morphology and longitudinal profile, considering the importance of each tributary in the river system as defined by Strahler rank (Carrière et al., 2023).
- Density of obstacles in the catchment and proportion of the upstream river system that is impeded by at least one obstacle. In order to take account of the differences in impact between weirs and dams due to the height of the obstacle, a weighting relative to the nature of each obstacle can be applied.
- Sum of all upstream reservoir volumes divided by the average annual inflow into the lake to be classified. The volume of the lake to be classified should not be included.

Data required: maps with a delineation of the lake catchment and position of barriers on the tributaries. Knowledge on damming tributary rank or barrier size can give more accurate information on changes in inflow volume. For the last metric, in addition to the maps showing the location of the reservoirs, data are required on the volume of the upstream reservoirs and flux measurements on the tributaries where lakes or reservoirs are located, measured at the entrance to the lake.

- Size of catchment area transferred in/out of the catchment under consideration.

Data required: maps with a delineation of the lake catchment, including the points on the watercourse where hydrological transfers between catchments occur to allow subtraction of the part of the catchment subjected to water abstraction.

- Qualitative assessment of inflow changes through expert knowledge of the presence, absence, intensity of human alteration of water discharge into the lake.

Data required: expert knowledge of the human activities in the catchment and of their possible impact on water inflow, is obligatory. Knowledge of these alterations in the catchment of other lakes is useful to enable a comparison and provide a “ranking” of the flow alteration for different lakes.

- Qualitative assessment of outflow changes through expert knowledge on the presence, absence, intensity of abstraction from the lake.

Data required: no data but expert knowledge of the human activities on the lake and of their possible impact on water outflow is obligatory. Knowledge of these alterations on other lakes is useful to enable a comparison and provide a “ranking” of the flow alteration for different lakes. Some field data (presence, number or importance of pumping structures for example) can help to qualify the intensity of this change.

It should be noted that information on the size, construction and operation of reservoirs upstream of the site for measuring the change in water flow also provides information on the degree of change in sediment transport.

2.2. Change in periodicity and seasonality of inflow

In addition to changes in volume, alteration of flow may be observed in the periodicity and seasonality of exchanges between lakes and the connected rivers. Therefore, the following metrics are proposed for assessing the WFD parameter 'quantity and dynamics of water flow'.

- Percentage of days with annual minimum level in spring and summer.
- Percentage of days with annual maximum level in autumn and winter.
- Average deviation in metres during the winter period or the summer period between the current mean water level and the unregulated mean water level (according to the reference ratio). The period with the worst status indicates the status of deviation in winter or summer water levels. Status classification should be performed for the entire surface water body as a unit.
- Change in date of filling expressed as a number of days changed in the start of filling compared with the date in the natural condition.
- Change in date of emptying expressed as a number of days changed in start of emptying compared to the date in the natural condition
- Severity of winter drawdown calculated by the difference between water level during ice formation and lowest water level during the period of ice cover. This value can be compared with average depth or change in water covered area.
- Mismatch between the high and low water regimes of the lake and the hydrological regime of its tributaries. This metric is calculated by comparing, over the same year, the difference between the minimum and maximum monthly mean water level of the lake with the difference between the monthly mean water level of the lake when the inflow is minimum and the monthly mean water level of the lake when the inflow is maximum.

Data required for all these metrics: time series of water level and date of the beginning of water regulation. For the last two metrics, date of ice formation and time series of inflow are necessary, respectively.

- Qualitative assessment through expert knowledge on the presence, absence, intensity of changes in water balance.

Data required: knowledge of the use of the lake studied and of other lakes (to allow comparison).

2.3. Change in intensity of water level fluctuation

Water level fluctuation is another feature characterizing the dynamics of water flow. Its assessment is generally quantitative.

- Measurement of water level variations.

This can be measured at a daily, weekly, monthly or annual scale. The variations may refer to increases or decreases in the mean water level, maximum water level, or minimum water level. The metrics can be expressed as a mean for a given period, a maximum or a minimum. The reference condition can be historical or measured before regulation began. Lake level fluctuations can also be related to depth (%) e.g., calculated by subtracting the lowest regulated water level from the highest regulated water level, divided by mean depth of the lake when filled.

- Highest regulated water level - lowest regulated water level (according to operating concession) divided by mean depth.
- Mean annual number of level reversals (daily measurement).
- 90-percentile of the daily or weekly water level changes, i.e., the increase and decrease that is exceeded in 10 % of the identified events. It is assumed that the variation is negligible in the situation prior to regulation.
- Variance of current water level changes compared with historical data, or previous human impacts characterize the discontinuity in water level fluctuations.
- Difference in change in the water level between two adjacent days relative to the natural unregulated water level change.

Data required: time series of water level data (remote sensing or recorded in the field) and for some metrics, mean, maximum or minimum depth of the lake as well as historical information on filling and emptying dates.

2.4. Proxies for the alteration of water flow

Volume and area of the lakes are used to calculate proxies of the alteration of quantity and dynamics of water flow.

- Ratio between the volume of the lake before alteration and the volume after alteration.
- Lake volume fluctuation calculated as a ratio between "maximum water volume minus minimum water volume" and "volume of water at designed retention level" (for reservoirs).

Data required: bathymetric map and time series of water level measurements (in the field or assessed by remote sensing) before and after the alteration, or field work (followed by desk analysis) evaluating artificial inputs or extractions from the lake.

- Changes in the water surface area of the lake assessed by the difference between the present situation of water surface area and the reference surface area or between the area at the lowest level and at the highest filling.

Data required: remote sensing images.

2.5. Residence time

This feature corresponds to the time necessary for the complete renewal of the water in the lake basin. As for other metrics, change in residence time can be assessed qualitatively by expert knowledge of existing human influence on residence time.

However, some metrics can be used to quantify deviations in residence time following human alteration:

- Ratio between the lake volume and the mean rate of inflow of all tributaries, or the mean rate of outflow, or by using hydrological models, calculated before and after regulation.

Data required: bathymetric map and hydrometric monitoring station at the entrance or outlet of the lake with time series covering the period before and after alterations (at a minimum), or the implementation of models powered by different information (see next section) are necessary.

Another approach used in Spain, more suitable for temporary lakes, consists of measuring whether the period during which the lake contains water corresponds to its natural residence period.

- Difference between current residence time and reference residence time, assessed by comparing the present situation with the historical reference series. Analysing whether the lake has water during periods in which should be present, and whether it does not have water when water should be absent.

Data required: current and historical reference series of remote sensing information. In Spain, historical data are considered prior to 1984, i.e., prior to the remote sensing (RS) studies.

2.6. Modification of the connection to groundwater

This aspect is difficult to measure because information on the potential exchange zones between lakes and groundwater is often lacking; however, this process can be indirectly assessed:

- Where information on the status of the groundwater body is available, modification of the connection can be assessed as follows: if groundwater body is in good status i.e., unaffected by water extraction, it can be considered that the connection between the lake and the groundwater can be considered to be unaffected.

- The proportion of impermeable concrete banks can be considered in some cases as a proxy for an alteration of the lake and groundwater connection, as it affects near-surface groundwater inflows and outflows.

Data required: the use of these metrics requires at least a map of groundwater bodies and/or subsoil permeability characteristics and accurate information on groundwater withdrawals.

2.7. Lake depth variation

Lakes are of different origin (Hutchinson, 1957; Bengtsson et al., 2012). Whereas natural lakes were formed as a result of tectonic, glacial or fluvial processes, artificial lakes are created by human activities, i.e., the excavation of clay, sand, gravel and/or lignite, or by damming rivers and the creation of reservoirs. Consequently, there may be differences in lake bathymetry between the different lake types. These include variation in both maximum and mean depth as well as in the proportion of the different depth zones within the lake. The littoral zones are particularly important for the biological functioning of lake ecosystems, as they have high productivity by providing habitat for vascular plants, charophytes and other benthic algae. Consequently, lake littoral zones harbour much of the lake biodiversity.

In already existing lakes, modifications of lake depth can result from filling, artificial inputs of materials or active dredging. Another non-natural lake depth variation can result from installation or building of artificial structures to raise the water level of the lake.

- Change in mean depth – This is the preferred metric when the whole lake bed is modified by inputs or outputs of materials.
- Change in maximum depth.

Data required: knowledge of when, where and which anthropogenic activities leading to changes in bathymetry of the lake occurred. Bathymetry maps before and after alterations or in case of increase in water level, measurement of the water level before and after regulation.

2.8. Alteration of sedimentation processes

This type of alteration is related to the WFD parameter "Quantity, structure and substrate of the lake bed".

The measurement of the alteration of this process is difficult because it is often not easy to disentangle natural sedimentation rates and substrate quality from human influences. The following direct measurements are proposed:

- Presence or absence of non-natural sedimentation on the shoreline.
- Importance of non-natural sedimentation, siltation or both on the shoreline measured on the whole perimeter or at a number of stations and possibly weighted by lake depth.
- Importance of non-natural sedimentation in the deep lake zone expressed in cm of sludge weighted by lake depth.

Data required: cartography of the nature of the substrate gathered by observations or aerial photography in the shoreline, or acquired with an echo sounder in the deeper zone of the lake; maximum depth of the lake.

In reservoirs, an alteration of sedimentation processes can also be indirectly assessed by:

- Calculating the difference between the volume related to designed retention level and the volume corresponding to the year analysed.

Data required: this calculation requires precise information on the bathymetry of the reservoir.

2.9. Alteration of substrate

This type of alteration is related to the parameter "Quantity, structure and substrate of the lake bed". Measuring the substrate in the littoral zones of lakes involves various metrics that assess composition, grain size, naturalness, anthropogenic changes, spatial distribution, and material flux. These metrics are typically expressed in area (m^2), percentage (%), scores or index values.

- Comparison of substrate composition in the sublittoral zone factorized by grain size (reference unit: shore section).
- Comparison of substrate composition in the eulittoral zone factorized by grain size (reference unit: shore section.

- Presence of woody material and other organic material that forms part of the bottom substrate of the entire lake or at several stations, or in the inlet and outlet of the lake.
- Change in the spatial distribution of substrate of the entire lake area or at several stations.
- Importance of non-natural substrate in the littoral zone. This metric can be measured for the whole lake perimeter or at a number of littoral sampling stations. The type of non-natural substrate can be distinguished (e.g. introduced sand or gravel).
- Presence of various forms of erosion.
- Area of the lake or of the littoral zone with sediment extraction expressed in m^2 or as a percentage compared with the whole lake area or to the whole littoral zone.
- Area of the lake with dumping expressed in m^2 or as a percentage compared with the whole lake area.
- Area of lake with high density moorings within the whole lake or the littoral zone expressed in m^2 or as a percentage.

Data required: cartography (complete or partial) of the nature of the substrate and other artificial structures (moorings) in the area of interest. For the metrics that are weighted, the area of the lake or of the littoral zone has to be known. Historical records can be useful for measuring changes in substrate composition..

2.10. Alteration of the riparian zone

This alteration provides information about the “structure and condition of the lake shore zone” as defined in the WFD.

According to the scientific literature, the width of the riparian zone where impacts have been quantified range from 10 m to more than 100 m from the lake shoreline.

- Percentage of riparian vegetation loss along the shoreline.

This parameter can be calculated by observing the extent to which areas close to the lake lack riparian vegetation compared with the situation before human intervention. The alteration can be calculated as a distance (km) or a surface area (m^2) where the vegetation has been removed or altered divided by the total shore length of the lake or area of the riparian zone. Some lake basins may naturally be totally or partially free of woody riparian vegetation, such as lakes at high altitude and latitude or some Mediterranean lakes.

- Percentage of the riparian zone that consists of actively cultivated land or landscaped areas, or with non-natural land cover measured in km or in m^2 and weighted (or not) by the total length or area of the riparian zone as for the previous metric. In some methods, these observations are only made at sampling stations.
- Fragmentation of existing riparian cover.

Fragmentation is calculated using the landscape division index from Jaeger (2000), when riparian vegetation is surrounding the lake in reference condition. The reference condition generally corresponds to the presence of a continuous wooded fringe around the water bodies, although some lake basins may naturally be totally or partially free of woody riparian vegetation.

Data required: cartography (complete or partial if only sampling stations are monitored) of the nature of the vegetation (land cover) in the riparian zone. For the metrics that are weighted, the total length or area of the riparian zone has to be known. Historical maps can help to determine the level of alteration in land cover of the riparia zone.

2.11. Alteration of the bank

This alteration provides information about the “structure and condition of the lake shore zone” as defined in the WFD.

Most of the metrics include at least one of the following human factors. Different types of alterations can be grouped or kept as separate metrics in the assessment of bank alteration: artificial compaction, non-natural erosion, human changes of the profile, changes resulting from forest fires, human inputs of material, artificial structures on the banks including hard engineering and bank protection.

- Length of the perimeter (or number of monitoring stations) affected by at least one or more of the listed alterations, weighted by the overall perimeter of the lake or the total number of stations monitored.
- Fragmentation resulting from the presence of artificial structures. Its calculation is adapted from the degree of landscape division as defined by Jaeger (2000) which reflects the probability that two random locations in a landscape are located in two dissociated areas. This feature can be calculated for the area or the length of the banks.

Data required: cartography of the banks based on field survey or aerial images with information on the different alterations. Where weighting is used, it is necessary to measure the overall perimeter of the lake

2.12. Alteration of the shore zone

This alteration provides information about the parameter '“structure and condition of the lake shore zone”' as defined in the WFD.

Many alterations are linked to human activity: presence of embankments, reinforcement or other type of erosion protection (that in some cases may reflect a loss of lateral connectivity), presence of built structures along the shoreline (e.g. boathouses, piers, piling, outfalls and off-takes), presence of soft and hard engineering or the presence of erosion.

- Length of the perimeter (or number of stations) affected by one or several of the listed types of alteration, related to the overall perimeter of the lake (%) or to the total number of monitoring stations. Different types of alterations can be grouped or not in the calculation of the final value and therefore generate a single metric or several different metrics.

Data required: cartography of the shore based on field survey or aerial images with localization of the degraded shore. Where weighting is used, it is necessary to measure the overall perimeter of the lake.

- Changes in shore zone area caused by dewatering following lake level regulation. This metric can be expressed as a percentage of the littoral zone affected by the regulation (measured horizontally) or by the difference between dewatered areas at the lowest level compared with the total area at the highest level (measured horizontally).

Data required: digital maps and time series of water level.

2.13. Alteration of the aquatic vegetation

This alteration provides information about the parameter '"structure and condition of the lake shore zone" as defined in the WFD. It can be particularly important in the case of shallow lakes, where large areas of reeds can provide important habitats for wildlife and a means of protecting the lake from sedimentation and pollution.

- Zone covered by aquatic vegetation.

This feature can be characterized in area (m^2) or in length of the littoral zone. It can also be measured at sampling stations. These values can be weighted by the importance of the colonizable zone i.e., aquatic vegetation is expected to be present throughout the euphotic zone and in the littoral zone, provided the slopes are sufficiently gentle and the bottom substrate sufficiently fine to allow their colonization.

- Diversity of vegetation.

Measurements could be based on the calculation of diversity indices such as the Simpson's index. The number of aquatic macrophyte life forms can also provide valuable information on the human alterations if data are collected before and after the alterations or compared with appropriate reference sites.

- The presence of invasive species can be used to downgrade the quality rating of the littoral zone.

- Manipulation of aquatic macrophytes (by damage or removal) expressed in area of the lake weighted by the overall area of the lake.

Water level fluctuations exert impacts on the zone suitable for macrophyte development. Therefore, the area of this zone can allow, indirectly, the alteration of aquatic vegetation to be quantified.

- Increase in the shore zone area due to dewatered areas caused by water level lowering and regulation, i.e., the difference between dewatered areas at the lowest level compared with the total area at the highest level (measured horizontally).

Data required: cartography of the distribution of aquatic macrophytes based on field survey or aerial images before and after alterations (time series). Where weighting is used, information on the total colonizable area is necessary (nature of the substrate, slope, Secchi depth, water level, etc.).

2.14. General features

Several metrics can be considered as general parameters that may affect HYMO features, such as sedimentation, structure of the bank and the inflow.

- Human activities and naturalness of land use in their catchment areas have an impact on lakes. The intensity of this stress is generally evaluated by measuring the proportion of the area (%) of non-natural land cover. However, this parameter is also indicative of other stressors, eutrophication in particular, and should be assessed using more direct HYMO parameters.

- Lake use and the intensity of activities on the lake can provide information on the extent of HYMO degradation. This includes the main socio-economic uses (e.g. flow regulation) but also recreational uses (such as bathing or boating with an impact on erosion and the presence of artificial structures). Although qualitative information is generally easy to obtain (high or low visitor use for example), quantifying these uses (e.g., number of people using the lake shore, number of boats on the lake, importance of navigation) can be more complex, with a high degree of variability over time.

3. Tools and techniques for collecting HYMO information and examples of their application

This chapter reviews the tools and methods that can be used to collect the data for calculating the metrics described in the previous chapter. The descriptions of these methods are more or less complete, and readers may refer to the bibliographical references for further information.

3.1. Historical maps and images

If historical maps and images are available, they are of outstanding value in understanding the evolution of lakes and undertaking Integrated Water Resource Management (IWRM) of lakes and all other wetlands.

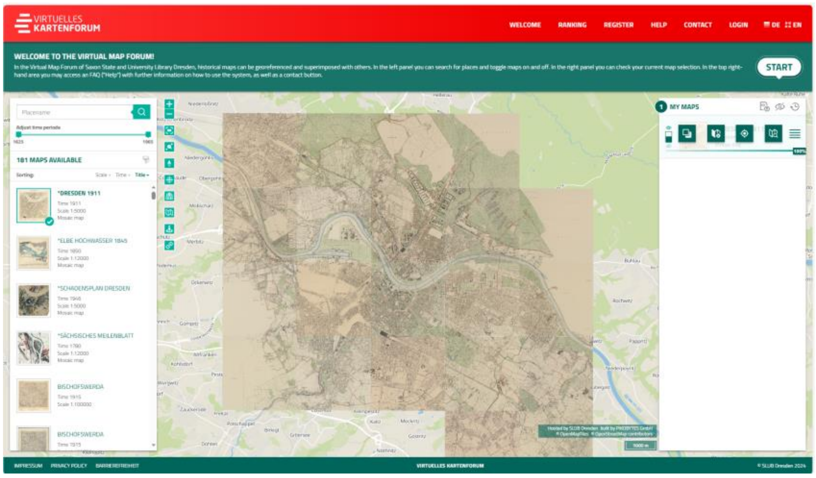
They can be used for understanding and defining reference conditions for lakes, their biocenoses and catchment area as well as their water regime. They also help to understand their primaeval hydrology and morphology, and from that base the natural evolution of the HYMO characteristics or the human alterations to the lake ecosystems.

Some examples of historical maps and images are given below.

In Germany,

— virtual Map Forum of Saxon State and University Library Dresden https://kartenforum.slub-dresden.de/en/. This contains 8973 historical maps with up-to-date maps in the background (Figure 1). These maps are available in English.

Figure 1. Example of Virtual Map Forum site view



Source: https://kartenforum.slub-dresden.de/en/

In Italy,

- the Military Geographic Institute preserves a historical cartographic heritage consisting of approximately 36,000 maps, of which 8,000 are pre-unification (before 1861) and 28,000 that were created after 1861 (IGM maps). Consultation can take place online, at https://www.igmi.org/carte-antiche.

In Czech Republic,

- maps from four military mapping exercises are currently available online (1st: Josephian land survey, 2nd: Francis land survey and 3rd: Francis-Josephian land survey). All together, they cover the period 1763-1880. http://oldmaps.geolab.cz/map_root.pl?z_height=70&z_width=0&z_newwin=0&map_root=1vm&lang=cs; http://oldmaps.geolab.cz/map_root.pl?lang=cs&map_root=2vm , http://oldmaps.geolab.cz/map_root.pl?lang=cs&map_root=3vm. The Military topographic maps of Czechoslovakia 1:25.000 / 1 : 10.000 (1953 – 1957, 1957 – 1972) used photogrammetry and are of great accuracy. https://ags.cuzk.cz/archiv

In Finland,

- digitized old maps, some dating back to the late 19th century, are available on the website vanhatkartat.fi. Users can explore maps from different periods and compare them simultaneously with the most recent terrain map of Finland. This platform provides a valuable tool for examining historical changes in the landscape, urban development, and geographical features over time.

In Sweden,

- the Swedish National Mapping Authority have one of the world's largest map treasures. In the digital archives there are more than one million historical maps, and they reach as far back in time as 1628. More information and links to the maps are available on the website https://www.lantmateriet.se/en/maps/our-map-services/historical-maps/.

In Hungary,

- a public map database "Arcanum Maps" is available. In this collection, historical city, country, and cadastral maps (i.e. containing details of ownership) can be found for several European cities and former empires. The website is available in six languages. https://maps.arcanum.com/en/.

In Norway,

- the national lake database includes information on ~ 243 000 lakes in Norway (https://kartkatalog.nve.no/#kart). Among this high number of lakes, bathymetric maps prepared for direct processing in a GIS are available for ~360 lakes or reservoirs. The majority of these lakes have contour lines with a height difference ranging from 2 m and 10 m. In addition to those maps already prepared for analysis in a GIS, the bathymetry of some lakes and reservoirs are available as scanned paper maps (as PDFs). Historical images are available from the publicly open Web-site www.norgeibilder.no. This service offers very new photos as well as a selection of historical orthophotos. In addition to those directly available, old images/photos can be ordered from the Norwegian Mapping Authority (Kartverket). The images offered have undergone a process of geo-referencing, so that the image matches the map.

3.2. Field survey

Many of the methods developed in Europe use field activities for data and information collection (Argillier et al., 2022). The field evaluation of HYMO characteristics can cover an important part of the physical characteristics of the lakes such as artificial and bank structures, shore zone features such as the presence or absence of vegetation, quality of the littoral habitat, typology of the substrate and variation in depth of the lake basin. In addition, field survey enables information to be gathered on human activities and to measure their consequences on lake habitats.

Water managers are closest to the field and in contact with many of the users (e.g., anglers, citizens) of aquatic environments. These users often have a good understanding of these aquatic environments and of their dynamics, so they are able to provide useful information for characterizing lake hydromorphology and to obtain data from monitoring locations such as water gauging stations, or from national or regional inventories.

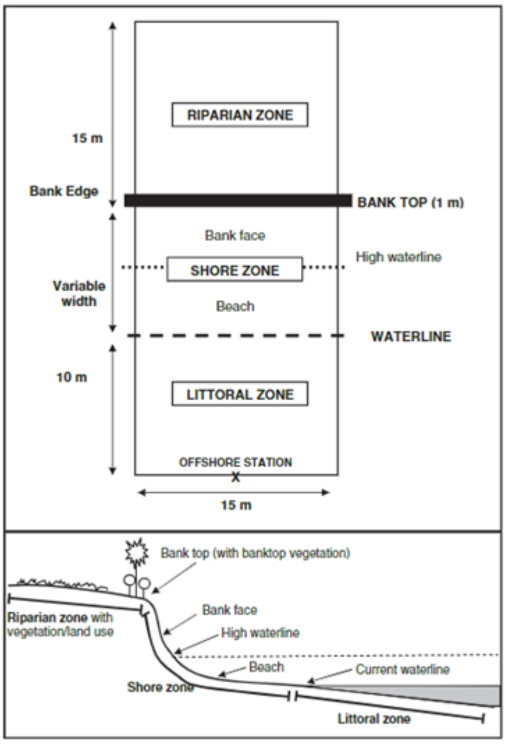
Field campaigns are also often necessary to implement special techniques when the HYMO parameters of interest cannot be directly observed, particularly in deep lake zones.

During field survey, in addition to data on hydromorphology, it is recommended that chemical-physical data are collected (transparency, dissolved oxygen, temperature, etc.), generally at the deepest point of the lake, to get a better understanding of ecosystem processes.

3.2.1. Direct observations

During field survey, by direct observations, it is feasible to evaluate the quality of four distinct parts of the lakes: Riparian, Bank face/Shore, Bank face/Beach and Littoral zone (**Figure 2**).

Figure 2. Parts of the lake easily observable by field survey.



Source: Rowan et al., 2006

In the riparian zone, the following parameters can be estimated: ground cover as trees, shrubs, grass, etc. but also land use, the presence of alien species, the features of the top of the lake shore and the presence of artificial features or infrastructure. These characteristics are easy to evaluate, especially if aerial photos are used as initial information when organizing the field trip. This part of the lake can be observed from the shore or from the water, on foot or by boat.

In the shore zone, descriptions should be made of the height and slope for the bank face, erosion or deposition, the prevalent material, the presence of vegetation, any artificial structures on the bank. Some of these features, such as the height and the slope of the lake shore, can be simply evaluated using an altimeter and slope can also be categorized easily as gentle to vertical.

The material of the bank face may or may not be visible. In the latter case, it is possible to determine whether the bank face is composed of natural or artificial material, as well as its type, based on a survey of neighbouring areas i.e., the presence of vegetation, the slope and the characteristics of the riparian zone. The presence of erosion or deposition is generally simple to identify but its natural or human origin is more difficult to determine. For the beach, the type of material, the slope, the presence of vegetation, the beach modifications, the presence of erosion and/or deposition are also easily observed by visual inspection. To recognize the presence of alien plant species a computerized key or a handbook where photos and names of possible alien species are shown can be helpful. Like the riparian zone, the shore zone of the lake can be observed from the shore but it is often easier from the water, on foot or by boat according to the depth.

In the littoral zone, observations of the prevalent substrate and presence/distribution of aquatic vegetation are strictly linked to the transparency of water. Inspection by boat is preferred.

Several field methods have been proposed to characterize many of these hydromorphological parameters like the Lake Habitat Survey (Rowan et al., 2006; Miler et al., 2015) or others (Reynaud et al., 2020).

The HYMO characteristics of the central part of the lakes are far less easily described by direct observations than in the littoral area except in very shallow lakes where the transparency is higher. The bedform patterns (topography) and landform features of the lake bottom (bathymetry) can be determined by measuring depth at various geo-referenced points in the lake. Lake bottom substrate and vegetation can then be described in the same way as in the littoral area or with specific equipment (see below and 3.2.2). The knowledge of transport and deposition of sediments is important for understanding the development of bedform, above all considering climate changes. In shallow lakes it may be important to evaluate the sediment transport and the natural erosion/deposition patterns that can greatly influence the quality and availability of habitats.

Therefore, with the exception of riparian characteristics, which can be observed from the lake shore, characterizing the HYMO features of lakes requires some basic equipment, in particular a boat. Scuba diving has been considered by some as the best way to make sub-aquatic direct observations in the littoral or central part of a lake (Melzer, 1999; Brosse et al., 2001; Jäger et al., 2004) but it is also very expensive. However, some specific equipment can be very useful for improving the descriptive quality of lake HYMO features. It is recommended thatan aquascope is used to observe the substrate or vegetation in shallow areas. The use of cameras can also be helpful for sub-aquatic observations of macrophytes for example, even if performance seems limited e.g., highly dependent on transparency, species determination difficult, etc. (Free et al., 2006). When the area or length of the shore has to be measured, a distance-measuring tool is required. **Table 2** shows the equipment useful for applying a field method.

Table 2. Equipment required for a field survey.

Equipment	**Details**
Field form, pencil, notebook, PC (tablet or laptop)	For new surveys or surveyors this manual may also be useful, in addition to a computerized application key or handbook or the help of an expert method
Topographic regional map or satellite images	Full large-scale map and smaller-scale maps for better identifying individual observation points
GPS	To position correctly at the observation points and record the location of particular features and/or impacts
Binoculars	To identify habitats and impacts if located at a distance
Camera	To photograph the different features of observation points (habitat, artificial infrastructure, human activities, riparian and littoral vegetation, invasive species)
Rangefinder	To measure the distance from the shore or the distance of infrastructure
Graduated rod	To verify the characteristics of the substrate and evaluate the water depth at the observation point
Aquascope	To examine the characteristics of the substrate under water and of vegetation; especially useful for evaluating the structure of vegetation and its percentage by volume
Rake and/or grappling hook	To take some samples of aquatic vegetation when turbidity and/or depth do not allow a view of the lake bed
Boat	To reach the deepest areas in particular

Source: Own production

Some lakes are not accessible or very difficult to reach, which generally increases the cost of field survey, for example by using a helicopter. Field monitoring of very large lakes in adverse weather conditions, or when regulations exclude the use of combustion engines, can also be difficult. In addition, field survey is generally considered to be time consuming, in particular in northern countries where many lakes are subject to mandatory regulatory monitoring.

The advantages and disadvantages of field methods can be summarized as follows:

PROS

- It is possible to clearly identify the physical habitats and some specific characteristics such as the material of the bank, the substrate in shallow areas, the type of vegetation and aquatic plants;
- It is possible to collect samples of water, sediment and vegetation (even aquatic plants);
- It is possible to obtain information on the different uses and their intensity that may have impacts on lake hydromorphology);

CONS

- It is necessary to have trained personnel;
- It can be time consuming and expensive for very large lakes or for a large number of lakes;
- Its application is not possible if the lake is not accessible or its accessibility is unsafe.

It may be necessary from time to time to evaluate the costs and benefits of field survey and to consider using hybrid methods: e.g. field and aerial photos and/or remote sensing.

3.2.2. Hydrometric monitoring stations

Evaluating water level fluctuation is usually done through sensors placed in the field and installed at hydrometric stations. These stations can be equipped with different types of sensors which can be grouped into "contact" sensors (piezometric probes, differential pressure transducers) and "non-contact" sensors (ultrasonic and radar sensors) (World Meteorological Organization, 1994). The hydrometric station can be powered by mains electricity or via a solar panel depending on its location. It can have a battery to support the solar panel for any real-time data transmission (Ciampittiello et al., 2021). The data can be acquired at different time scales depending on the precision of the information required and its use; this measurement precision varies between 1 cm to 3 mm for a continuous hydrometer (World Meteorological Organization, 2018). For example, during flood events or large withdrawals, a detailed evaluation may be necessary. The storage of all these data over time can allow the evaluation of statistical trends of level fluctuations and therefore of the change in the quantity of water available to the lake ecosystem. Continuous measurements of lake levels can improve the management of water resources, the safety of human populations, and the quality of lake ecosystems (Ciampittiello et al., 2021).

A real-time assessment of the fluctuation in levels can also be used to verify withdrawals and manage the water resource taking into account all needs. The hydrometric station can be quoted with respect to sea level so that the level data measured by the sensor can have an absolute reference (above sea level).

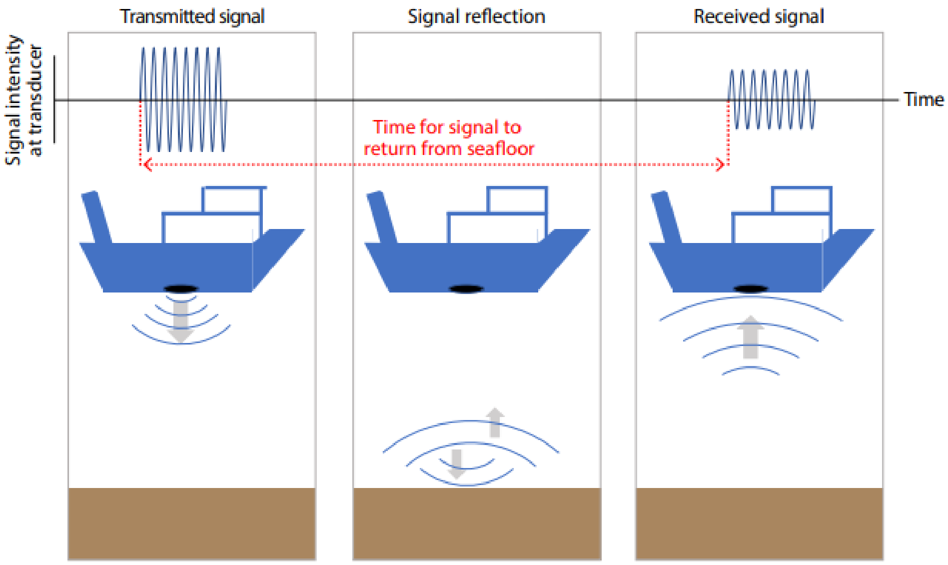
3.2.3. Acoustic data

Active acoustic methods are very convenient for surveying large parts of underwater areas **(Figure 3)**. They have been used for decades to map bed floor topography, as well as benthic habitats, of seas and lakes. The principle is based on the use of a piezoelectric transducer to deliver short acoustic pulses in the water column and record the echo that is reflected from the sea- or lake-floor. Characterization of the back-scattered signal, in particular its amplitude and return time to the transducer, can be used to derive information about the depth and the nature of the bottom. Coupled with the acquisition of GPS data, the information can be spatialized to quickly produce homogeneous maps of the sea- or lake-floor.

The most common echosounding system uses a single-beam transducer characterized by its acoustic signal frequency, the number and duration of acoustic pulses, and the focal angle of the beam. Frequencies associated with underwater acoustic surveys usually range between 30kHz and 200kHz (McCauley & Siwabessy, 2006) depending on the purpose of the survey and the distance to the sea or lake floor because of differences in energy loss and penetration (de Moustier, 1988). Basically, absorption of sound is weaker at low frequencies, allowing deeper areas to be explored, whereas shallow areas require the use of high-frequency sounds to limit the penetration of the signal into the sediment (Schlagintweit, 1993).

For bathymetric surveys that focus on mapping sea- or lake-floor topography, depth at each measurement point is calculated from the time delay between the transmission of the signal by the transducer and the recording of the first echo reflected from the bottom. This correlation is influenced by the speed of the acoustic wave that is a function of salinity, temperature, and pressure (Mackenzie, 1981; de Moustier, 1988) and thus may vary along the water column.

Figure 3. Principle of echosounding detection of underwater bottom

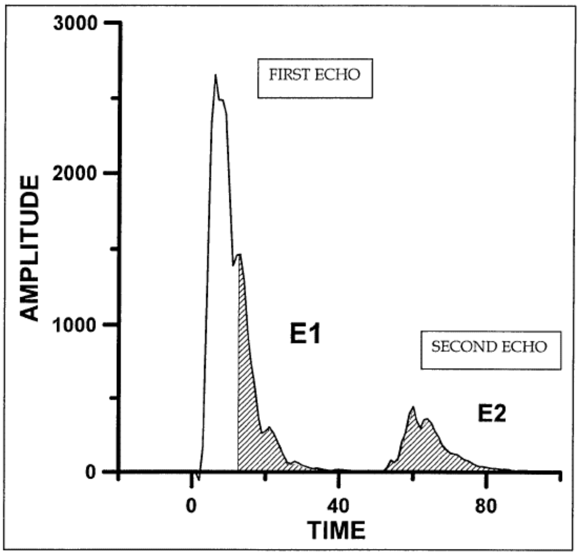


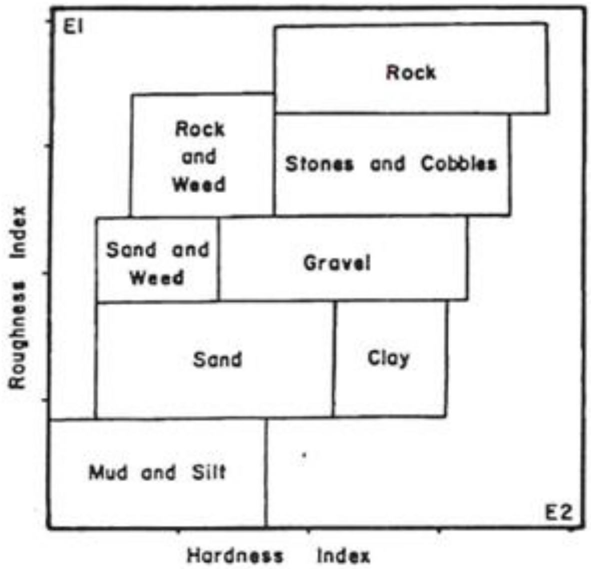
Source: Levine et al., 2020

Back-scattering, i.e., the energy returned by a target, depends on its mechanical characteristics. Acoustic methods used for classifying bottom substrates in the seas or lakes are based on this principle to assess both roughness (size) and hardness (density) of the substrate (Hamilton, 2001). According to the study of Orłowski (1984), correlation seems better when multiple echoes are measuremed but this requires the water surface to be smooth to reflect the first echo back to the bottom on the same vertical axis and thus get a second echo of the same bottom area. Although each contains components of both, energy of the first echo is mainly related to roughness whereas, owing to the double bottom interaction, energy of the second echo is mainly related to hardness

(Hamilton, 2001). Each pair of echoes can thus be assigned to a specific type of substrate. For example, using the RoxAnn system's multi-echo energy classification method that defines E1 as a bottom back-scatter index from the first echo tail and E2 as an acoustic reflectivity index from the second echo, low (E1; E2) indicates muddy sediments and high (E1; E2) indicates a rocky bottom (**Figure 4**). Assignment of substrate classes to acoustic measurements can be achieved using various supervised or unsupervised classification methods. However, it is obvious that in-situ calibration, performing ground-truth bottom samples in conjunction with acoustic measurements at several locations in the lake, greatly improves classification accuracy. In addition, Mouget et al. (2017) found that adding information on the depth of measurement also contributes to improved substrate discrimination.

Figure 4. Parts of the echoes used by the RoxAnn system (left) and an example of a classification diagram (right)





Source: Hamilton, 2001 on the left and Schlagintweit, 1993 on the right

Horizontal resolution of bottom mapping relies on the density and regularity of measurement points. For single-beam systems, increased resolution is achieved by reducing the distance between i) successive points on the same measurement track (lower ship speed and/or higher ping rate) and ii) successive parallel measurement tracks. Cross and Moore (2014) recommended transect spacing of 50 m to achieve 95% accuracy for estimates of lake volume, which is the sampling resolution also recommended by Levec and Skinner (2004) for bathymetric surveys on lakes <200 ha. The number of simultaneous acquisitions at different points may also be increased by the use of several single-beam echosounders facing different measurement points at the same time, or the use of multi-beam echosounder systems that sample a full swath of the sea- or lake-floor. Resolution (horizontal as well as vertical) is also dependent on depth, as the surface area reached by the acoustic wave increases with depth because of the cone shape of the beam, thus reducing the spatial accuracy of measurements.

It is worth noticing that sea- or lake-floor monitoring can be strongly affected by submerged aquatic vegetation, as the acoustic reflectivity of the gas-filled plant stems or leaves may generate an early echo leading to the underestimation of the true bottom depth (Sabol & Johnston, 2001). Thus, bottom tracking in areas colonized by dense submerged vegetation can be problematic. Although this phenomenon must be carefully considered to avoid erroneous measurement in bottom mapping, it can be used to conduct monitoring surveys of vegetation cover. Hydroacoustics is indeed proving to be one of the most objective, repeatable and scalable methods for monitoring and mapping aquatic macrophyte cover in lakes (Stocks et al., 2019).

Multibeam echosounders (MBESs) can also be used to determine the depth of water and characteristics of the sea or lake bottom. Unlike single beam echosounders that measure depth along a single track, MBESs emit a multidirectional radial beam that can generate detailed bathymetric maps over a broader area. MBES systems emit sound waves, called pings, in a fan shape that enable hydrographers to map the lake bed in 3D. The sound waves interrogate the lake bottom along a perpendicular line, called a swath, beneath the vessel carrying the instrument. The achievable swath width of a multibeam system is mostly determined by the depth of the lake. The shallower the water, the smaller the part covered. The maximum depth that can be mapped by MBESs depends on the acoustic frequency it uses. The lower the sound frequency, the further the sound can go. There are different types of MBES systems that use different frequencies to map shallower or deeper water.

Hydroacoustic measurements suffer limitations in very deep areas (signal dilution, more uncertainty owing to increased footprint) as well as in shallow waters (<1-2 m "blind" zone, depending on echosounder and vessel). The latter problems may be overcome with additional lateral or side-scanning systems emitting a fan-shaped beam parallel to the water surface (e.g., Sonnenburg & Boyce, 2008) or by combining with other methods such as digital photogrammetry (e.g., Alvarez et al., 2018; Lubczonek et al., 2022) or LiDAR (Light Detection and Ranging).

Apart from systematic biases - mainly due to instrumentation, speed and movement of the ship (pitch, roll, yaw) or spatial and temporal variation in sound speed - many other errors may affect acoustic data, as described in the IHO Manual on Hydrography (2005): bottom slope, bubble attenuation, false echoes (e.g., from fishes or aquatic vegetation), penetration in smooth sediments, ambient background noise, and weather (affecting water surface roughness and causing heave). Recent studies also focus on correcting sound speed errors caused by refraction of non-vertical acoustic waves for multibeam echosounders. Therefore, producing high-quality cartography requires control of all these errors, which can be achieved through careful calibration and compliance with precise measurement procedures.

3.3. Remote sensing tools

The science of remote sensing includes a range of techniques and methods to acquire information about spatial objects and phenomena without direct contact (Lillesand et al., 2015). In the field of ecology, remote sensing most often refers to the measurement of electromagnetic radiation emitted and/or reflected by an object of interest, in particular aerial and satellite imagery. Hydroacoustic measurement techniques are specifically described in section 3.2.2.

Remote sensing (RS) approaches to measure inland water quality date back to the early 1970s (beginning of the satellite era). They have been extensively applied for about 20 years to characterize inland water systems and analyse spatiotemporal trends (Topp et al., 2020) in spite of their somewhat limited capabilities for such applications, given that most sensors are primarily intended for land-based monitoring (Palmer et al., 2015). Therefore, RS is identified as a support to the implementation of the WFD and has already been successfully used in this context (Papathanasopoulou et al., 2019). RS has demonstrated its ability to quantify HYMO characteristics of surface aquatic environments at many functional spatial scales and thus constitutes an effective way to support HYMO assessment and monitoring (Bizzi et al., 2016). However, at present RS tools are still not commonly used for HYMO assessments of lakes in Europe, except for aerial photographs (Ciampittiello et al., 2017; Argillier et al., 2022; 2023).

3.3.1. Aerial imagery

Since they require high to very high spatial resolution data, HYMO characteristics of aquatic ecosystems have been mainly retrieved from aerial orthophotos and, more recently, from LiDAR (Piégay et al., 2020). At the European scale, such studies have been made possible and encouraged by the availability of aerial images with a national coverage in many countries (Bizzi et al., 2016). However, orthophoto databases are rather "static" and only give a very limited view over time, so resurvey campaigns should be planned regularly to keep them up to date. On the other hand, aerial images, the first of which date back to the 1930s, are also a very interesting way of going back in time and exploring past conditions. In some cases, these historical data could even be used as a source to help define reference conditions, required by the WFD, and used to assess changes over recent decades.

3.3.2. Remote sensing to monitor water level fluctuations

Water detection is a crucial preliminary step, but also a real challenge, when using remote sensing for studying surface aquatic ecosystems. Whereas delineation of water bodies may in some cases be provided through pre-existing geographical datasets, systematic detection becomes necessary when focusing on water surface changes. This issue has led to the development of several (semi) automatic methods and algorithms for surface water mapping. Although most of them are still based on Spectral Index (Bijeesh & Narasimhamurthy, 2020), like the NDWI (Normalized Difference Water Index) designed by McFeeters (1996) or its modified version (MNDWI) (Xu, 2006), those methods often fail to provide accurate and consistent results across a wide range of water bodies, whatever their type and characteristics. In addition, they suffer from many false positives, arising from confusion with snow, ice or shadows. However, thanks in part to the global improvement in computing capabilities, new processes have recently emerged to overcome these difficulties. Some examples of products resulting from these studies are given below.

1. The Global Surface Water database (Pekel et al., 2016) derived from the full archive of Landsat imagery – from Landsat 5 to Landsat 8 – provides long term (1984-2015) information on surface water dynamics at a worldwide scale, covering any water surface visible from space with areas larger than 30 m^2. This global dataset can be a useful source of information for lake HYMO monitoring to assess water seasonality and mid/long-term occurrence and changes in water level.

2. By using a combination of optical (Sentinel-2) and radar (Sentinel-1) imagery, SurfWater (Nguyen et al., 2024) offers an alternative/improvement with higher spatial resolution (10 m) and increased number of observations, providing high-confidence water masks over a 15-day period as well as yearly and monthly water occurrence maps. As for its application within the framework of the WFD, it should be noted that this method is particularly well-suited to Europe, where revisit times are more frequent.

3. Isikdogan et al. (2017; 2019) developed a data-driven model based on a fully convolutional neural network trained using characteristics of water bodies across the world. The DeepWaterMap model is publicly available (https://github.com/isikdogan/deepwatermap) allowing the processing and extraction of water surface data using Landsat 7-8 or Sentinel 2 imagery over any area in the world, even when partially obstructed by clouds. Although this method may not provide synthetic data over time periods unlike the previous ones, it does increase the quantity and frequency of data available for monitoring water level fluctuations in lakes.

4. The French–American Surface Water and Ocean Topography (SWOT) satellite launched in late 2022 will significantly improve global monitoring of continental water bodies, in particular through direct measurement of water levels using radar altimetry. The first estimations using the large-scale SWOT simulator show that SWOT will allow water levels to be measured with a precision of a few tens of centimetres even in lakes of only a few hectares (Ottlé et al., 2020). Although the SWOT 21-day standard cycle theoretically allows an approximately monthly monitoring, overlapping orbits mean that the same area may be observed several times during a single cycle depending on its location in the world.

3.3.3. Green LiDAR for the mapping of lake bathymetry

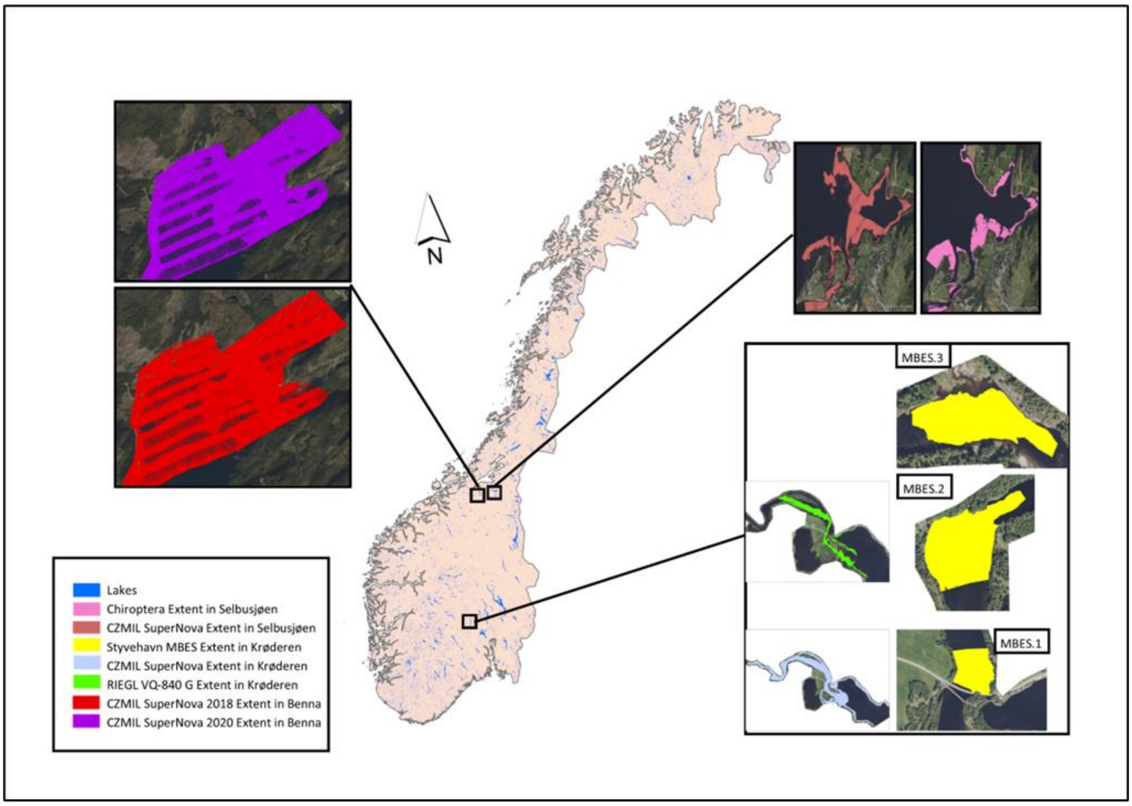
LiDAR is a technique for detailed mapping of the Earth's surface. LiDAR sensors can be divided into two main types: Red LiDAR uses electromagnetic waves in the infrared wavelength spectrum (1064 nm) and can be applied to map terrestrial areas (Harpold, 2015), i.e., areas not covered with water. The second type is Green LiDAR, which penetrates water surfaces by using the green spectrum of electromagnetic waves (532 nm). Green LiDAR is a more complex and costly technology and for this reason has so far been less applied. As Green LiDAR can be used to map the bottom of lakes, rivers and marine waters, it is sometimes described as bathymetric LiDAR, in contrast to topographic LiDAR (Red LiDAR).

Green LiDAR technology is typically applied by using an air-borne platform such as a drone for lightweight sensors, or more commonly from fixed wing airplanes or helicopters (Szafarczyk & Toś, 2023). Green LiDAR can potentially do bathymetric mapping with very high spatial resolution and good precision and can potentially cover large and complete/continuous areas in one flight campaign. The penetration depth is related to the clarity of the water, i.e., the sensor penetrates deeper into clear water than more turbid water.

In the freshwater environment Green LiDAR has mainly been applied internationally for mapping rivers (e.g., Kinzel et al., 2007; Mandlburger et al., 2015). Green LiDAR has been tested on rivers in Norway for estimating flood inundation (Awadallah et al., 2022) and for environmental assessments (Kastdalen & Heggenes, 2023; Kastdalen et al., 2024). Alfredsen et al. (2023) summarize various studies carried out at the Norwegian University of Science and Technology (NTNU) over the last few years with a focus on Green LiDAR application in river system engineering.

The use of Green LiDAR was recently evaluated based on three lakes in Norway (Ahmed et al., 2023), i.e., Selbusjøen and Benna in Trøndelag, and Krøderen in Viken, which was the first study of this kind in Norway (**Figure 5**; **Table 3**). The overall objective of the application was to investigate to what extent this technology could be applied for detailed mapping of the bottom topography (bathymetry). This was carried out by systematically comparing the performance of different LiDAR sensors, as well as MBESs, against each other in those lakes where two or more datasets were available and the datasets overlapped spatially.

Figure 5. The three lakes in Norway where Green LiDAR measurements were performed and analysed. Benna is the one in the upper left, Selbusjøen in the upper right, and Krøderen in the lower right. CZMIL: Coastal Zone Mapping and Imaging LiDAR; MBES: multibeam echosounder; RIEGL VQ-840 G: integrated airborne laser scanning system for combined hydrographic and topographic surveying



Source: Ahmed et al., 2023

Table 3. Overview of the sensors applied in each lake. CZMIL: Coastal Zone Mapping and Imaging LiDAR; MBES: multibeam echosounder; RIEGL VQ-840 G: integrated airborne laser scanning system for combined hydrographic and topographic surveying

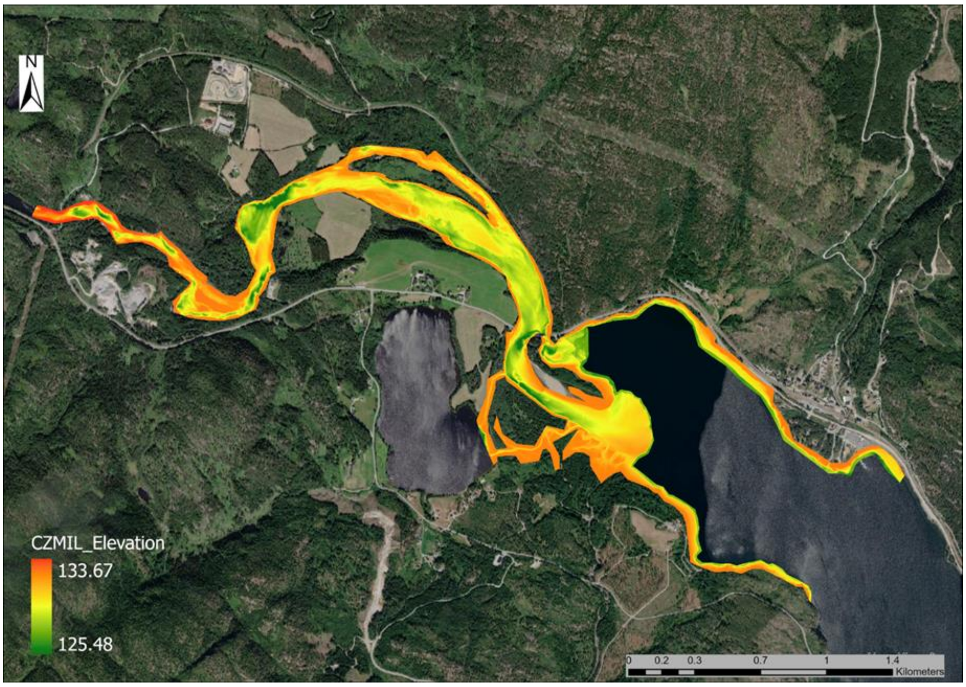
Lake	Sensor
Selbusjøen	CZMIL SuperNova
	Chiroptera
	Norbit MBES
Krøderen	CZMIL SuperNova
	RIEGL VQ-840 G
	RIEGL VQ-880 G
	Norbit MBES
Benna	CZMIL SuperNova (overflying in 2018 and 2020)

Source: Ahmed et al., 2023

The general conclusion from the study was that when comparing the residuals between MBESs with Green LiDAR measurements, as well as Green LiDAR measurements against each other, the residuals are generally very small (Ahmed et al., 2023). The residuals are in most cases much less than 10 cm, based on the mean and median residuals. When the Green LiDAR sensors are compared with each other, the residuals are close and normally distributed around 0 cm, indicating no systematic error. The outliers that were found in the datasets are typically in very shallow water or at the maximum penetration depth where one sensor might have a deeper penetration than others. In the analysis reported by Ahmed et al. (2023), the point densities of the Green LiDAR range from 80 points/m^2 (in some smaller parts of one of the lakes) to 30-50 points/m^2 as the depth increases, and decrease to fewer than 2 points/m^2 in the final metres before the maximum penetration depth is reached.

Under perfect flying conditions and clear water, Green LiDAR can be capable of measuring down to around 20 m below the lake surface (**Figure 6**). This was the case for the very clear Lake Benna, whereas in most lakes penetration depth much less than 20 m depths is to be expected. Furthermore, the study concluded that Green LiDAR is suitable for mapping shallow parts of lakes, in contrast to MBESs that are less feasible to operate in shallow waters. MBES can cover deeper parts of the lakes, while at moderately deep parts of the lake, both technologies seem suitable and overlap. Note that the Green LiDAR technology was tested to measure the bathymetry of Hungarian lakes. However, owing to their shallow depth and richness in suspended solids (biomass, humic substances, and minerals), it was shown to be of limited value.

Figure 6. Bathymetric map generated from Green LiDAR (Coastal Zone Mapping and Imaging LiDAR sensor) from the upper end of Krøderen, where the River Krøderen flows from left into Lake Krøderen to the right. The darker (grey/black) areas of the lake cannot be measured as they are too deep for the sensor to penetrate.



Source: Ahmed et al., 2023

3.4. Hydrological models

The key output of most of hydrological models is discharge (runoff, water flow). Nevertheless, discharge itself is not always relevant for assessing the HYMO or ecological status. Richter et al. (1996) published a set of hydrological indices as a basis for assessing the ecological impacts resulting from river regulation, based on two time series of discharge. The indices are organized in five groups, describing the changes in the regulated situation, compared with the unregulated basin. These five groups of indices cover changes in; i) the magnitude of monthly water conditions, ii) the magnitude and duration of annual extreme water conditions, iii) the timing of annual extreme water conditions, iv) the frequency and duration of high and low pulses, and v) the rate of frequency of changes in water conditions. Within each of these groups, a set of more detailed indices are described, based on time series of discharge from before and after river regulation. This concept has many similarities with the EU WFD where the changes from pristine (unregulated or reference conditions) to the present situation are analysed. The basis for such an assessment is that the hydrological conditions before and after human intervention to the hydrological cycle can be found, and hydrological models are key instruments to analyse this, either by simulating the unregulated situation, or by simulating the effect of different sets of human interventions.

It should be emphasized, however, that hydrological models are not perfect representations of the real world situation, neither before nor after human interventions. As such, use of the results generated from a hydrological model should be treated with care.

The following

Table 4 provides some key references to hydrological models commonly used. More information is available in the review by Singh (2018). In addition, application of hydrological modelling in Southern and Central Europe and in Norway are given in Annex 1 and Annex 2.

Table 4. Commonly applied models and references.

Model name	Key reference
Water Evaluation and Planning model - WEAP	Yates et al., 2005
Soil and Water Integrated Model – SWIM	Krysanova et al., 2005
HBV-model	Bergström, 1992
MIKE SHE	Abbott et al., 1986
Soil and Water Assessment Tool (SWAT) model	Arnold et al., 2012
HYPE	Lindström et al., 1997
PHYSITEL, a specialized GIS for supporting the implementation of distributed hydrological models	Rousseau et al. 2011
Terrestrial Hydrology Model with Biogeochemistry (THMB)	Le Coz et al. (2009)
Distributed Hydrology Soil Vegetation Model (DHSVM)	Zhao et al. 2016
HEC-HMS models	Gyawali & Watkins, 2013
Gravity Recovery and Climate Experiment (GRACE)	Seka et al. 2022
Total water storage (TWS) from the Gravity Recovery and Climate Experiment (GRACE)	Buma et al. 2016
Raster-based distributed hydrologic model, Coupled Routing and Excess STorage (CREST)	Khan et al. 2011

Source: Own list

3.4.1. The basic concepts of hydrological modelling

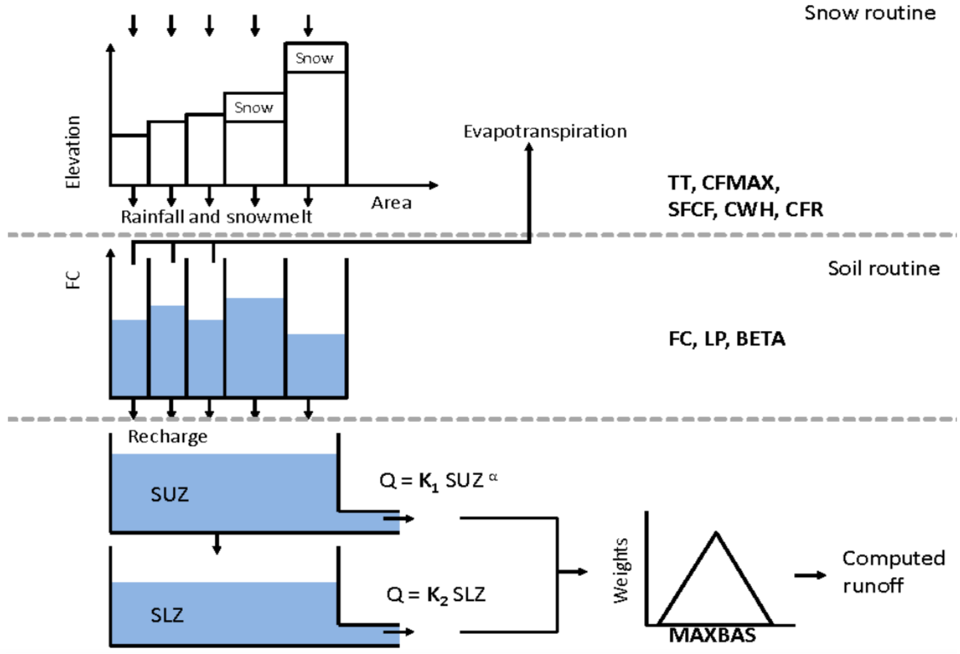
Hydrology can be defined as the study of the distribution and movement of water both on and below the Earth's surface, as well as the impact of human activity on water availability. Hydrological modelling is the science of representing hydrological processes using computational algorithms. Hydrological models include purely data-driven models where relationships between input and output are described by statistical relationships. These are 'black-box models', meaning that the hydrological processes are not, or to a very limited extent, described by physical relationships. By contrast, physical-based models describe hydrological processes and relationships by physical laws. In between these two approaches are conceptual models, which have some level of physical description of the system but do not apply physical equations. Conceptual models are the most commonly applied hydrological models and are the ones described here. Such hydrological models are thus a conceptualized description of the hydrological processes occurring in the real world. They describe the hydrological cycle, or parts of the hydrological cycle, by a simplified set of relationships for the purpose of understanding, predicting and managing water resources. Except in extremely data rich environments, these modelling approaches are usually considered superior to more complex approaches. The practice of hydrological modelling is therefore generally hampered by uncertainties in process and the overwhelming influence of heterogeneities.

Hydrological models are usually driven by meteorological input, where the most important are normally precipitation and temperature. By means of their conceptual description of catchment processes, these models transform this input to hydrological output, such as river flow, groundwater recharge and lake water levels. Even though many hydrological models are conceptual, they require a set of data describing the characteristics of the catchments to which they are applied, such as catchment size, altitudinal distribution (hypsographic curve), lake/reservoir percentage, land use and soil properties.

The fact that hydrological models introduce a simplification of the real world also means that the hydrological data and the description of the catchment properties might have a looser connection to its more fundamental physical meaning. The data in the conceptual model are then turned into model parameters, which sets up the model treated as 'free', in the meaning that they will be tuned and reset during the process of model fitting. This procedure is termed 'calibration'.

A common way of making a conceptualized version of catchment processes is the so-called linear-bucket approach (see example in **Figure 7**). A linear-bucket based model mimics the natural processes in a catchment by the analogy of the water entering the bucket, or a set of linked buckets, and the transformation of this into hydrological output, e.g., to runoff. There are many variants of linear-bucket hydrological models, adapted to the behaviour of the dominating hydrological processes in the geographical region of its application, as well as to the core purpose of the model.

Figure 7. The HBV-model as implemented by Staudinger et al. (2015). The catchment processes are conceptualized as linear buckets. The codes given are model parameters (TT, CFMAX, SFCF, CWWH, CFR and FC, LP, BETA, and K1, K2, Alpha, MAXBAS) that are adjusted and fixed during the calibration process, and state variables (SUZ, SLZ) that are calculated during simulations

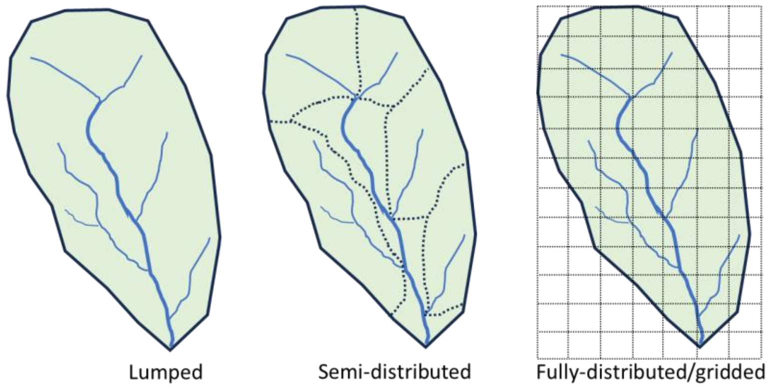


Source: Bergström, 1992; Staudinger et al., 2015

3.4.2. Spatial and temporal resolution and extent

For the combined reasons of access to detailed data as well as computational constraints, hydrological models represent the real world in a spatially aggregated manner. This means that catchments, or parts of catchments, are modelled as uniform units. The simpler modelling approach is to handle the entire basin as one homogeneous unit (lumped), whereas more advanced approaches divide the basin into smaller units, typically sub-basins. The most advanced models are gridded/fully distributed models that describe the basin in uniform cells (**Figure 8**). In a gridded model, each cell is handled as a uniform unit (**Figure 9**), whereas for a semi-distributed models each sub-basin is handled uniformly, and takes uniform and unique input at the sub-basin level. This means that each spatial unit can take unique meteorological input and unique parameter sets, and will generate unique results for each unit.

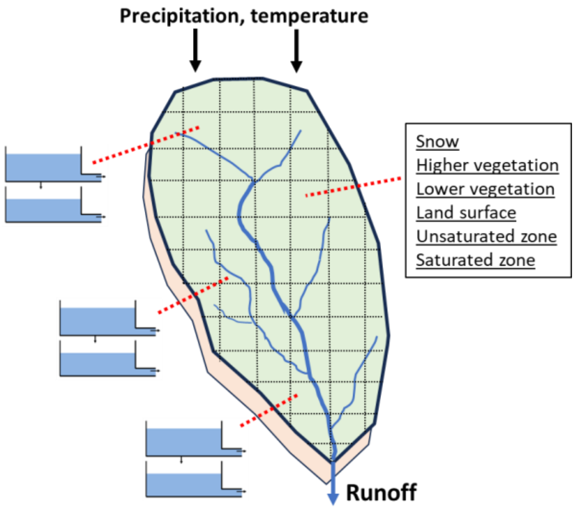
Figure 8. The principal differences between spatially lumped (left), semi-distributed (centre) and gridded (fully distributed) hydrological models.



Source: Own production

There are pros and cons of selecting the different approaches/models that depend on factors such as the availability of data, the purpose of the modelling study and the available resources for setting up the model. If meteorological input varies extensively across the modelled catchment and these data are available, a semi-distributed or gridded model would be preferred. This is typically in larger basins with varied topography, and where spatially distributed meteorological data are available. Large internal differences in land use are also a valid reason for selecting spatially distributed model. The model output is normally also limited to the outlet of the catchment for a lumped model, while spatially distributed models can generate output at several locations in the basins, according to the way the model is designed. A spatially distributed model requires that each of the sub-basins (or grid cells) are connected, as they are in the real-world catchment, to ensure that the water is properly transported to the downstream areas of the basin.

Figure 9. The principles of a gridded hydrological model, where the hydrological equations are solved in each of the cells.



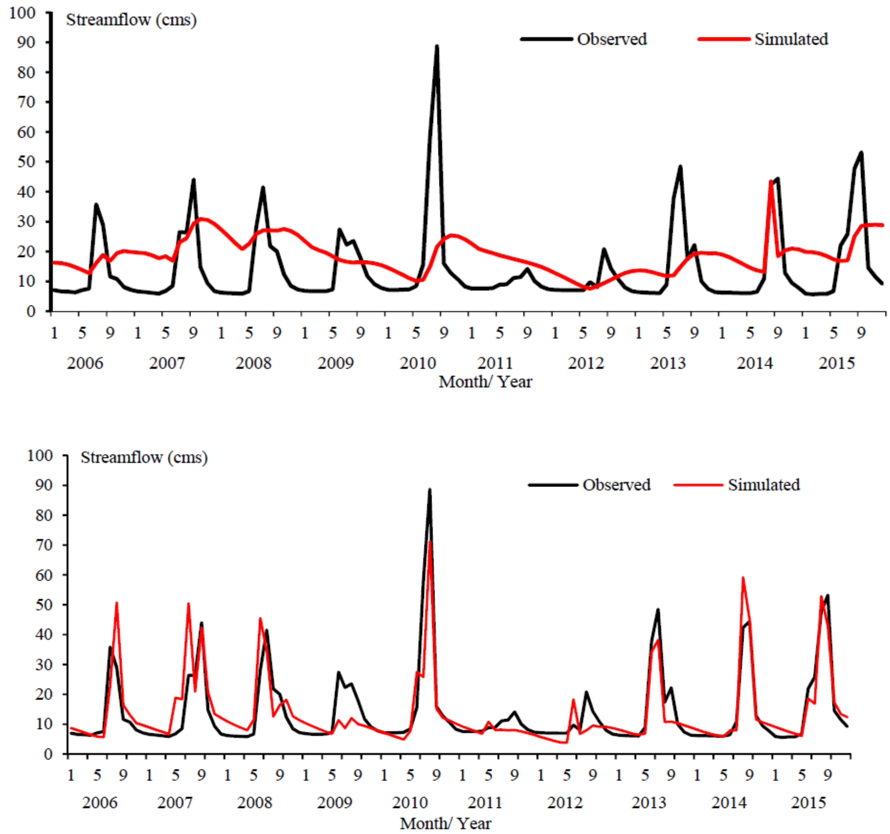
Source: Own production

Hydrological models can often be operated with variable time resolution, defined by the problem at hand and the capability of the model. Long-term planning, such as assessing the long-term effect of climate change on future water availability, can be analysed with a monthly time step, whereas short-term forecasting, such as predicting floods in the near future may require simulations with a one-hour time step. Processes that vary extensively throughout a day will also benefit from the simulation with a one-hour time step. The resolution of the time step can be limited however, by the availability of data (especially meteorological data) with a similarly short time step. If data with a one-hour time step are not available, the model will not produce informative output with detailed temporal discrimination. Note that a finer time resolution, combined with a long simulation period (and possibly fine spatial resolution) will increase the computational time of the simulation.

3.4.3. Data acquisition, model calibration and validation

Hydrological models must be adjusted (calibrated) before reliable results can be generated, owing to the enormous heterogeneity of catchments and the need for simplifications made in hydrological models. This process involves systematically comparing and evaluating model output against observations, for overlapping periods (**Figure 10**). The model parameters are adjusted until a satisfactory match between model output and the observations is reached. The most common way to calibrate a hydrological model is to compare simulated discharge against observed discharge, although observations of water level in lakes or reservoirs, snow storage and other hydrological data can also be used for this purpose.

Figure 10. The figure shows a non-calibrated Soil and Water Assessment Tool (SWAT) model (upper part) and a calibrated SWAT model (lower part) where simulated (red) and observed (black) monthly streamflow are compared. Streamflow measured in cubic metres per second (cms).



Source: Jin & Jin, 2020

The calibration process normally starts with a visual comparison of simulated results and observations, but to fine tune the models, statistical criteria for finding the “best” parameter set are useful. These criteria are often termed “objective functions”. The output from the statistical assessment determines the “goodness of fit” of the model. Common statistical criteria are the Percent Bias (PBIAS) (Gupta et al., 1999) and Nash–Sutcliff Efficiency (NSE) (Nash & Sutcliffe, 1970), and several studies have published recommended values for classifying the goodness of fit.

Equations commonly used in calibration are given below.

Equation 1. Calculation of the probability bias

$$PBIAS = \frac{\sum_{i=1}^{n}(Qmod - \ Qsim)}{\sum_{i=1}^{n}(Qobs)}$$

Equation 2. Calculation of the Nash–Sutcliffe Efficiency

$$Nash\text{–}\ Sutcliffe\ Efficiency\ (NSE) = 1 - \frac{\sum_{i=1}^{n}(Qmod - \ Qobs)\text{^}2}{\sum_{i=1}^{n}(Qobs - Qmean)\ \text{^}2}$$

In equations 1 and 2 *Qmod* is modelled/simulated runoff, *Qobs* is observed runoff, and *Qmean* is the mean/average of the observed runoff values.

PBIAS is useful for evaluating the simulation of the water balance for the entire simulation period, year by year, or on a seasonal or monthly basis. The precision is given as the percentage bias from the observations, where 0 % bias is a perfect fit (perfect water balance).

The Nash-Sutcliffe Efficiency (NSE) is suited for describing the model’s performance in capturing variations in time, e.g., the variation in runoff. NSE is particularly useful for assessing the magnitude and timing of the simulated values against the observations. The NSE criterion ranges from negative infinity (- ∞) to 1, where NSE = 1 describes a perfect model. With a NSE = 0, the model has the same predictive power as the mean of the time series in terms of the sum of the squared error.

Hydrological models can have built-in tools for automatic calibration, which can reduce the time spent on calibration. If an automatic tool is applied, the model must support simulation of a large number of model runs, coupled with an algorithm that automatically tests different sets of parameters and compares the model output against observations. The parameter estimation tool (calibration tool) will run until a specified goodness of fit is reached, or a maximum number of simulations has been reached.

Some hydrologists argue that the best parameter sets found during calibration should be tested on an independent set of observations. This second step is called validation, typically executed on a different time period of data from the one used for calibration. A validation procedure is usually applied when long time series of observations are available, as the validation dataset will not be used to improve the model performance, but rather to confirm the performance of the model achieved in the calibration.

A well calibrated model will give confidence that the model is a reasonable representation of the real world, and the better the performance the more likelihood there is that the model will be able to simulate future situations (where observations are not available). Conversely, a hydrological model with poor performance will give little confidence of its ability to simulate other situations with acceptable precision. A model with satisfactory performance is, therefore, essential to be able to apply to future situations, such as climate changes, land use changes, or introduction of water-related infrastructure, and give results with limited uncertainty.

3.4.4. Hydrological models and uncertainties

The results of hydrological models depend, to a large extent, on the quality of the data used as model inputs. In addition, model outputs are to some extent determined by the model chosen and the way it is applied. As discussed above, the number of locations that can be generated in the output is given by the spatial configuration of the model. It is also possible, however, to generate output from other locations by further processing of the results; for instance, by the use of areal-scaling or by different methods of transferring the output to new locations. The time resolution of the output cannot be finer than the time step of the simulation, but results with a fine time resolution (e.g., sub-daily) can be aggregated into daily, weekly, monthly or annual values.

A well calibrated model will normally generate output with higher precision than a model with a lower calibration performance. One way to handle the uncertainty embedded in the model results is to secure proper uncertainty assessments. This can be made by systematically analysing the uncertainty in the input data (meteorological forcing) and the model parameters. Such an uncertainty assessment will generate results indicating the range of expected output, possibly presented by their statistical distribution, e.g., by 25- & 75-percentiles.

3.5. Overview of commonly used sedimentary models and key references

Lake hydromorphology describes the interactions between water and sediments in lake basin and fluvial systems feeding lake basins throughout the catchment and the corresponding processes across all spatial and temporal scales.

Interactions between the hydraulic characteristics of the water body and the available sediment across all spatial and temporal scales forms morphological features of the lake basin (Haun & Dietrich, 2021). Sedimentology can be defined as the study of sediment formation and evolution. More specifically, the study of sediment evolution can be explained both through net erosion and net deposition at specific sites and through change in transport mechanisms (Bragg et al., 2003).

One of the WFD requirements for assessing the HYMO quality of lakes is to assess changes in lake depth variations. For assessing lake depth variations, the Directive seems to allow a certain amount of flexibility as these lake depth variations could be a function both of basin morphology and water level movements (Bragg et al., 2003). For assessing water level movements, hydrological models are particularly useful as described in the previous sections. Regarding the assessment of the morphological evolution of the lake basin, the focus should be on the sedimentary evolution of the lake basin and changes in sedimentation rates. However, impoundment or lowering the water level can passively modify the shape of the lake basin as well as changes in rainfall patterns at the catchment scale (input data for hydrological models), as they change the rate of erosion and thus the quantity of sediment reaching the lake basin (Bragg et al., 2003; Gonzalez Rodriguez et al., 2023).

Several methods can be used to determine the sedimentation rate of a lake:

- The usual methods for determining sedimentation rate of a lake are direct in-situ measurements such as bathymetry and core chronology, or more recently remote measurements using satellite and remote sensing imagery with a continuous monitoring effort (Gonzalez Rodriguez et al., 2023);
- When data from in-situ or remote measurements are unable to determine with sufficient precision or even predict the sedimentation rate of a lake, physics-based semi-empirical sedimentary models can be used.

Physics-based semi-empirical sedimentary models are numerical models that can be fed by either in-situ measurement data or remote measurement data as described above.

In addition, physics-based semi-empirical sedimentary models can be fed by two types of data with regard to spatial scale: either data collected directly from lake sediment layers, or data aggregated at the scale of the catchment feeding the lake basin.

Most models that use data collected directly from lake sediment layers are geo-chronological models of radioactive decay. Determining sediment evolution and sediment rate in the lake basin is achieved by studying various sites of net erosion and net deposition on the lake bed. This is a points-based method (Gonzalez Rodriguez et al., 2023). The input to the model is provided by time series data from cores at different sites of the lake and fed into a mathematical equation system to provide a spatial distribution of lake sedimentation at model output. In this type of model, the decay of radioactive isotopes (such as ^{210}Pb, ^{226}Ra and ^{137}C) is used for dating lake sediments and thus calculating the rate of sedimentation of the lake. For example, in Chen et al. (2019) the chronologies and sedimentation rates of sediment cores were calculated using various mathematical models based on ^{210}Pb. The most encouraging technique for determining the sedimentation rate is to use the ^{210}Pb method as this is a naturally occurring radionuclide with a half-life of 22.3 years, derived from the decay chain of ^{238}U (Singh & Vasudevan, 2001). In Chen et al. (2019) the final model was validated by comparing it with historical data on the lake basin studied.

This method seems to be suitable for calculating chronologies and sedimentation rates in shallow lakes and rapidly flushed floodplain lakes and other similar lacustrine systems within floodplain landscapes (McCall et al., 1984; Chen et al., 2019). Another advantage of this type of method for estimating the sedimentation rate of a lake is the spatial information that can be extracted about the lake basin thus avoiding data interpolation.

Despite the advantages of these methods, the assumptions of mathematical models are not always fulfilled in nature (e.g., constant sedimentation rate and constant lead (Pb) concentration) (Gonzalez Rodriguez et al., 2023). The chronology of the sediments may thus prove to be very difficult to disentangle with the radiometric dating method, as several factors affect sedimentary conditions and sediment accumulation rates (Singh & Vasudevan, 2001). This is why radiometrically corrected dating models are generally used, which means additional processing time.

With regard to models using data aggregated to the scale of the catchment area feeding the lake basin, the most widely used physics-based semi-empirical model is the RUSLE erosion model (Revised Universal Soil Loss Equation; Oguz et al., 2019). Determination of sediment evolution and sediment rate of the lake basin is achieved by studying mechanisms of supply and transport of sediment at the scale of the lake's catchment area. The input data for the model are provided by hydrological, morphological and land use characteristics of the catchment. These are fed into a mathematical equation to produce an erosion height at the scale of the catchment area feeding the lake and thus a sediment flow arriving in the lake as model output. These are semi-empirical physical models based on the universal law of conservation of mass, which links the temporal evolution of the topographic surface to the divergence of sediment flow. Here, in the RUSLE equation (Eq. 3), the divergence of sediment flow is represented by the multiplication of erosion factors influencing the mechanics of sediment supply and transport at the catchment scale. Soil erosion is thus directly affected by a region's rainfall erosivity, soil erodibility, slope length, slope steepness, land use type and supporting practices (Oguz et al., 2019). The model produces an output in the form of a spatial distribution of annual soil loss per unit area (Renard et al., 1996).

Equation 3. Calculation of soil erosion

$$A = RKLSCP$$

In equation 3, A is the soil loss expressed in the units selected for K and for the period selected for R, usually expressed in ton*acre^{-1}*yr^{-1}, but other units can be selected (i.e., ton*ha^{-1}*yr^{-1}); R, the rainfall-runoff erosivity factor – the rainfall erosion index plus a factor for any significant runoff from snowmelt; K, the soil erodibility factor; L, the slope length factor; S, the slope steepness factor; C**,** the cover management factor; P, supporting practice information.

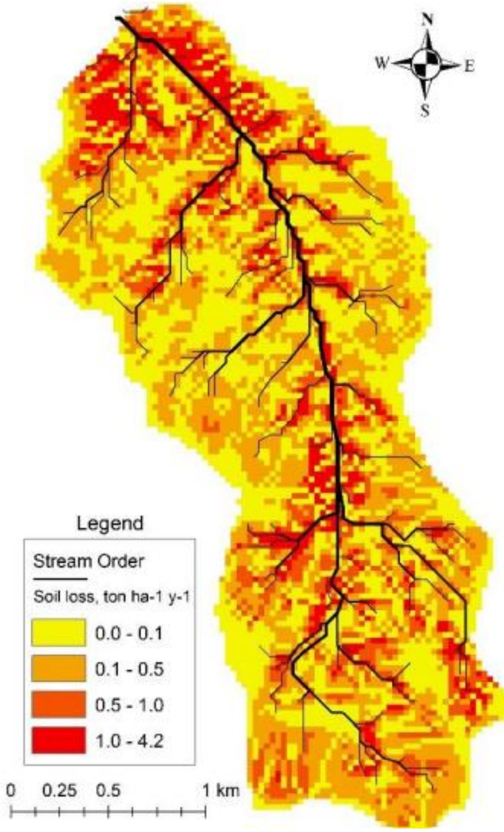
The RUSLE erosion model has been used and accepted by many researchers for erosion prediction owing to its high reliability in erosion prediction studies (Ullah et al., 2018), and is particularly suitable for low sedimentation regions. For example in Oguz et al. (2019) the final model was successfully validated by comparing it with bathymetric data of the lake basin studied.

On the other hand, because of the phenomenological nature of the input factors, the RUSLE approach relies on calibration and expert knowledge making the approach less robust and less objective. Furthermore, the approach is an empirically based equation for sheet and rill erosion but it does not represent gully or stream erosion (Oguz et al., 2019) which may account for a significant proportion of the sediment transport feeding the lake basin.

For example, in the study of Oguz et al. (2019), the sediment accumulation of a reservoir was calculated as 21,600.75 m^3 over the 42-year study period. The minimum (370.03 m^3) and the maximum (775.69 m^3) annual sediment amounts were calculated in 2008 and 1984 respectively. The simulated data fit in well with the bathymetry measurement data (**Figure 11**).

Quantifying sedimentation rates remains somewhat challenging (Palinkas & Russ, 2019) and the various methods commonly used for this purpose each have their own advantages and limitations, as described in Gonzalez Rodriguez et al. (2023). In addition, many of these HYMO processes cannot be described analytically yet, owing to their stochastic behaviour and the multitude of processes involved across spatial and temporal scales even though the assessment of these HYMO processes is crucial (Haun & Dietrich, 2021).

Figure 11. Potential soil loss map of a basin according to the RUSLE method.



Source: Oguz et al., 2019

4. How to set reference conditions

4.1. Reference texts

Defining reference conditions is an important step, as it determines the final score that will be given to the quality of the ecosystem being assessed. According to Annex II. 1.3 of the WFD, type specific HYMO reference conditions, like biological and physico-chemical references, have to be defined. Annex V states that reference conditions correspond to "*no, or only very minor, anthropogenic alterations to the values of the hydromorphological quality elements for the surface water body type from those normally associated with that type under undisturbed conditions.*"

In Annex V.1.2.2 there is a specific description of the reference conditions for morphology and hydrology in lakes (**Table 5**).

Table 5. High status (reference condition) for hydromorphological quality elements in lakes

Element	High status
Hydrological regime	The quantity and dynamics of flow, level, residence time, and the resultant connection to groundwaters, reflect totally or nearly totally undisturbed conditions.
Morphological conditions	Lake depth variation, quantity and structure of the substrate, and both the structure and condition of the lake shore zone correspond totally or nearly totally to undisturbed conditions.

Source: Annex V.1.2.2 in the WFD.

Guidance n°10 (European Communities, 2003b) provides "*principles, ways of reasoning and suggestions on alternative pathways of action*" when reference conditions have to be defined. In addition to a reminder of the normative conditions, various methods for establishing reference conditions are presented in **Table 6**. This suggests, for example, that reference conditions can be site-specific (rather than type-specific as recommended in the original text) or based on expert judgement.

Table 6. Different approaches that can be used to define reference conditions.

Approach	Strengths	Weaknesses
Spatially based using survey data	Region specific	Expensive to initiate
Predictive modelling	Site specific	Requires data, calibration and validation
Historical data	Often inexpensive to obtain	Variable data, few parameters and data quality may be poor or unknown, static measure
Paleo-reconstruction - **Direct** - **Indirect**	Incorporates both physico-chemical and biological data Site-specific Calibration models currently available for modelling a number of stressor variables; pH, total phosphorus and temperature reconstructions	Basically limited to lakes, high initial costs Few parameters
Expert opinion or best judgement	May incorporate both historical data/opinion and present day concepts	Bias may be present

Source: European Communities, 2003b

Considering more specifically HYMO conditions of lakes, standard EN 16039:2011 (CEN, 2011) on assessing the HYMO features of lakes, currently under revision, also refers to the definition given in Annex V of the WFD. In this standard, reference conditions are considered as "pristine conditions" of the lake types, and criteria to apply are given for six HYMO Processes or Forms.

When defining hydrological reference conditions and some morphological reference conditions (e.g., lake depth variations), it is important to take climate changes into account. Therefore, taking historical references as a starting point may not be appropriate. In this case, it may be useful to search for current reference conditions at sites not subject to direct human intervention (no or very low human activities in the catchment and without lake use).

4.2. Reference conditions for hydromorphology – in practice

In 2023, information was collected on the reference conditions defined for different HYMO parameters (and metrics describing them) currently included in HYMO assessment methods developed in EU countries (Argillier & Carrière, 2023). For most of these methods, reference conditions are defined for each metric whatever the origin of the lake.

With regard to natural lakes, there seems to be a consensus to seek the HYMO reference in situations without direct human impacts e.g., no upstream impoundment, no activity on the lake, no anthropogenic substrate change, no shoreline modification, etc. This is consistent with the definition in Annex V of the WFD and also with Guidance no.13 (European Communities, 2005) stating that "*for high ecological status, the values for the HYMO quality elements correspond totally or nearly totally to undisturbed conditions*". It is also consistent with Guidance no. 37 (European Communities, 2019): "*The ecological status of a water body is classified based on five classes (from high to bad) and is defined on the basis of the degree of deviation from the conditions that would occur if the water body had no or negligible pressures acting on it. Reference conditions describe this undisturbed or very slightly altered situation that is based on type-specific characteristics for all* *quality elements. This requires that hydromorphological conditions are properly considered in the definition of water body types, minimizing within-type variability at reference conditions*."

In theory, deviations from these strict conditions could be accepted when their impact on biology is weak. Thus, generally based on expertise, some methods tolerate a deviation from these absolute conditions when defining the reference values of certain metrics. For exemple, shore re-enforcement scoring 0 or 1 out of 7 to 8 observed stations (Latvia, method given in the 3rd River Basin Management Plan), an average deviation of the water level of no more than 0.05 m from unregulated conditions (Sweden[7]), or the presence of trees in more than 80 % of the area in the 100 m around the lake (Hungary, method included in the 3rd River Basin Management Plan), are considered as low deviations without impact on biology.

Typologies may be useful in setting reference conditions, but in order to take into account some of the environmental characteristics of lakes, the types to be considered are not necessarily those included in the national typologies defined according to criteria A or B of the WFD. For example, Latvia is developing a completely new hydrological typology for the definition of HYMO status. In Croatia, broad types of lakes were considered i.e., oxbow, salted and reservoirs. In Greece, new criteria (e.g., water level fluctuations) were considered for grouping existing types. Similarly, in Italy the lake types obtained from the B classification system have been grouped into macro-types with lakes and reservoirs considered separately, and for each taking into account the depth and rainfall in the definition of hydrological reference conditions. In the French method, a distinction is also made in defining reference conditions for metrics according to the naturalness or human origin of the lake, as well as consideringhistorical conditions such as the presence of forest or wetland around the lake (Carrière et al., 2023).

As there is no clear understanding of the biological tolerance threshold at specific levels of HYMO alteration in natural lakes, it is probably preferable to apply the precautionary principle and consider the absence of degradation as the reference.

With regard to defining reference conditions by lake type, the decision is highly dependent on the diversity of the environmental characteristics of the lakes in question. When lake characteristics are very similar to each other (Norway) or when site-specific conditions can be defined, the use of a typology should not be required.

4.3. Special case of Heavily Modified Water Bodies, reservoirs in particular

Heavily Modified Water Bodies (HMWBs) represent dominant lake types in several countries and serve several functions, such as hydropower production, irrigation, drinking water supply, flood control, etc. According to WFD Annex V 1.1.5, "*the quality elements applicable to artificial and heavily modified surface water bodies shall be those applicable to whichever of the four natural surface water categories […] most closely resembles the heavily modified or artificial water body concerned*". Guidance No. 4 (European Communities, 2003a) recommends the application of this rule, but it also recognizes that this definition may be difficult to apply to some HMWBs. Reservoirs are ecosystems that result from substantial changes in the hydrological and morphological conditions of the original

[7] https://www.havochvatten.se/download/18.4705beb516f0bcf57ce1c145/1576576601249/HVMFS%202019-25-ev.pdf

water body. These water bodies are new ecosystems, which no longer have the primary characteristics of the river from which they originate, but which are also not similar to natural lakes, from a HYMO and functional point of view, and they host different biological communities. Thus, considering lakes classified as HMWBs, the principles above for natural lakes are generally difficult to apply. Therefore it is important to consider the site-specific conditions when reference conditions for reservoirs designated as HMWBs are described according to the description of the reference conditions for HMWB hydromorphology given in Annex V.1.2.5 of the WFD (**Table 7**). Guidance No. 37 includes specific steps for defining site-specific reference conditions (i.e., maximum ecological potential (MEP)).

Table 7. Maximum ecological potential (reference condition) for hydromorphological quality elements in HMWBs.

Element	Maximum ecological potential
Hydromorphology	The hydromorphological conditions are consistent with the only impacts on the surface water body being those resulting from the [...] heavily modified characteristics of the water body once all mitigation measures have been taken to ensure the best approximation to ecological continuum, in particular with respect to migration of fauna and appropriate spawning and breeding grounds.

Source: Annex V.1.2.5 of the WFD

Some examples of current practices can be given:

- In the Romanian method, reference conditions for reservoirs are those determined at the design stage. For example, the reference condition for water level is the difference between the designed retention level and thalweg level; the level of sedimentation at reference condition is calculated with respect to the volume corresponding to the date the reservoir was operational (1^{st} year).
- In Norway, reference conditions are a mix of the extent to which a reservoir is used and its deviation from a natural condition. This suggests a good knowledge of the conditions before regulation and each definition is reservoir-specific.
- In Latvia, reference conditions are based on alterations from natural conditions and the degree of regulation of reservoirs.
- In France, reference conditions are considered for each HMWB (i.e. site-specific). For certain metrics reflecting characteristics altered by use, the induced alteration is not considered in the definition of the reference that is not a "true reference" but only when defining a maximum reference value. For example, when calculating the metrics used to describe the artificialization of the banks, the length of the dam is not taken into account and the reference is 100% naturalness of the shoreline length without the length of the dam. Conversely, for the other metrics not directly affected by use, the reference conditions are identical to those of natural lakes (Carrière et al., 2023).
- Lithuania suggests that reference conditions should be defined after the reservoir has been built but before it is put into operation. These reference conditions for hydromorphology should be compatible with the environmental objectives set for other water bodies whose status depends on the operational pattern of the reservoir.

In addition, it should be noted that according to Guidance no. 37, "*The mitigation measures for defining MEP should be a selection of measures which are relevant to each of the hydromorphological alterations, ecologically effective and which alone or in combination ensure the best approximation of ecological continuum*." Despite the recommendations included in this guidance, defining reference conditions of HMWBs in practice still represents a challenge and requires dialogue both with stakeholders and environmental management structures. In conclusion, WG ECOSTAT lake hydromorphological assessment group could engage in a special reflection on the subject, but these initial elements may allow different approaches to be tested and guide thinking according to the characteristics of the ecosystems to be assessed.

5. From metrics to indices – guiding principles for the aggregation of metrics/features

5.1. General considerations

Developing an index from several metrics involves various steps, all of which are likely to influence the final index value. Numerous practical recommendations have been published to help in the choices that have to be made at each of these steps (Nardo et al., 2005; Hering et al., 2006; OECD & JRC, 2008). The criterion shown to be the most influential is related to the choice of the metrics to include in the index (Zucchetta et al., 2020). Under certain conditions, these authors showed an increase in the level of correlation between the multimetric index and the stress, with an increasing number of metrics (varying between 4 and 12 in their study). However, the unbalanced number of metrics used to describe several parameters can influence the value of the final index obtained after aggregation. Indeed, equal weighting of the metrics will result in giving more weight to the parameter with the largest number of metrics (OECD & JRC, 2008; Dobbie & Dail, 2013). This can be justified when this parameter is suspected as being more important for ecosystem structure or functioning, but this assumption is often difficult to demonstrate using robust scientific data. The quality or availability of the data required for the metric calculation may also be criteria used to weight the metrics (Freudenberg, 2003). Even if they have their limitations, statistical methods exist to overcome this issue (OECD & JRC, 2008).

One critical step is to determine the appropriate combination rule to apply, and several techniques have been proposed in the literature. These range from the simplest, such as averaging the metric scores, to more sophisticated methods that consider the sensitivity of the metrics to stressors and their variability (Drouineau et al., 2012). Alternative aggregation methods include using extreme values (minimum or maximum) or logic rules. The approach of taking the value corresponding to the worst-case scenario, known as the "one out/all out" principle, was strongly promoted during the implementation of the WFD, often in combination with averaging (European Communities, 2005).

In theory, six metrics, one for each parameter describing hydrological and morphological changes, could be included in the lake hydromorphology index. However, the inclusion of additional metrics can be promoted to cover all the HYMO features listed before and to improve the quality of the assessment tool.

5.2. Calculation of a HYMO index

Calculation of a hydromorphological quality index is required for reporting on the ecological status of lakes. Applying Guidance No. 13 (European Communities, 2005) to assess a lake's hydromorphological condition can provide a framework for developing this HYMO index.

Schematically, the procedure can be broken down into three calculation steps illustrated on **Figure 12** and described below.

— Step 1

Each of the two broad categories of HYMO alterations – i.e., hydrological regime and morphological condition – is described with one or several metrics related to one or several of the six hydromorphological quality elements given in Annex V of the WFD. These hydromorphological quality elements (and related metrics) are chosen to best reflect the alterations observed in each country. In other words, in the absence of certain alterations (e.g., connection to groundwater on **Figure 12**), calculation of the hydrological regime will not consider this stressor.

It should be noted, however, that certain pressures may emerge in future, and that monitoring all hydromorphological stressors may prove useful in the long term.

In the absence of more precise information on the relative importance of different parameters, no weighting of the metrics is generally preferred (Argillier & Carrière, 2023) but attention has to be paid to the number of metrics used to describe hydrological regime and morphological condition to avoid overestimating the weight of one of them described by a larger number of metrics.

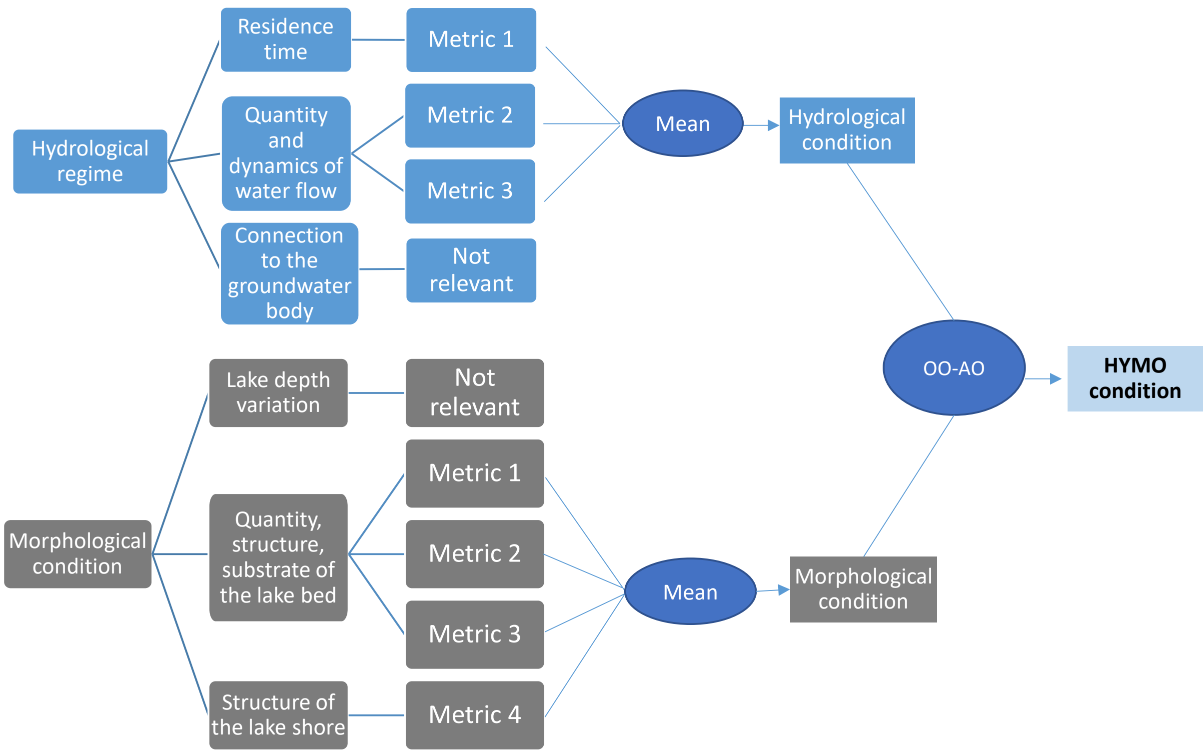
— Step 2

Averaging the values of all metrics used to describe the hydrological regime, and doing the same for the metrics used to describe the morphological conditions provides two values.

— Step 3

The final lake HYMO condition is then determined by the worst condition among the hydrological metrics and the morphological metrics according to the one out - all out (OO-AO) principle.

Figure 12. Example of HYMO index development.



Source: Own illustration

However, it is important to note that other methods of combining metrics may be better suited to the HYMO data available and the characteristics of the water bodies being measured.

6. How to relate HYMO and biology and setting boundaries

6.1. Effects of hydromorphological changes on aquatic biota

Human activities within lakes and lake catchments result in habitat modifications, which may have serious ecological consequences, although the impact of HYMO alterations on the condition of lake biota is not fully understood in Europe (Boon et al., 2019; Poïkane et al., 2020a).

The effects of hydrological and morphological transformations on biota are diverse (**Table 8**) and depend on both the type of pressure and the variability of the organisms' susceptibility/resistance to external factors and their ecology (Lyche-Solheim et al., 2013; Ciampittiello et al., 2017).

Morphological pressures result from an array of human activities that modify the physical structure of habitats, mainly in the shore zone. They usually significantly affect littoral biota such as aquatic macrophytes and zoobenthos as well as fish, whereas the impact on phytoplankton is less adverse (see part 6.2.). Unlike morphological modifications, hydrological pressures are better known for their significant impact on pelagic organisms, i.e., phytoplankton and zooplankton assemblages.

Table 8. Summary of the impacts on aquatic biota caused by hydromorphological pressures; "-": no impact, X: impact, XX: strong impact; PP: phytoplankton, MP: macrophytes, BMI: benthic macroinvertebrates, F: fish.

Hydromorphological features	PP	MP	BMI	F
Shoreline	-	X	X	X
Structure of the littoral area	-	XX	XX	XX
Quality and quantity of the substrate	-	XX	XX	XX
Increase of the sediment of the bottom	X	X	X	X
Water level fluctuation	X	XX	XX	XX
Influence of groundwater	-	-	-	-
Residence time	XX	-	X	X
Changes in the thermocline depth, stability, stratification	XX	-	-	-

Source: Modified from Ciampittiello et al., 2017

Qualitative and quantitative changes in different groups of organisms may vary over time, which result from the time of their reaction to changes. The scale of the HYMO impacts of lakes on organisms is not yet sufficiently understood. Most studies focus on one selected group of organisms (Poïkane et al., 2020a), while more complex reviews are still scarce. The following section reviews the literature on the impact of HYMO elements as listed in the WFD on the aquatic biota used to assess the ecological status or the ecological potential of lakes.

6.2. Examples of observed biological responses and underlying processes

This paragraph is a non-exhaustive review of studies highlighting the links between biology and hydromorphology with explanations of the underlying processes involved. An additional review was published in 2020 (Poïkane et al., 2020b).

6.2.1. Phytoplankton

Phytoplankton has been shown to respond mainly to hydrological alterations of lakes.

Changes in water level determine the composition and biomass of phytoplankton (Hutorowicz, 1992a; Huszar & Reynolds, 1997) both annually and seasonally (Naselli-Flores, 2000). High water levels are mainly characterized by nanoplankton, whereas at low water levels, diatoms and cyanobacteria dominate (Huszar & Reynolds, 1997). Lowering the water level may result in the release of dissolved inorganic forms of nitrogen from the hypolimnion, which becomes available to algae. Higher availability of light and nitrogen increase phytoplankton biomass and change species structure (Valeriano-Riveros et al., 2014). Comparing the impact of water level regulation in two lakes with similar morphometric conditions, Solis et al. (2016) found more intense development of toxic cyanobacteria in a reservoir with artificial water level regulation than in a lake with natural fluctuations. This study indicates that manipulation of water levels in morphologically altered lakes is reflected in greater seasonal changes in the taxonomic composition and phytoplankton biomass than in lakes with undisturbed hydrological conditions.

Indirect effects of water level changes have also been observed. For example, flooding of shore zones results in the washing out and transfer of organic matter decomposition products to the water column, creating anaerobic conditions that favour the release of phosphorus from bottom sediments. Consequently, an increase in the biomass of phytoplankton and modification of species composition were observed after artificial damming (Gołdyn, 1990).

Walz and Welker (1998) showed a positive relationship between chlorophyll concentration and water retention time. Reduced retention time stimulates an increase in phytoplankton biomass (e.g., due to the loading of nutrients from inflowing tributaries, enhanced temperature of stagnant water), whereas an increase in water flow reduces biomass because of mechanical leaching (Huszar & Reynolds, 1997; James et al. 2002).

The lowering of the water table in lakes leads to the mixing of waters below the epilimnion and release of nutrients, which simulates an increase in algal biomass. This is mainly related to the intake of water from the lake, which increases the ratio of the volume of the mixing zone to the volume of the euphotic zone. As a result, algae remain longer in the aphotic zone, leading to the taxonomic reconstruction of the community. The dynamics of algal development (especially cyanobacteria) can be disrupted by a shorter stratification period and taxonomic homogenization between littoral and pelagic phytoplankton can be observed (Hutorowicz, 1992b).

Dredging shallow eutrophic lakes may reduce cyanobacterial blooms (owing to a reduction of nutrients in sediments) and influences bacterioplankton communities (Wan et al., 2021).

Changes in lake morphology that create new wind shelter zones (e.g. construction of soft sediment islands), have been shown to have an impact on phytoplankton production and, more generally, on the trophic structures of certain shallow lakes (Jin et al., 2022).

6.2.2. Macrophytes

Aquatic macrophytes are a key component in the functioning of lake ecosystems. The presence and abundance of emergent, floating leaved and submerged species is affected by various factors. Water depth influences light availability and thus determines the depth zonation of aquatic plants. Thus, the higher the proportion of the lake area suitable for plant growth, the greater the functioning role aquatic plants are likely to play in lake ecosystems.

Under natural conditions, aquatic plants usually adapt to the prevailing hydrological regime and, consequently, to natural (even significant) water fluctuations (Mitsch & Gosselink, 2000). However, non-natural increases of the amplitude of water level fluctuations caused by human activities change the extent of the littoral zone and the heterogeneity and diversity of habitats, alter the light availability and sedimentation processes, and increase the supply of nutrients (Hussner et al., 2010; Logez et al., 2016). All these lead to a deterioration in the quality of the macrophyte habitats and significantly affect biodiversity and the taxonomic composition of macrophyte communities (Smith et al., 1987; Hellsten, 2000; Logez et al., 2016; Wang et al., 2016), as does direct physical interference, such as dredging the bottom (McParland, 2006). An artificial lowering of the water level was shown to result in the decrease of reedbed area (Yamamoto et al., 2006) as well as its biomass and ability to absorb nutrients (Ławniczak et al., 2010). Furthermore, Schmieder et al. (2004) showed that an increase in the maximum annual water level by 1 m in Lake Constance (Germany) resulted in a loss of 24% of the rushes.

Periodic exposure of large areas of the littoral to oxygen and sunlight results in quantitative and qualitative changes of macrophyte structure towards the dominance of emergent vegetation (Partanen & Hellsten, 2005). Lowering of the water table interrupts hydrophyte zonation owing to the increased availability of light in the deeper parts of the littoral (Blindow et al., 1993). It also increases the depth of the freezing zone, which in the case of Scandinavian lakes leads to the elimination of large isoetids and some vascular plants, but promotes populations of small isoetids (Hellsten, 2000; 2001; Hellsten & Mjelde, 2009). Increasing the freezing zone within the water column affects the mechanical destruction of macrophytes during the ice period resulting in helophytes being "mown" by the ice (Hellsten, 1997).

On the other hand, flooding of the lake surroundings caused by raised water levels stimulates surface runoff, which accelerates eutrophication. This results in the development of phytoplankton, which reduces the transparency of the water, reduces the availability of light, and causes the withdrawal of hydrophytes (Middleboe & Markager, 1997; Kolada, 2014). A sudden increase in water levels increases the process of soil erosion, thus affecting the physical properties of sediments, destroying shore vegetation, and reducing the taxonomic diversity of macrophytes. Ultimately, the zonal arrangement of lake vegetation may be disturbed, with a simultaneous taxonomic depletion of macrophyte communities (Hellsten et al., 1996). The disappearance of many plant species or a reduction in their range and density is then observed, in particular of stoneworts (charophytes) and marsh species (Hellsten et al., 1996).

Lakes with an unnatural hydrological regime (reservoirs subject to water level regulation) are more prone to alien invasions, host more alien species, have lower species richness, and a smaller number of rare species than lakes with a natural hydrological regime (Hill et al., 1998). However, a periodic drop in water level or periodic drying of the flood zone may also stimulate the development of macrophytes (Van Geest et al., 2005; 2007).

Morphological alteration of lakes is also known to impact macrophytes communities (Akasaka et al., 2010). The bottom substrate strongly determines the composition and abundance of both macrophytes and benthic fauna (Peters & Lodge, 2009). Ecological consequences of shoreline armouring, including the use of riprap and retaining walls, is manifested in a decreased abundance of submerged and emergent macrophytes and a reduced amount of coarse woody material (Chhor et al., 2020).

The development of infrastructure on the shores of lakes is associated with the elimination of tree cover, contributing to reduced shading of the water surface. It has a direct impact on species diversity, biomass and primary production of macrophytes (Sand-Jensen et al., 1989; Abernethy et al., 1996). Shading also affects aquatic plants indirectly by influencing the chemical changes of nutrients in water (Wilcock et al., 2004). The effect on submerged vegetation is mainly related to increased surface runoff, which contributes to the reduction of water transparency and changes in the bottom substrate. A reduction of reeds and floating leaf vegetation was also shown in response to dense residential development in the immediate vicinity of the lake, probably because of littoral habitat degradation (Jennings et al., 2003).

6.2.3. Macroinvertebrates

The impact of water level fluctuations on macroinvertebrate species is well documented. Macroinvertebrates have adaptations that allow them to survive with natural water level fluctuations but exceeding this natural amplitude causes ecological stress, leading to a significant reduction in faunal abundance and diversity in reservoirs (Aroviita & Hämäläinen, 2008; Zohary & Ostrowsky, 2011; Soszka et al., 2012). Invertebrates with long life cycles are particularly vulnerable to water level changes. During metamorphosis, many macroinvertebrates, including mayflies and dragonflies, climb onto the emergent parts of macrophytes where they shed their exoskeletons and transform into adult individuals (Thorp & Covich, 1991). A sudden rise in water level results in flooding of vegetation, prevents the metamorphosis of benthic larvae into imagines and significantly reduces their chances for survival and further reproduction. Water level fluctuations in lakes create new ecological niches for alien and invasive species (Zohary & Ostrowsky, 2011) that eventually displace native species (Meinis & Meinis, 2008).

Water level fluctuations have remarkable effects on littoral habitats and on the structure and functioning of macrofaunal and meiofaunal assemblages. Furthermore, an increase in water level may cause erosion, and rolling of the littoral substrate that pose macro- and meio-faunal assemblages under stress (Boggero et al., 2022).

Macroinvertebrates are also very sensitive to lake morphology and its alteration. A decrease in taxonomic richness is mainly linked to the lower habitat diversity and complexity of developed shorelines (Brauns et al., 2007a) and may have far-reaching consequences for the whole lake ecosystem (Brauns et al., 2011; McGoff et al., 2013b). The bottom substrate creates habitats for benthic organisms and its size and nature strongly determine the composition and abundance of benthic fauna (Tolonen et al., 2001; Mavromati et al., 2023). These authors showed that the type of substrate (sandy, stony or macrophyte-covered) is a statistically significant factor influencing the number of benthic macroinvertebrate taxa, the relative abundance of Oligochaeta and the relative abundance of Odonata. Particular attention should be paid to coarse plant material. Coarse organic matter serves as a refuge and provides development sites for insect larvae (Christensen et al., 1996; Harmon et al., 2004). Its removal and decline resulting from changes to areas adjacent to the water may affect the species number and taxonomic composition of invertebrates. Dredging causes an increase in water depth, turbidity, and changes in sediment composition (an increase in the proportion of sand and a decrease in clay), and has been found to decrease macrozoobenthos taxon richness and diversity indices (Meng et al., 2018).

Lake sites with artificial shore reinforcement have been shown to have reduced abundance and diversity of macrozoobenthos compared with sites lack of reinforcement (e.g., Lake Geneva, Bänzinger, 1995). In particular, the presence of beaches and shore reinforcement using riprap has been demonstrated to affect adversely the species richness and abundance of various groups of zoobenthos, while the number of tolerant Chironomidae increased (Brauns et al., 2007b). However, Søndergaard and Jeppesen (2007) noted that shoreline modifications may not have a negative impact on littoral invertebrates, provided that the shoreline structure remains varied and provides a variety of habitats. Doi et al. (2010) observed an increase in benthic abundance owing to the concreting of pond shores, which led to the disappearance of fish foraging areas and a reduction of stress caused by predators.

Alterations to the riparian vegetation of lakes can also have impacts on littoral benthic macroinvertebrate assemblages (e.g., number of taxa, number of families and Margalef diversity index) and the sensitivity/tolerance group (Littoral Fauna Index, Biological Monitoring Working Party) (Free et al., 2009; Peterlin & Urbanic, 2013; Mavromati et al., 2021).

6.2.4. Fish

In order to complete their life cycle, in particular for resting, feeding and reproduction, fish species are dependent on many morphological factors, such as habitat complexity, specific substrate type or the presence of macrophytes (Jennings et al., 1999; Kaufmann et al., 2014; Logez et al., 2016). In addition, fish distribution can be influenced by the hydrological regime.

Many species use the littoral zone as a spawning ground (Winfield, 2004) or as a habitat for fry (Yamamoto et al., 2006). Water level fluctuations have a direct impact on habitat availability for fish (Tonn & Magnuson, 1982) and their reproductive success (Martin et al., 1981). In spring in Lake Võrtsjärv (Estonia), a rise in water level led to an increase in the spawning area for pike, which contributed to its increased abundance (Järvait & Pihu, 2002). Raising the water level led to the disappearance of reed beds, which were primarily the habitat for cyprinid fry. Owing to a decrease in water level in Lake Constance, the fry of *Lota lota* were observed to migrate to the pelagic zone much earlier than in years when changes in water level were not significant. This was most likely related to the loss of habitats that provide hiding places for juvenile forms (Fisher & Ohl, 2004).

Water level fluctuations also have indirect effects on the distribution and condition of fish species. For example, the raised water level was shown to decrease lake temperature, significantly improving the living conditions of some species (e.g., whitefish; Järvat & Pihu, 2002). Decreasing the water level in lakes can cause or intensify periodically occurring low oxygen concentration (hypoxia) or lack of oxygen (anoxia), leading to serious ecological stress or even to fish kills (Stefan et al., 2001; Wetzel, 2001).

Dredging the bottoms of lakes has an adverse impact on fish fauna (Wenger et al., 2017), as disturbing the sediments increases water turbidity. Some species, such as rainbow trout and salmon, avoid areas where even the slightest turbidity occurs (Collin & Hart, 2015). Turbidity often impairs vision, leading to disturbances in activities requiring the use of sight. This is particularly important for fish species with a pelagic larval phase, for which the ability to find a suitable habitat is crucial for early development and survival. If fry settle in a suboptimal habitat, they become more vulnerable to predation and their growth rate may slow down because ofa lack of access to an adequate food supply (Coker et al., 2009; Feary et al., 2009). Dredging can also lead to the mechanical removal of eggs and juvenile fish from dredged areas (Reine et al., 1998), in addition it is costly (Panagos et al., 2024).

The diversity of littoral habitats influences fish species distribution and abundance (Lewin et al., 2014; Logez et al., 2016). For example, it was shown that the relative abundance of some fish species varied between shorelines with and without reinforcement. Largemouth Bass (*Micropterus salmoides*) were more abundant at natural shoreline sites than at riprap shoreline sites. Conversely, Rock Bass (*Ambloplites rupestris*), Pumpkinseed (*Lepomis gibbosus*), an invasive alien species of EU concern (Official Journal of the European Union, Commision Implementing Regulation (EU) 2019/1262), but also Bluegill (*Lepomis macrochirus*), Yellow Perch *(Perca flavescens*), and baitfish from the family Cyprinidae were more abundant at armoured sites. The growth rates of fishes may also be affected by pressure from residential development. As demonstrated by Schindler et al. (2000), the size-dependent growth rates of two common fish species (*Lepomis macrochirus* and *Micropterus salmoides*) were negatively correlated with the degree of residential development along the shoreline.

A study was carried out on fish communities in some lakes in the Southern Alps by Mor et al. (2022) to verify the effects of habitat alterations and other stress factors on the resistance of native compared with non-native species. The authors reported that alien species were more competitive, more resistant to habitat alteration in the riparian and littoral areas, and more resistant to morphological alterations in general.

6.3. How to use biology and hydromorphology in the assessment of lake ecological status

For European lakes, 37 methods for assessing the status of biological elements are reported to address HYMO pressures. However, in most cases, these HYMO pressures are mentioned as a secondary pressure in addition to eutrophication, and pressure–response relationships are not always demonstrated. The methods addressing HYMO pressures and highlighting biological responses are mostly based on the communities of benthic invertebrates (14), fish (11), or macrophytes (9).

There are only four methods specifically addressing HYMO pressures (**Table 9**): these are the benthic invertebrate methods for Austria, Germany, Slovenia, and the Spanish macrophyte method (in this case, there are separate modules addressing eutrophication, HYMO and exotic species).

Table 9. Overview of Member States' HYMO-specific biological assessment methods (with relationships demonstrated with HYMO metrics), together with HYMO pressure proxies. LMI - Lakeshore Modification Index, R^2: coefficient of determination.

MS/region	Biological quality element assessed	HyMo pressure proxy	R^2	Reference
Austria	Benthic invertebrates, littoral zone	Austrian pressure index	0.41	Wolfram et al., 2017
Germany (alpine lakes)	Benthic invertebrates, littoral zone	Stressor index	0.23-0.45(1)	Miler et al., 2013b
Slovenia	Benthic invertebrates, littoral zone	LMI	0.80	Solimini et al., 2014; Urbanič, 2014
Spain	Macrophytes	Hydrological / Morphological alteration level	0.49	Camacho et al., 2020

(1) for different types of lakes
Source: Modified from Poïkane et al., 2020b

In several cases, relationships with HYMO pressure were shown for methods addressing several pressures (**Table 10**), although some values of the coefficient of determination were very low or very high. In these cases, HYMO pressure is usually described as being of second importance to eutrophication.

Table 10. Overview of Member States' biological assessment methods addressing HYMO pressures in addition to eutrophication (and demonstrating relationships with HYMO metrics). HMI - Hydromorphological Index, LHMS - Lake Habitat Modification Score; R^2: coefficient of determination.

Member State/region	**Biological quality element assessed**	**HyMo pressure proxy**	**R^2**	**Reference**
Finland	Macrophytes (transects)	Winter drawdown	0.58	Vuori et al., 2009
Germany (lowland lakes)	Benthic invertebrates, littoral zone	Morpho-index	0.1-0.25	Miler et al., 2013b
Greece	Fish fauna (benthic and pelagic gillnets)	LHMS	0.74	Petriki et al., 2017
Greece	Benthic invertebrates, littoral zone	Artificial shoreline (%)	0.476[(1)]	Mavromati et al., 2021
Lithuania	Benthic invertebrates, littoral zone	Morpho-index	0.11	Šidagytė et al., 2013
Lithuania	Fish fauna (benthic gillnets)	HMI	0.19	Virbickas & Stakėnas, 2016
The Netherlands	Benthic invertebrates, littoral zone	Shore alterations	0.45	Altenburg et al., 2007; Böhmer et al., 2014

[(1)] Spearman Rank correlation coefficient
Source: Modified from Poîkane et al., 2020b

6.3.1. Biological metrics used in BQE assessment methods for addressing HYMO pressures

Most HYMO pressure/biological impact relationships have been established with macroinvertebrates. However, some relationships have also been shown with macrophytes and fish (**Table 11**; **Shifts in** the taxonomic composition of benthic invertebrates in response to lakeshore alterations have often been reported (Urbanič et al., 2012; Urbanič, 2014). There are far fewer studies investigating the effect of lake shore alterations on functional groups of macroinvertebrates (but see García-Criado et al., 2005; Brauns et al., 2007b). An increase in the relative abundance of the collector/gatherer feeding group along the pressure gradient has been reported for lakes in Denmark, Germany, Ireland and the UK (Miler et al., 2013a); an increase in the ratio of r to K strategists was also observed for lakes in Italy, Germany, and Slovenia (Urbanič et al., 2012; Miler et al., 2013a; Miler et al., 2013b). These functional metrics are included in the Austrian and German assessment methods.

Table 12). Most of the metrics used in these assessments were based on changes in species composition, and richness/diversity.

Table 11. Metrics included in HYMO-specific lake assessment methods. Metrics are grouped into categories according to Hering et al. (2006) and Kanninen et al. (2013). ↓: metrics decrease along ecological status gradient; ↑: metrics increase along ecological status gradient.

Quality element	**Type of metric**	**Metrics and direction of change across pressure gradient**	**Member State in which metric is applied**
Macro-invertebrates	Composition / abundance	Odonata (% abundance) ↓	Germany (Alpine lakes)
		Oligochaeta (abundance) ↑	Austria
		Neozoa (% abundance) ↑	Austria
	Richness / diversity	Margalef diversity index ↓	Slovenia
		Shannon-Wiener diversity index ↓	Germany (Alpine lakes)
		Number of taxa ↓	Austria, Slovenia
	Functional	r/K = taxa ratio of r and K strategists ↑	Austria, Germany (Alpine lakes)

		Collector feeding group (% abundance) ↑	Austria, Germany (Alpine lakes)
	Sensitivity / tolerance	Fauna index (according Miler et al., 2013b)	Germany (Alpine lakes)
		Littoral fauna index (Urbanič, 2014) ↓	Slovenia
	Composition / abundance	Cover of hydrophytes ↓	Spain
Macrophytes		Total cover of macrophytes ↓	Spain
		Cover of helophytes ↓	Spain

Source: Modified from Poïkane et al., 2020b

Shifts in the taxonomic composition of benthic invertebrates in response to lakeshore alterations have often been reported (Urbanič et al., 2012; Urbanič, 2014). There are far fewer studies investigating the effect of lake shore alterations on functional groups of macroinvertebrates (but see García-Criado et al., 2005; Brauns et al., 2007b). An increase in the relative abundance of the collector/gatherer feeding group along the pressure gradient has been reported for lakes in Denmark, Germany, Ireland and the UK (Miler et al., 2013a); an increase in the ratio of r to K strategists was also observed for lakes in Italy, Germany, and Slovenia (Urbanič et al., 2012; Miler et al., 2013a; Miler et al., 2013b). These functional metrics are included in the Austrian and German assessment methods.

Table 12. Metrics included in lake assessment methods addressing HYMO as one of multiple pressures (and showing relationship with HYMO metrics). Metrics were grouped into categories according to Hering et al. (2006) and Kanninen et al. (2013). ↓ metric decrease along ecological status gradient; ↑ metric increase along ecological status gradient.

Quality element	**Type of metric**	**Metrics and direction of change across pressure gradient**	**Member State in which metric was applied**
Macroinvertebrates	Composition / abundance	Coleoptera, Odonata and Plecoptera (COP) (%) ↓	Lithuania
		Odonata (% abundance) ↓	Germany (lowland lakes)
	Richness / diversity	Hill`s number ↓	Lithuania
		Number of Coleoptera, Ephemeroptera, Plecoptera taxa (CEP) ↓	Lithuania
		Number of Epheremoptera, Trichoptera, Odonata taxa (ETO) ↓	Germany (lowland lakes)
		Relative richness of indicator (for water type) species in a sample ↓	The Netherlands
	Functional	Habitat type lithals (%) ↓	Germany (lowland lakes)
	Sensitivity / tolerance	Average Score Per Taxon ASPT ↓	Lithuania, Poland
		Fauna index (Miler et al. 2013b)	Germany (lowland lakes)
		Positive and negative dominant taxa (% abundance) ↓↑	The Netherlands
		Type-specific indicator taxa (% abundance and % taxa number) ↓	The Netherlands
Macro phytes	Composition / abundance	Proportion of type-specific taxa ↓	Finland
		Percent Model Affinity ↓	Finland
	Sensitivity	Reference Index ↓	Finland
Fish fauna	Composition / abundance	Non-native + translocated species (% biomass) ↑	Greece, Lithuania
	Functional	Benthivorous species (% biomass) ↑	Lithuania
		Omnivorous species (% biomass) ↑	Greece
		Stenothermic species and *Perca fluviatilis* (% biomass) ↓	Lithuania
	Sensitivity / tolerance	Number of type-specific sensitive species ↓	Lithuania
	Age structure	Mean weight of *Rutilus rutilus* ↓	Lithuania

Source: Modified from Poïkane et al., 2020b

6.3.2. Common biological assessment methods addressing HYMO

Many studies have aimed to develop EU-wide (Moss et al., 2003) or regional assessments methods (Lyche-Solheim et al., 2013; Phillips et al., 2013) to be applicable and used by more than one country (common metrics). For HYMO pressures, such common indices have been developed for lake benthic invertebrates (Miler et al., 2013b; Poïkane et al., 2016) and lake macrophytes (Mjelde et al., 2013) (**Table 13**). However, in most cases, countries have chosen to develop country-specific indices and the application of these common indices has been limited, in particular owing to biogeographical differences (Birk et al., 2013; Miler et al., 2013b).

Table 13. Overview of common biological assessment methods addressing HyMo pressures (with relationships demonstrated with HYMO indices). LIMCO – Littoral Invertebrate Multimetric based on Composite samples; LIMHA - Littoral Invertebrate Multimetric based on habitat samples; ALP-ICM – Alpine region intercalibration common metric; CB-ICM – Central Baltic region intercalibration common metric. EPTCBO: Ephemeroptera, Plecoptera, Trichoptera, Coleoptera, Bivalvia and Odonata taxa; ETO: Ephemeroptera, Trichoptera, Odonata taxa; R^2: coefficient of determination.

BQE; pressure and proxy	Member States	Metrics included in the pressure index	R^2	Reference
Benthic invertebrates - shore morphological alterations; Pressure proxy - hydromorphological stressor index	Denmark, Germany	LIMCO: Margalef diversity; Gatherers and collectors %; Chironomidae %; No. EPTCBO taxa	0.48	Miler et al., 2013b
		LIMHA (stones): Margalef diversity; Gatherers and collectors%; Coleoptera %; No. EPTCBO taxa	0.53	
	Finland, Sweden	LIMCO: No. families; Shredders %; Crustacea %; No. Odonata taxa	0.15	
		LIMHA (macrophytes): Evenness, Predators %; Coleoptera %; EPTCBO taxa %;	0.19	
	Ireland, UK	LIMCO: Margalef diversity; Gatherers and collectors%; Diptera%; No. ETO taxa	0.22	
		LIMHA (sand): Shannon diversity; Swimming/ diving %; Diptera %; EPTCBO taxa %	0.50	
	Central and Northern Italy	LIMCO: Margalef diversity; r/k ratio; Odonata %; No. ETO taxa	0.24	
		LIMHA (silt): Shannon diversity; Psammal %; Oligochaeta %; EPT taxa %	0.16	
Benthic invertebrates - shore morphological alterations; Pressure proxy - morphological index	Germany, Slovenia	ALP-ICM: Fauna index; No. taxa; Gatherers and collectors %; r/k ratio	0.32	Poïkane et al., 2016
	Belgium, Germany, Estonia, Lithuania, the Netherlands, UK	CB-ICM: ASPT index; No. EPTCBO taxa; Lithal %; ETO taxa%	0.18	
Macrophytes – water level fluctuations; proxy – winter drawdown	Finland, Norway	Water level drawdown index: ratio between sensitive and tolerant macrophyte species	0.09 (lakes) 0.77 (storage reservoirs)	Mjelde et al., 2013

Source: Poïkane et al., 2020b

6.3.3. Guidance for MSs on how to link HYMO stressors and biology

Hydromorphology is considered a supporting quality element, which means that it is intended to support the living conditions of aquatic biota. Therefore, HYMO assessment systems must reflect the level of HYMO alterations that affect (or do not affect) the functioning of biological assemblages and, for developing methods, pressure–responses relationships have to be established.

It is important first to describe the pressure gradient with appropriate pressure metrics (cf. metrics in Chapter **Error! Reference source not found.** that allow characterization of the hydromorphological features of lakes). MSs are welcome to elaborate as many indices as needed to assess different pressures affecting their waters, e.g., eutrophication, salinization, acidification. HYMO alterations are considered another set of pressuresthat require dedicated and responsive biological metrics.

These biological metrics can be those already aggregated in an existing biological quality index or more specific metrics aggregated or not (see for examples **Table 11**, **Shifts in** the taxonomic composition of benthic invertebrates in response to lakeshore alterations have often been reported (Urbanič et al., 2012; Urbanič, 2014). There are far fewer studies investigating the effect of lake shore alterations on functional groups of macroinvertebrates (but see García-Criado et al., 2005; Brauns et al., 2007b). An increase in the relative abundance of the collector/gatherer feeding group along the pressure gradient has been reported for lakes in Denmark, Germany, Ireland and the UK (Miler et al., 2013a); an increase in the ratio of r to K strategists was also observed for lakes in Italy, Germany, and Slovenia (Urbanič et al., 2012; Miler et al., 2013a; Miler et al., 2013b). These functional metrics are included in the Austrian and German assessment methods.

Table 12 and **Table 13**).

Ideally, the relationship should then be sought between the value indicative of the hydromorphological status of the lake (i.e., HYMO condition in **Figure 12**) and biology, whether it is an existing index or one or more new biological metrics.

For HYMO assessment systems in MSs, at least three case scenarios are used:

- MS has no hydromorphological assessment method;
- MS has hydromorphological assessment method but there is no correlation between HYMO metric(s)/index and any BQE in use.
- MS has hydromorphological assessment method with a clear correlation between HYMO metric(s)/index and biological metrics;

6.3.3.1. MS has no hydromorphological assessment method

This scenario requires elaboration of the hydromorphololological method reflecting the level of HYMO alteration and the effect on biological assemblages at the same time. The most relevant steps include:

- Identify hydrological and/or morphological pressures most relevant for lakes in your country, e.g.., water level fluctuations, water abstractions, land use changes, touristic and recreational use. Remember that certain hydromorphological alterations may not be relevant today, but may become so in future. This argues in favour of characterizing all hydromorphological pressures, even if it means considering some of them as null in the overall assessment of hydromorphological alteration (next step).
- Select/elaborate HYMO metric(s)/index describing pressure(s); to do this, information included in Chapters 2, 4 and 5 may serve as valuable support.
- Test the relationship between new HYMO metric(s)/index and BQEs because most existing biological metrics were elaborated to detect pressures other than HYMO, such as eutrophication, acidification, and organic pollution. It is very likely that none of the existing biological metrics will respond to HYMO pressures; in this case, new biological metrics/index responsive to HYMO pressures must be elaborated;
- As previously presented in this chapter, not all biological elements respond to HYMO pressures equally well; biological metrics already included in existing methods presented earlier may serve as good examples developing biological metrics.

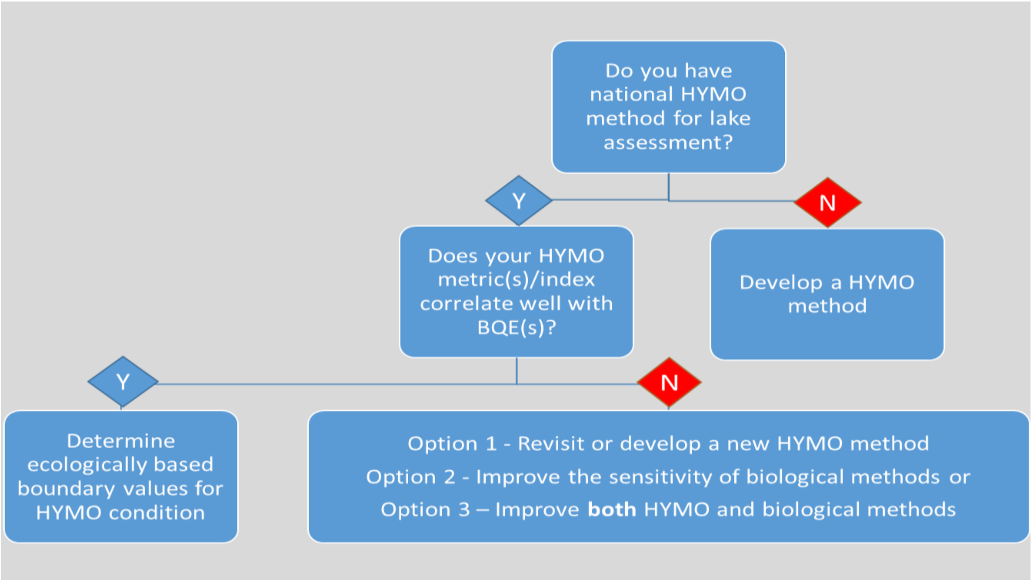
6.3.3.2. MS has hydromorphological assessment method but HYMO condition is not correlated with BQEs

The most complex situation is when a MS has developed and uses HYMO methods but they do not correspond to biotic conditions (no statistically significant relationships between HYMO and any of the BQEs). In such cases, MSs should either elaborate a new HYMO method or review their existing biological methods to find significant statistical links. The latter can be done through:

- reviewing HYMO method to better correlate with existing BQEs,
- elaborating new biological metrics responsive to existing HYMO index,
- revisiting and revising **both** HYMO and biological metrics.

These situations are illustrated in **Figure 13**.

Figure 13. Illustration of the decision-making process for including hydromorphology in the assessment of lake ecological status.



Source: Own illustration

The different steps in the figure above can be described as follow.

— **Step 1. *Overview of existing methods to assess HYMO condition***

Check if the national method for assessing the HYMO condition of lakes is available in your country.

- If you have such a method, you can directly go to step 2.
- If you do not have a method to assess the HYMO condition of lakes, the development of a dedicated method that links HYMO pressure and biotic condition is necessary. Based on a literature review and the present document, you should identify a list of potential HYMO metrics describing the most relevant HYMO pressures. It should be emphasized, however, that the range of a hydromorphological pressure may be country-specific and some relationship may be observed in one biogeographical area and not in another. After developing the method, go to step 2.

— **Step 2. *Correlation of HYMO and BQE***

Check whether the HYMO condition correlates with the Ecological Quality Ratios (EQRs) for biological elements. This is recommended where, in your expert judgement, the metrics included in the BQE method that respond (for example) to eutrophication, are also considered to be sensitive to the HYMO stressors.

- If your HYMO metric(s)/multimetric index satisfactorily links with biological conditions[8], go to step 3.

[8] Biological responses to alterations must be ecologically interpretable. The quality of the relationship can only be interpreted as a function of the method used to establish it. It is therefore left to the operator's judgement.

- If it is not correlated, go to step 4.

— **Step 3. *Determine ecologically based boundary values for HYMO metrics/multimetric***

The most challenging aspect of this situation is determining the boundary values for the HYMO index to ensure that the HYMO assessment accurately represents the extent of biota alteration caused by morphological and/or hydrological changes. The work of Phillips et al. (2019) can be helpful in this step.

— **Step 4. *Revisit and revise or develop new HYMO index***

If the existing HYMO condition is not well correlated with BQEs, three options are possible:

- Revisit and revise existing HYMO metric(s)/index. You can select metrics reflecting the stressors observed on your lakes (chosen, for example, among those given in Chapter 2) or create a multimetric index following the recommendations in Chapter 5). If you are using a HYMO index that has to be revised, you can check if its component metrics are correlated with the biological metrics/multimetric or you can select new metrics based on the overview presented in Chapter 2.
- Develop new BQE method responsive to HYMO pressures. In this case, you can use information provided in this chapter and in the bibliography to the existing BQE indices (Tables 11, 12 and 13) listed below.
- Revisit and revise **both** HYMO and biological metrics.

6.3.3.3. MS has hydromorphological assessment method and HYMO index is correlated clearly with biological metrics

This scenario is the most desirable, where HYMO metric(s)/index already links with biotic condition, i.e., the correlation between HYMO metric(s)/index and at least one biological metric at a sufficiently high statistical level. The most challenging issue in this case is setting boundary values for HYMO metric(s)/index so that the HYMO condition reflects the extent of biotic alteration caused by morphological and/or hydrological alterations.

6.4. Success story – one example for Finnish lakes

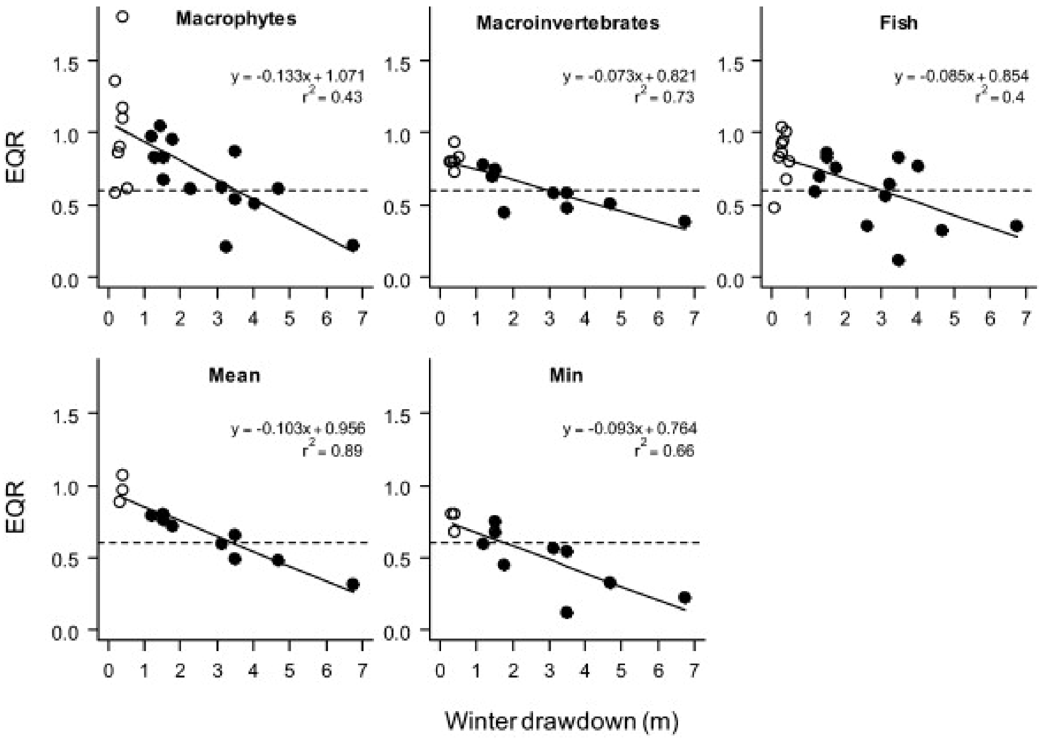
In Finland, the most prevalent HYMO pressure affecting the ecological status of lakes is water level regulation for hydropower and flood protection. More than 300 natural lakes are affected to some extent by water level regulation, encompassing approximately one-third of the total lake area in the country. In a typical annual regulation cycle, water level is drawn down 1-7 m during each winter when the need for electricity is the highest and water inflow to lakes is low. Before spring snowmelt, space in the lake basin needs to be created to accommodate snowmelt runoff. Owing to these hydrological changes and natural morphological characteristics of Finnish lakes (shallow and gently sloping shores), water level regulation has a strong impact on lake littoral habitats. Large portions of the littoral area can be frozen and depressed by thick ice (50-100 cm) as a result of lowering the water level during winter and early spring. Moreover, natural spring floods are dampened and delayed, whereas water levels are maintained relatively stable during summer and early autumn. The biological community structure of lake littorals is strongly affected by these hydrological alterations, especially in the most heavily regulated lakes.

Studies conducted on Finnish lakes (e.g., Aroviita & Hämäläinen, 2008; Keto et al., 2008; Sutela et al., 2013) reveal remarkable and consistently strong relationships between littoral biological quality elements (such as macrophytes, benthic macroinvertebrates, and fish) and the HYMO impact (winter water level drawdown). Based on these studies, a decrease in water level of 3 m during winter could serve as a critical threshold level, indicating a high risk of failing to achieve good ecological status (Keto et al., 2008; Sutela et al., 2013).

Consequently, lakes with ecological status below 'good' and experiencing winter drawdown of more than 3 m or more than 50 % of mean depth (Aroviita & Hämäläinen, 2008; Keto et al., 2008; Sutela et al., 2013) can be provisionally designated as HMWBs. Subsequently, the designation test procedure (Guidance No. 4; European Communities, 2003a) can be applied to a final designation of HMWBs with less stringent environmental objectives, if achieving good ecological status proves unattainable without significant harm to the water body's important uses.

In these studies, several biological metrics were developed or examined against the stressor impact gradient. For littoral fish, a stressor-specific multimetric index was developed by combining three individual metrics (total fish density, proportion of disturbance-sensitive species, and occurrence of juveniles in disturbance-sensitive species). Disturbance-sensitive species include lithophilic or benthic fish species such as *Phoxinus phoxinus*, *Cottus gobio*, *Barbatula barbatula*, and *Lota lota* (Sutela & Vehanen, 2008). The occurrence of juveniles in disturbance-sensitive species is quantified as an average number of disturbance-sensitive species with 0+ age individuals per electrofished littoral areas in a lake. For each metric, EQRs were calculated (EQR = metric value/average value in near-natural reference lakes) and combined as a mean of the three metric EQRs (see details in Sutela et al., 2011). For macrophytes, the metrics used include the occurrence of type-specific reference taxa, abundance of all taxa, and abundance of taxa sensitive to regulation. The three metrics used to quantify benthic macroinvertebrate community attributes are the occurrence of type-specific reference taxa, Percent Model Affinity and EPT/Non-EPT-taxa ratio. Otherwise, the combination of individual metric EQRs to final metric scores followed the same approach as for fish (**Figure 14;** see details in Sutela et al., 2013).

Figure 14. Relationships between mean EQRs (rescaled Ecological Quality Ratios) and water level drawdown for three groups of organisms for the 12 lakes with data for all three groups, and their mean EQR and minimum EQR in non-regulated reference lakes (open dots) and regulated lakes (black dots). In each panel, the solid line represents the fitted linear regressions and the dashed line indicates EQR = 0.6 (good/moderate status class boundary).



Source: Sutela et al., 2013

7. Conclusions

This report addresses the issue of how to describe the HYMO alteration gradient. At present, countries use many different approaches (Argillier & Carrière, 2023), including specific and non-specific metrics characterizing general degradation, such as land use. This report provides a list of metrics that can be used to account for different types of hydromorphological alterations. It also suggests ways of aggregating them to construct an indicator of hydromorphological condition. We have also attempted to gather all the information needed to calculate these indicators. Without necessarily being exhaustive, Chapter 3 presents a review of these different methods, which can be applied to a range of lake types and locations.

The choice of metrics should take into account the data available at national level, as well as their quality and ease/cost of acquisition. The national context, i.e., the nature of the main pressures observed on water bodies, is another criterion for choosing metrics. We would point out, however, that it may be preferable to consider all the hydromorphological parameters listed in Annex V of the WFD, as it cannot be ruled out that certain pressures not currently observed may become significant in the future, particularly in a context of global change which is likely to modify lake uses.

Regarding the aggregation of metrics, one simple option is proposed: an average of the metrics characterizing hydrology and morphology, then applying the OO-AO principle to the two values. Other options are possible, from the simplest to the most sophisticated. The choice of methods falls to the MS which will have to take into account the quality of the data (with a given confidence) and the importance of the stressors for their lake types. Note that expert knowledge can also be mobilized to support decision making among several aggregation options (Carrière et al., 2021).

Some very general ideas have also been put forward for the definition of reference conditions. This seems relatively straightforward for the hydromorphological parameters of natural lakes but much more complex for non-natural lakes. For these highly modified and artificial water bodies there is no real consensus and work will need to continue to achieve a shared vision of what is considered an achievable good HYMO status that considers water uses.

Despite the importance of HYMO pressures (EEA, 2018), only 16 countries have at least one biological assessment method addressing HYMO, reported to the Commission. However, the survey conducted among Member States showed that in 2020, 33 methods were applied or in development in 20 countries (Argillier et al., 2022; 2023). Most of the biological assessment methods addressing HYMO pressures lack pressure–-response relationships; only 11 methods in nine countries consider these relationships but in some their strength seems very low, leading to concerns about the significance of the outputs. It is clear that far more must be done to demonstrate the links between HYMO condition and biology. We noted that the LHMS index (Rowan et al., 2006; Latinopoulos et al., 2018), widely used to characterize lake HYMO status, has rarely been used for developing pressure–response methods. In addition, this index was not always significantly correlated with metrics of composition and diversity calculated for macroinvertebrate communities (McGoff et al., 2009; 2013a). It is well understood that HYMO pressures include several types of impacts such as shore alterations and water level fluctuations, and it is recognized that HYMO pressures have significant effects on all biological communities (Zohary & Ostrovsky, 2011; Jeppesen et al., 2015). However, among the methods reported to the Commission, most assessment tools focus on littoral benthic invertebrates, and most HYMO methods focus on shore alterations. Therefore, we hope that this report will help in the choice of new bioindicators of hydromorphological status that take into account in particular the hydrological alterations of lakes. In future it may be possible to observe the effects of HYMO alterations on other biological groups, such as the composition and abundance ofphytoplankton in response to a modification of the residence time.

In addition, although multiple-stressor assessment methods can be useful (Kanninen et al., 2013; Poïkane et al., 2017), efforts are needed to develop stressor-specific tools that environmental managers can use to identify the cause of degradation and determine the management measures needed to improve lake status. Indeed, most of the biological methods that currently address HYMO condition and reported to the Commission do not allow the impacts of different stressors to be evaluated separately. Instead, they aim to assess general degradation caused by multiple stressors.

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Annexes

Annex 1. Application of hydrological modelling in Southern and Central Europe

Considering the changes in the water cycle induced by climate change, the frequency and intensity of flood events and prolonged dry periods, the development of adequate techniques to model the quantity of available water is essential. The inputs of hydrological models are not always accurate and certain, given the uncertainties linked to the simplification of physical meteorological processes. To reduce these uncertainties several studies have used terrestrial water storage (TWS) data that can be obtained from Gravity Recovery and Climate Experiment (GRACE) (Soltani et al., 2021). TWS refers to the total vertically integrated water content in an area regardless of the reservoir (surface water, groundwater, soil moisture and permafrost, snow and ice, or wet biomass) (Ahmed et al., 2016). Among the different models used for the definition of the hydrological balance and runoff, the one most used in Europe (Andersen et al., 2005; Hinderer et al., 2006; Weise et al., 2009; Li et al., 2012; Springer A., 2019; Lenczuk et al., 2020; Wang at el., 2020), is the mesoscale Hydrological Model (mHM) GRACE satellite-based TWS anomalies. Through satellite data relating to TWS obtained through GRACE, it is possible to obtain a deeper understanding of hydrological dynamics (Moravec et al., 2019). The GRACE mission has been active since 2002 and has two satellites in polar orbit for monitoring the earth continuously with an altitude ~ 500 km, by means of a joint project between the National Aeronautics and Space Administration (NASA) in the USA and the German Aerospace Center (DLR) in Germany (Tapley et al., 2004). The variability in gravity field solutions mapped by the GRACE satellite is directly related to the redistribution of mass at or near the Earth's surface. The largest signals observable in the GRACE data are coming from spatial and temporal variations in TWS. The traditional spherical harmonics solutions produced from GRACE are typically obtained through an optimization of the gravity field data at the global scale and are generated by a number of processing centres around the world. Alternatives to this global approach include so-called "regional techniques", for which many variants exist but whose common trait is that they only use the gravity data collected over the area of interest to generate the solution (Klees et al., 2008).

Using GRACE TWS data in hydrological models is not easy as errors and assimilation techniques must be managed correctly. Recently the application of GRACE TWS was made for: i) monitoring TWS change; ii) evaluating hydrological components; iii) drought analysis and iv) detecting glacier mass balance. GRACE TWS data can also be used by advanced statistical and numerical analyses to improve the output of hydrological models by reducing the input uncertainty resulting from climate data. GRACE TWS data resolution is in the order of a day or a month, depending on the method of data analysis. The spatial resolution is a few hundred kilometres; for this reason it performs poorly for small basins (Soltani et al., 2021). To isolate the hydrological part arising from the signal from the GRACE satellites, several methods have been used which all consider that the storage of water in the continents and its changes in time and space can be obtained from gravimetric observations of the satellites. The use of GRACE satellite data in hydrological models can effectively separate TWS into its components to improve its resolution and to constrain model simulations and their parameters through data assimilation (Soltani et al., 2021).

Annex 2. Application of hydrological modelling in Norway

The precipitation-runoff model HYPE (Lindström et al., 1997) has been tested for the purpose of supporting the implementation of the EU WFD with hydrological data in Norway (Schönfelder et al., 2017; Adera et al., 2018). HYPE is a process-based semi-distributed rainfall runoff model which has been developed at SMHI (Swedish Meteorological and Hydrological Institute). The model is open-source and based on the concept of HBV (Bergström, 1992). It can be used to predict discharges in gauged and ungauged basins, as well as water quality parameters. Further information on HYPE can be found on the HYPE wiki (http://www.smhi.net/hype/wiki/doku.php).

The performance of the model was tested by comparing model output with observations at selected sites across Norway. The studies concluded that HYPE was capable of simulating seasonal and low flow conditions with acceptable quality (Schönfelder et al., 2017; Adera et al., 2018). The studies also confirmed that HYPE seems to be well suited to simulate the hydrology in unregulated basins, i.e., to simulate the reference hydrological conditions.

Based on a well-calibrated hydrological model, a set of different hydrological indices can be derived, which was exemplified with selected indices of ecological relevance (**Table 14**).

Table 14. Results for the simulation of changes in selected hydrological indices at two hydrological gauging stations Nordsetfoss and Brattset in Norway.

	Nordsetfoss			**Brattset**		
Parameter	**Index unregulated**	**Index regulated**	**Rel. change of hydrol. Index**	**Index unregulated**	**Index regulated**	**Rel. change of hydrol. Index**
Average runoff [m³/s]	85.6	42.6	-50 %	31.3	9.8	-69 %
Annual 1 day max [m³/s]	500.3	190.5	62 %	274.5	143.8	-48 %
Annual 30-day max [m³/s]	274.9	89.8	-67 %	122.4	41.4	-66 %
Annual highest 7-day average flow [m³/s]	417.3	153.4	-63 %	205.2	84.7	-59 %
Number of rises [-]	20.9	1.9	-91 %	25.9	6.4	-75 %
Number of falls [-]	14.0	1.8	-87 %	19.7	5.0	-75 %

Source: Schönfelder et al., 2017

In the context of the EU WFD, hydrological models have been used mainly to simulate the hydrological status and changes in rivers in Norway (Table 14). We foresee a wider use of hydrological models to simulate the present, future or historical conditions in lakes and reservoirs. The reasons for the relatively less common use of hydrological models to assess, for instance, water level changes in lakes and reservoirs are; i) the regulated water bodies are usually affected by hydropower regulations, and it has often been difficult to gain access to data on the operation of the hydropower plants, ii) many of the regulated lakes have been subject to regulation for many years (some since ~ 1900) and sometimes there is limited information about the situation before the lakes were regulated, iii) in general less attention, including research, has been given to assessing the HYMO pressures on lakes and their subsequent ecological effect.

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