Joint management of Latvian – Lithuanian trans-boundary river and lake water bodies (TRANSWAT) LLI-533

Methodology of lake ecosystem health assessment

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List of used abbreviations

BOD_5	biochemical oxygen demand over 5 days
BOD_7	biochemical oxygen demand over 7 days
BQE	biological quality element
Bream/RoachW%	roach and bream percentage by weight in a gill net with a mesh size 20-35 mm
Chl-a	chlorophyll-a
EQR	ecological quality ratio
EŽI	Lithuanian Lake Fish Index
GAAEs	Groundwater associated aquatic ecosystems
GDEs	Groundwater dependent ecosystems
GDTEs	groundwater associated aquatic ecosystems
GWBs	Groundwater body
LHS	the lake habitat survey
LVFI	Latvian Lake Fish Index
PerchW%	percentage of perch by weight in gill nets with mesh size of 20-35 mm.
PoMs	programmes of measures
RoachWavg	roach average weight (g) in a catch using nets with a mesh size of 20-35 mm
SWBs	Surface water body
TVs	threshold values
Wavg	average weight
WFD	Water Framework Directive (2000/60/EC)
WPUE	weight per unit of effort

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I INTRODUCTION

Achievement of good ecological and chemical status of water are main objectives of EU Water Framework Directive (WFD, 2000). Implementation of more effective measures in water bodies is the way to meet these objectives and improve water quality. As water cannot be divided by human drawn boundaries such as country borders, management of shared water bodies must be a joint effort.

The aim of the TRANSWAT project is to ensure joint assessment and management of trans-boundary river and lake water bodies which hydromorphological and/or ecological quality pose a risk for not meeting WFD's requirements.

The long-term aim of this study is to move towards Integrated Water Resources Management (IWRM) and the first attempt is made through incorporation of groundwater contribution and watershed evaluation into transboundary lake ecosystem health assessment. This methodology proposes procedures on how to carry out such complex tasks and benefit the implementation of main EU water policy - WFD.

Three trans-boundary river basins (Venta, Lielupe and part of Daugava/Dauguva) are covered by this Program area where the ecological status of many river and lake water bodies is below "good". Moreover, some river bodies are designated as "heavily modified" due to substantially altered natural conditions – the result of flow regime regulation by Hydro Power Plants (HPPs) cascades. Right now, mainly in Latvia lakes located on the cross-border area between Latvia and Lithuania are delineated as water bodies. Consequently, water quality monitoring and pressure analysis of these lakes is carried out primarily on Latvian side.

Project will address two main components: (1) assessment and management of rivers affected by HPP's cascades and (2) estimation and management of trans-boundary lakes ecological status and ecosystem health.

Second component (topic of this report) will assess the ecological status of five trans-boundary lakes located in Lielupe and Daugava/Dauguva river basins. Three lakes are a part of NATURA 2000 network. New data will be gathered during water quality and quantity monitoring, and detailed biological and fish surveys. Then, combining new data within depth analysis of lake catchments (such as assessment of surface-groundwater interaction and source apportionment modelling) the ecological status of five lakes will be evaluated. **A new joint methodology for lake ecosystem health assessment will be developed and then tested in the pilot area. This report presents the jointly developed methodology.** Finally, a harmonized Latvia-Lithuania monitoring program (MP) and Program of Measures (PoM) will be established to ensure sustainability of project results and further improvement of water quality.

1.1. Concept of Groundwater dependent ecosystems

Dynamic interactions between ground- and surface water are widely known, but the role of groundwater contribution in terrestrial and aquatic ecosystems is still too poorly understood and documented due to the spatiotemporal complexity (Terasmaa et al., 2020).

Groundwater dependent ecosystems (GDEs) are ecosystems whose current composition, structure and function rely on groundwater supply. GDEs are directly or indirectly protected by the number of European Union directives (Birds, Habitats, Groundwater, Floods) and international agreements such as the Ramsar Convention on Wetlands. Many GDTEs are included in the Natura 2000 network of protected sites. These ecosystems are typical of high value as they provide habitat for endangered species, support high biodiversity, and provide valuable ecosystems services (Kløve et al., 2011), namely, fish production, water purification and retention, climate regulation and recreation (Grizzetti et al., 2016).

Typically, GDEs are divided into two major groups terrestrial and aquatic, however, there are many subdivisions available in the literature. According to Kløve et al. (2011) GDEs can be groups into (1) rivers and lakes including aquatic, hyporheic and riparian habitats, (2) subterranean aquifers and caves, (3) wetlands and springs and (IV) estuarine and nearshore marine ecosystems. While Eamus et al. (2016) categorize GDEs as the ones that (1) reside within groundwater (e.g., karsts, stygofauna), (2) require the surface expression of groundwater (e.g., springs and wetlands) and (3) are dependent upon sub-surface availability of groundwater within the rooting depth of vegetation (e.g., woodlands or riparian forests).

Despite the chosen division all agree that GDEs provide valuable ecosystem services and are not fully understood, especially when it comes to documentation of groundwater contribution. Like surface watersheds, groundwater movement cannot be divided by human drawn boundaries such as country borders, thus assessment of transboundary water bodies should be a joint effort between neighboring countries which share the same resources (WFD, 2000). Currently, the first attempt in Baltics to jointly tackle GDTEs issues was made during Interreg Est-Lat project “GroundEco” (Retike et al., 2020), however GAAEs have not yet been addressed.

1.2. Concept of Ecosystem health assessment

“Ecosystem health” is a common term used in environmental science and management to describe the state of a system relative to a desired management target or reference condition (O’Brien et al., 2016; Rapport, 1989).

The concept of ecosystem health was firstly proposed by Costanza and Rapport (Costanza and Mageau, 1999; Mageau et al., 1998; Rapport et al., 1998). They reported that any ecosystem should maintain stability and elasticity for either long-term or sudden natural and man-made disturbance events. Since the concept of ecosystem health emerged and set new goals for environmental management in 1980s, the health of lake ecosystem has socially and academically become one of the hot issues and common concerns (Kane et al., 2009; Xu et al., 2001; Xu et al., 2011; Zhang et al., 2010). A concept of ecosystem health has been derived by an analogy with human health. Both humans and ecosystems are complex systems composed of interacting parts in a complex balance of interdependent functions (Costanza, 1992). If we observe that an ecosystem is not healthy, we want to know a diagnosis, causes of illness and options for a treatment. A definition of ecosystem health is linked to the system's diversity or complexity. The idea is that diversity or complexity are predictors of stability or resilience and that these are measures of health. A healthy ecosystem is defined as being stable and sustainable, maintaining its organization and autonomy over time and its resilience to stress. Ecosystem health is closely linked to the idea of sustainability, which is seen to be a comprehensive, multiscale, dynamic measure of system resilience, organization, and vigor (Norton et al., 1992).

Ecologists (Norton et al., 1992; Rapport et al., 1998) define ecosystem health in terms of system organization (diversity, structure, interactions between system components), resilience (systems' capacity to maintain structure and function in the presence of stress) and vigor (e.g., activity, metabolism, primary productivity), as well as the absence of signs of ecosystem distress. Numerous general, problem-specific and ecosystem-specific ecological indicators have been developed to assess ecosystem health. However, there is no simple set of indicators available that can be applied to reliably characterize ecosystem health. Ecosystems of the same type (e.g., lakes or eutrophic lakes) are different and case-specific indicators are needed (Jørgensen et al., 2005).

An effective assessment of ecological health in aquatic ecosystems has become an important issue for researchers, policy-makers and environmentalists globally (Kumar et al., 2015). The evaluation methods include single indicator species method and integrated indices method.

Lake ecosystem health assessment emerged in the late 1980s (Jørgensen et al., 2005). Lake ecosystem health assessment can be considered as a holistic approach which uses both quantitative and qualitative information (Figure 1). Several ecological indicators have been proposed for the lake ecosystem health assessment, such as indicators of single taxonomic group or species (Bista et al., 2015), ecosystem stress indicators (Rapport et al., 1985), eco-exergy (Ex) and structural ecoexergy (Exst) (Jørgensen, 1995a, b). An assessment indicator system comprising water quality, ecological and socio-economic criteria was established (Zhang et al., 2015).

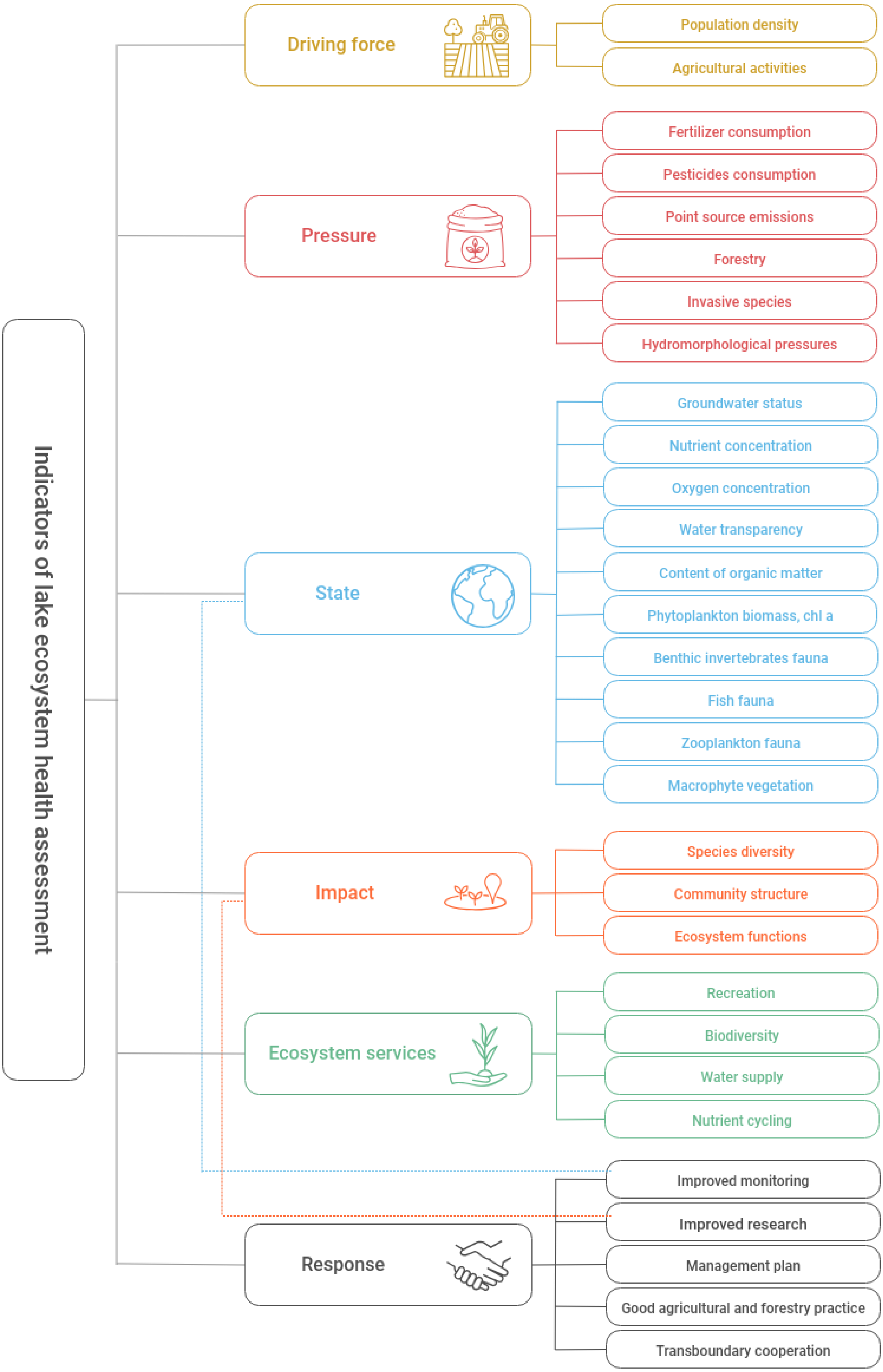

Figure 1. Compartments and indicators of the lake ecosystem health assessment procedure.

1.3. Nexus between groundwater and surface aquatic ecosystems in the WFD

EU Water Framework Directive (WFD, 2000) aims to protect all water resources - inland surface waters, transitional waters, coastal waters, and groundwater. According to the WFD requirements ecological quality of surface waters should be assessed based on biological quality elements (BQE), general physico-chemical parameters and hydromorphological parameters. While for groundwater good status is a combination of both, good quantitative (groundwater levels) and chemical status, thus, a series of conditions defined not only in the WFD, but also in the Groundwater directive (Groundwater directive, 2006) must be met. One of these conditions is to ensure that groundwater inputs to associated surface waters do not result in failure to meet the environmental objectives of those waters or result in significant diminution in status/ecological or chemical quality of those waters (European Commission, 2015).

In the WFD perspective, GAAEs are such surface water bodies (SWBs), including rivers, standing waters and transitional water where the surface water ecology and hydrology strongly depend on groundwater contribution to meet their environmental objectives under the Water Framework Directive (WFD, 2000). The environmental objectives may vary, thus associated environmental quality standards (EQS) or flow/level requirements of GAAE may differ between high status and good status SWBs (European Commission, 2015). WFD distinguishes between groundwater dependent terrestrial ecosystems (GDTEs) such as wetland or spring mires and groundwater associated aquatic ecosystems (GAAEs), for instance, lakes and rivers. Our study addresses lakes.

According to Groundwater directive, member states (including Latvia and Lithuania) shall derive and set threshold values (TVs) for GWBs failing to achieve good status resulting from significant damage to GDE. TVs are groundwater quality standards representing pollutant concentrations, which must not be exceeded to achieve good chemical status for a GWB (Hinsby et al., 2008) Usually, member states have derived or considered to derive TVs for nitrogen and phosphorus compounds and often they are the same as EQS in surface waters. Also, in transboundary water bodies such thresholds should be harmonized between neighboring countries (Retike et al., 2020). Currently, there are no such thresholds set for GDEs in Latvia and Lithuania.

All in all, this methodology aims to protect GAAEs and assess if groundwater is not a subject to anthropogenic alterations that may or has significantly damaged GAAE. The quality status of GWB is assessed based on evaluation of the surface water and ecosystem status and then on estimation of the possible transfer of pollutants and abstraction impacts.

II INDICATORS FOR LAKE ECOSYSTEM HEALTH ASSESSMENT

2.1. Bathymetry

The bathymetry and morphometrics are very important lake characterization indicators and provide necessary information on the status of a water body and its development in relation to its place and changes in the hydrological regime. The morphometric parameters of the lake are influenced by the location, origin, natural conditions, other water bodies and watercourses of the lake catchment area, as well as the anthropogenic impact (Tundisi and Tundisi, 2012). The identification of bathymetry is important for depth monitoring and determination of changes in morphometry (Jawak and Luis, 2015). Determination of lake's depth is important to understand flows of nutrients and water body productivity (Leinerte, 1992).

Bathymetric data can be used for monitoring changes in depth of waterbodies caused by climate changes. Models can be created based on bathymetric data for prediction of tides and currents, as well as hazards like coastal flooding. Scientists use bathymetric data to study the habitats of benthic (bottom-dwelling) organisms. Bathymetric maps can help scientists determine where fish and other marine life feed, live, and breed (NOAA, 2021).

Bathymetric data shall be used for sampling area planning for fish survey, macrophytes, and plankton sampling during the project TRANSWAT.

The bathymetric survey in this study shall be carried out in lake Garais, Galinu, lake Kumpinisku, lake Laucese and lake Skirnu. Aim shall be to survey the lakes and obtain depth measurement data.

The bathymetric study shall be carried out by marking the coastline using the free map server of the Latvian State Forest. The inventory necessary for the research has been prepared. It is necessary to determine an altitude by a certain point followed by levelling accurately at 4 locations on every lake. Based on the data obtained, average altitude shall be calculated. For altitude calculation, Trimble Catalyst GNSS shall be used.

Lakes shall be surveyed using a modern technology motorboat with a 20 hp Honda engine. The boat's echo sounder transducer is placed 15 cm below the water level, therefore, 15 cm shall be added to all data when processing it. The results of the study shall be reflected on a bathymetric map with a resolution of 100 cm. The bathymetric map of this resolution is easier perceived by the user. The lake depth measurement data shall be obtained with the Lowrance HDS Carbon 9 echo sounder. The echo sounder transducer adjustment is a critical point for the accuracy of the results. It is important to adjust the echo sounder correctly to minimize the risk of error and reduce data editing. The recording is performed in normal Primary and DownScan modes. Data recording is done by a boat moving in a circular motion and at steady speed to avoid errors.

After successful fieldwork and first-time data quality control, further data handling requires in-depth quality control of the depth data. It is necessary to make sure that the depth measurements are correctly located on the bottom of the waterbody. The location control can be performed with the ReefMaster sonar viewer. The quality control at this stage consists of deleting the location controls and error points. After converting the data, it is necessary to set up shp file geometry, coordinate system, and projection. This action is performed in QGIS. The research shall be carried out in the Latvian geodetic coordinate system LKS-92. After the successful construction of the isobaths using QGIS and ReefMaster tools, a depth region *shp file is constructed for high-quality visualization of the data.

2.2. Biological indicators

2.2.1. Phytoplankton

In Latvia, an adapted Estonian lake phytoplankton method is used to assess the ecological quality of lakes. The class boundaries of parameters, except Chlorophyll-a are like the original Estonian method, though the national ecological quality ratios (EQR's) are adjusted to Latvian conditions. The Latvian phytoplankton method consists of four parameters (Phillips et al., 2015):

- **Chlorophyll-a** (Chl-a, µg/l, all samples of the lake year).
- **Pielou evenness J.** Values range between 0-1. 1 is the theoretical maximum and therefore also the reference value in all lake types. The scale for each lake type is evenly divided into five classes. Index based on the hypothesis that species diversity distributed evenly in climax societies. Modified Pielou index used to calculate the Diversity index (H). Another part of the equation is the theoretical diversity (H_{max}). The higher the resulting index value, the better the quality of the ecological environment.

$$J = H/H_{max} \qquad (1)$$

- **Nygaard modified compound Quotient (PCQ)** used to determine the ecological status of the lake using the biomass of major groups. Ott and Laugaste (1996) has added two additional elements to the original formula: Cryptophyta and Chrysophyceae. Modified PCQ calculation (by Ott and Laugaste, 1996):

$$PCQ = \frac{Cyanophyta^{*}+Chlorococcales^{*}+Centrales^{*}+Euglenophyceae^{*}+Cryptophyta^{*}+1}{Desmidiales^{*}+Chrysophyceae^{*}+1} \qquad (2)$$

- **Description of a community (PCD)**, which consists of four possible categories:
  a. High and good ecological quality class - species abundance and biomass are very similar for different groups of species. It is difficult to distinguish between dominant taxa.
  b. Dominant species forming 60-80% of total biomass.
  c. Average ecological quality - dominated by 3-5 species (>80% of total biomass).
  d. Poor ecological quality - domination of a single species (>80%).

e. Bad ecological quality class - large biomass is dominated by species belonging to the genera Microcystis, Aphanizomenon, Radiocystis, Planktothrix, Limnothrix, Woronichinia, Anabaena or Chlorococcae. Chlorophyll-a content in a lake is more than 20 mg/m^3. The class boundaries of the above-mentioned parameters are given in Table 1.

Table 1. Summary of phytoplankton metric boundary values (Phillips et al., 2015).

Metric	National types	Ref.	High	Good	Moderate	Poor	Bad
Chl-a (μgl^{-1})	1 & 2	6.2	<9.9	9.9 - 21	21 - 42	42 - 84	>84
	5, 6, 9	3.2	<5.8	5.8 – 1.1	10 - 20	20 - 40	>40
Nygaard Quotient (PCQ)	1 & 2	2	<3.5	3.5-6.0	6.01 – 9.0	>9.0	>9.0
	5, 6, 9	2.5	<4.0	4.0 – 6.5	6.51 – 10.0	>10.0	>10.0
Pielou eveness (J)	1, 2, 5, 6, 9	1.0	0.81 – 0.99	0.61 – 0.80	0.41 – 0.6	0.21 – 0.40	<0.2
Community description (PCD)	1, 2, 5, 6, 9	A	A	B	C	D	E

The final score is determined using the principle of equal weight of each of the above parameters used for the calculation of the final lake phytoplankton. Each phytoplankton parameter value is scored according to the quality class: high – 5; good – 4; moderate – 3; poor – 2; bad – 1. The final lake phytoplankton score is calculated by determining the arithmetic average of each parameter score. Final score: high: 4.01 – 5.0; good: 3.01 – 4.0; moderate: 2.01 - 3.0; poor: 1.01 - 2; bad: ≤1.0.

To get EQR numerical values of the four metrics are summed up and divided by 20 (maximum sum of four parameters which corresponds to reference conditions), which gives joint EQR of all four metrics. Where I_x is a value of each parameter in a 5-point scale.

$$EQR(phytopl.) = I_{Chla} + I_{PCQ} + I_{PCD} + I_J/20 \qquad (3)$$

The sampling should be accomplished according to Table 2 (Phillips et al., 2015).

Current method shows a good pressure – response relationship with eutrophication pressure (total phosphorus) in Latvian lakes. In addition, it is correlated with the total pressure index (LCI) = estimated pressure from households + estimated pressure from land use + estimated pressure from cattle breeding + estimated pressure from secondary pollution (Phillips et al., 2014).

Table 2. Latvian approach to phytoplankton monitoring (Phillips et al., 2015).

Item	Description
Frequency per year	2-4 samples per vegetation season (May, July-September).
Sampling	ISO 10260:1992 for chlorophyll a (spectrophotometry). SM 10200: 2012 for phytoplankton, Utermöhl's technique; counting, using inverted light microscope.
Sampling method	Ruthner type water sampler, samples at 0.5 m deep in the middle of a lake, fixed by Lugole solution.
Level of identification	Species level if possible, but large taxa (class, order) are also used as indicators.

In **Lithuania**, German PSI (Phyto-See-Index) method (hereafter PSI) is used for assessment of ecological status of lakes of respective types. In the national legislation it is referred as "Ežero fitoplanktono indeksas" (EFPI) (TAR, 2016-08-09, Nr. 21814). The method significantly correlates with physical-chemical variables in Lithuanian lakes as well as with status assessment results according to benthic invertebrates and fish-based methods.

PSI index is calculated following original methodology (Mischke et al., 2008). The PSI consists of three metrics: "biomass", "algal classes" and the "Phytoplankton-Taxa-See-Index" (PTSI). Some of these metrics are multi-parameter variables.

1. Biomass metric is composed of:
    a. The total biovolume of phytoplankton in the epilimnic or euphotic zone of the lake (arithmetic mean in the vegetation period from May to September).
    b. Chlorophyll-a concentration (arithmetic mean in the vegetation period from May to September.
    c. Maximum Chlorophyll-a value if it deviates from the mean more than 25%.
2. Algal class metric: the biovolume or its percentage of total biovolume in specific annual periods (e.g., mean values of cyanophytes, dinophytes and of chlorophytes from July to October; mean value from chrysophytes from May to September).
3. PTSI (Phytoplankton Taxa Lake Index): this index evaluates the species composition based on lake-type specific lists of indicator species and their special trophic scores and weighting factors. The method works in two steps:
    a. Trophic assignment results in a PTSI index per sample or lake year.
    b. Assessment by comparing current trophic state with the lake type specific trophic reference status.

For use of PSI the original lake types are subdivided into sub-types based on VQ metric (Table 3). Summary of "Biomass" and "Algal class" metrics values per status class is presented in Tables 4 and 5.

Table 3. National lake types and sub-types for status assessment based on PSI.

Lake types	Lake sub-types	Metrics for division to sub-types*
2-3 (S-DS)	SDS 1	VQ < 1.5
2-3 (S-DS)	SDS 2	VQ > 1.5
1 (P)	P 1	VQ < 1.5
1 (P)	P 2	VQ > 1.5; mean depth < 3m; water residual time >30 days
1 (P)	P 3	VQ > 1.5; mean depth ≥ 3m; water residual time >30 days
1 (P)	P 4	VQ > 1.5; water residual time ≤30 days

* VQ=V*100/Q, where V – catchment area (km^3), Q – lake volume (thousand m^3).

Table 4. Boundary values of "Biomass" metric components.

Status	**Lake sub-types**				
class*	SDS1	SDS2	P1	P2	P3-P4
	Total biovolume of phytoplankton (mm^3 l^{-1}l)				
H/G	1.4	0.7	2.09	4.3	2.95
G/M	3.3	1.7	4.4	9.0	6.0
M/P	7.7	3.8	9.1	18.5	12.2
P/B	18.1	8.0	19.0	39.0	25.1
	Chl-a (µg l^{-1}) mean				
H/G	6.9	4.8	7.2	11.9	9.7
G/M	12.0	8.6	13.2	24.8	17.8
M/P	21.0	15.3	24.3	51.2	32.9
P/B	36.5	27.3	44.8	106.5	61.0
	Chl-a (µg l^{-1}) max				
H/G	15	9	12	22	17
G/M	25	16	24	41	33
M/P	42	28	45	78	63
P/B	70	50	87	145	120

* H – high, G – good, M – moderate, P-poor, B – bad.

Table 5. Boundary values of "Algal class metric" metric components.

Status	**Chrysophyceae** (May-September)	**Chlorophyceae** (July-September)	**Dinophyceae** (July-September)	**Cyanobacteria** (July-September)	**Dinophyceae + Cyanobacteria** (July-September)
class*	Lake sub-types SDS1 and SDS2				
H/G	2.5	0.11			0.9
G/M	1.2	0.20			2.0
M/P	0.6	0.38			4.4
P/B	0.3	0.72			10
	Lake sub-type P1				
H/G		----			1.1
G/M		----			2.29
M/P		----			4.75
P/B		>1			9.9
	Lake sub-type P2				
H/G		0.15	10	1.5	
G/M		0.4	5	3.5	
M/P		1.12	2.5	8	
P/B		3	1.25	19	
	Lake sub-type P3				
H/G				1.5	
G/M				3.0	
M/P				6.0	
P/B				12.0	
	Lake sub-type P4				
H/G		----			1.94
G/M		----			3.91
M/P		----			7.9
P/B		>1			16

* H – high, G – good, M – moderate, P-poor, B – bad.

$$PTSI = \frac{\sum(abundance\ category_i + TAW_i + stenoecy\ factor_i)}{\sum(abundance\ category_i + stenoecy\ factor_i)} \quad (4)$$

Where:
Abundance category – abundance category of the indicator taxon with the index i
TAW_i – Trophic score of the indicator taxon with the index i
Stenoecy $factor_i$ – Stenoecy factor of the indicator taxon with the index i.

Trophic score and Stenoecy factor are according to Mischke et al. (2008).

PSI is the average of the scores of all metrics. The final score is summarized using weighting factors of used components before averaging the metric results (details in Mischke et al., 2008). It is then transformed to a normalized EQR according to formula y = -0.2x + 1.1. The EQR values of PSI per status class are presented in Table 6.

Table 6. National class boundaries for PSI EQR (TAR, 2016-08-09, Nr. 21814).

Class boundary	High/good	Good/moderate	Moderate/poor	Poor/bad
EMI EQR	0.81	0.61	0.41	0.21

A general description of phytoplankton monitoring in LT is given in Table 7 (Valstybės žinios, 2004-04-10, Nr. 53-1827).

Table 7. Lithuanian approach to phytoplankton monitoring.

Item	Description
Frequency per year	4 samples per vegetation season (May, July-September).
Sampling	ISO 10260:1992 for chlorophyll a (spectrophotometry). LST EN 25667-2:2001 for phytoplankton; Utermöhl's technique; counting, using inverted light microscope.
Sampling method	Ruthner type water sampler; samples at 0.5 m deep in the middle of a lake, fixed by Lugole solution.
Level of identification	Species level whenever possible.

2.2.2. Macrophytes

Macrophytes are an important component of aquatic ecosystems and can be used to facilitate the monitoring of ecological status. In addition to their important ecological role, the use of macrophytes as indicators of ecological quality in standing waters is since certain species and species groups are indicators for specific standing water types and are adversely affected by anthropogenic impact. Declining of submerged vegetation is one of the symptoms of a eutrophication of lakes. Macrophyte structure and abundance in lakes several depend on different factors – trophic state, depth of light penetration, and water movements being the most important (Water quality…, 2007).

Latvian macrophyte assessment method (Daugavas upju baseinu..., 2015) for lakes is primarily based on dominating indicator taxa, adding two more parameters: species composition and depth limit of submerged plants.

Passing the littoral of the whole lake by boat, relative abundance of the macrophyte species of all belts (emergent, floating-leaved etc.) and all taxonomic groups are estimated for the lake in the 7-point scale. Using the plant hook with marked rope (or stock), the zonation and depth limits of macrophytes are determined on transects. The frequency of transects depends on the character of the lake; they have been made after 100-500 m.

Different parameters use for each lake ecological type, e.g., for lake very shallow hard water oligohumic lakes (Type 1) characteristic taxa for high.good ecological status are charophytes and *Potamogeton* sp.; indicator species are *Chara* sp. and *Nitella* sp.; total macrophyte species number is >15; charophyte species number is 4-5; filamentous green algae species abundance is 1-2 (Table 8).

Table 8. Latvian macrophyte method

	Quality class				
Type	**High**	**Good**	**Moderate**	**Poor**	**Bad**
Type 1: Very shallow hard water oligohumic lake					
EQR tot.	1	0.8	0.6	0.4	0.2
Characteristic taxa*	Char, Pot	Char, Pot	Nup, Pot	Cer, Lem, Nup	Cer, Lem, Nup
Indicator species for H/G quality	*Chara* sp., *Nitella* sp.	*Chara* sp., *Nitella* sp.			
Macrophyte species number	>15	>15	10-15	<10	<10
Abundance** of charophytes	6~7	4~5	2~3	1	0
Abundance of free-floating species	<2	2~3	4	5	6~7

	Quality class				
Type	**High**	**Good**	**Moderate**	**Poor**	**Bad**
Abundance of filamentous green algae	0	1~2	3~5	5	6~7
Type 2: Very shallow hard water polyhumic lake					
EQR tot.	1	0.8	0.6	0.4	0.2
Characteristic taxa*	Char, Pot	Char, Pot	Nup, Pot	Cer, Lem, Nup	Cer, Lem, Nup
Indicator species for H/G quality	*Chara* sp., *Nitella* sp.				
Abundance of charophytes	>4	3~4	1~2	0	0
Abundance of free-floating species	<2	2~3	4	5	6~7
Abundance of filamentous green algae	0	1~2	3~4	5	6~7
Type 3: Very shallow soft water oligohumic lake					
EQR tot.	1	0.8	0.6	0.4	0.2
Characteristic taxa*	Iso, Char, Bry	Iso, Char, Bry	El, Pot, Char		
Indicator species for H/G quality	*Isoëtes* sp., *Lobelia dortmanna*	*Isoëtes* sp., *Lobelia dortmanna*			
Abundance of helophytes	<2	2~3	4	5	6~7
Abundance of isoetids	7	5~6	3~4	0	0
Abundance of elodeids	<2	2~3	4	5	6~7
Abundance of nympheids	<2	2~3	4	5	6~7
Type 4: Very shallow soft water polyhumic lake					
EQR tot.	1	0.8	0.6	0.4	0.2
Characteristic taxa*	Iso, Char, Bry	Iso, Char, Bry	El, Pot, Char	Cer, Lem, Nup	Cer, Lem, Nup
Indicator species for H/G quality	*Sphagnum* sp., *Utricularia* sp., *Nuphar lutea*	*Sphagnum* sp., *Utricularia* sp., *Nuphar lutea*			
Abundance of helophytes	<2	2~3	4	5	6~7
Abundance of isoetids and charophytes	2~4	2~4	1	0	0
Abundance of elodeids	<2	2~3	4	5	6~7

	Quality class				
Type	**High**	**Good**	**Moderate**	**Poor**	**Bad**
Abundance of nympheids	<2	2~3	4	5	6~7
Type 5: Shallow hard water oligohumic lake					
EQR tot.	1	0.8	0.6	0.4	0.2
Characteristic taxa*	Char, Pot	Char, Pot	Nup, Pot	Cer, Lem, Nup	Cer, Lem, Nup
Indicator species for H/G quality	*Chara* sp., *Nitella* sp.	*Chara* sp., *Nitella* sp.			
Abundance of charophytes	>5	4~5	2~3	1	0
Abundance of free-floating species	<2	2~3	4	5	6~7
Abundance of filamentous green algae	0	1~2	3~4	5	6~7
Colonization depth (m) of submerged macrophytes	>3	2.5~3	1.5~2.5	1~1.5	<1
Type 6: Shallow hard water polyhumic lake					
EQR tot.	1	0.8	0.6	0.4	0.2
Characteristic taxa*	Pot	Pot	Nup, Pot	Cer, Lem, Nup	Cer, Lem, Nup
Indicator species for H/G quality	*Chara* sp., *Myriophyllum alterniflorum*	*Chara* sp., *Myriophyllum alterniflorum*			
Abundance of helophytes	<2	2~3	4	5	6~7
Abundance of free-floating species	<2	2~3	4	5	6~7
Abundance of filamentous green algae	<1	1~2	3~4	5	6~7
Colonization depth (m) of submerged macrophytes	>2	1.5~2	1-1.5	0.5-1	<0.5
Type 7: Shallow soft water oligohumic lake					
EQR tot.	1	0.8	0.6	0.4	0.2
Characteristic taxa*	Iso, Char, Bry	Iso, Char, Bry	El, Pot, Char		
Indicator species for H/G quality	*Isoëtes* sp., *Lobelia dortmanna*	*Isoëtes* sp., *Lobelia dortmanna*			
Abundance of helophytes	<2	2~3	4	5	6~7

	Quality class				
Type	**High**	**Good**	**Moderate**	**Poor**	**Bad**
Abundance of isoetids	>6	5~6	1~4	0	0
Abundance of elodeids	<2	2~3	4	5	6~7
Abundance of nympheids	<2	2~3	4	5	6~7
Colonization depth (m) of submerged macrophytes	>3	2.5~3	1.5~2.5	1~1.5	<1
Type 8: Shallow soft water polyhumic lake					
EQR tot.	1	0.8	0.6	0.4	0.2
Characteristic taxa*	Iso, Bry	Iso, Bry	Nup		
Indicator species for H/G quality	*Isoëtes* sp., *Lobelia dortmanna*	*Isoëtes* sp., *Lobelia dortmanna*			
Abundance of helophytes	>2	2~3	4	5	6~7
Abundance of isoetids	>2	2~4	1	0	0
Abundance of elodeids	>2	2~3	4	5	6~7
Abundance of nympheids	>2	2~3	4	5	6~7
Type 9: Deep hard water oligohumic lake					
EQR tot.	1	0.8	0.6	0.4	0.2
Characteristic taxa*	Char, Pot	Char, Pot	Nup, Pot	Cer, Lem, Nup	Cer, Lem, Nup
Indicator species for H/G quality	*Chara* sp., *Nitella* sp.				
Abundance of charophytes	6~7	4~5	2~3	1	0
Abundance of free-floating species	<2	2~3	4	5	6~7
Abundance of filamentous green algae	0	1~2	3~4	5	6~7
Colonization depth (m) of submerged macrophytes	>3	2.5~3	1.5~2.5	1~1.5	<1

*Char – charophytes, Bry – bryophytes, Pot – *Potamogeton* sp., Cer – *Ceratophyllum* sp., Nup – *Nuphar* sp., Lem – *Lemna* sp., *Spirodela polyrhiza*, Iso – *Isoëtes* sp., *Lobelia dortmanna*, El – *Elodea canadensis.*

**The abundance is estimated using 7-point scale, where 1 – very rare (<1%), 2 – rare (1-3%), 3 – quite rare (2-10%), 4 – frequent (10-25%), 5 – common (25-50%), 6 – abundant (50-75%), 7 - very abundant (75-100%).

The bidirectional supply of light and nutrient resources not only affects the biomass development of planktonic and periphytic algae but may also be involved in determining the distribution of all primary producers. Submerged and floating-leaved macrophytes absorb nutrients both from the water and the sediment, which means that their supply of nutrients will seldom be growth limiting. Emergent macrophytes absorb nutrients from the sediment and photosynthesize above the water, a strategy that makes them competitively superior to all other primary producers with respect to acquisition of nutrients and light. Submerged macrophytes and phytoplankton absorb light in the water column, but the macrophytes take up nutrients from the sediment and thus will dominate over phytoplankton in less productive lakes. Different growth forms of aquatic macrophytes are segregated along depth gradients, which can be seen in most lakes. Closest to the shore, emergent macrophytes dominate, further out floating-leaved and then submerged macrophytes are most abundant (Brönmark and Hansson, 2010).

In **Lithuania**, the modified German Reference Index is used for macrophyte-based assessment of ecological status of lakes (Central Baltic Lake... 2014; Valstybės žinios 2013). In the national legislation it is referred as “Makrofitų etaloninis indeksas” (MEI) (TAR, 2016-08-09, Nr. 21814).

For index calculation, Macrophytes are sampled 1 time per year per water body from July to August. The sampling sites are selected according to expert knowledge, random and stratified sampling, covering all available habitats per water body. The minimal number of transects is determined according to the lake area size-class (Keskitalo and Salonen, 1993). The sampling is made in transects perpendicular to shoreline. Transects are divided into 0–1 m, 1–2 m, 2–4 m and >4 m depth zones. Maximum depth of growth (vegetation limit) is also recorded. At least three samples of macrophytes are taken from each depth zone. The tools used are grapnel and aquascope. Indicatory species belong to these ecological groups: lemnids (freely floating), floating and submerged macrophytes, but the abundance of emerged macrophytes species is also evaluated. Mosses and macrophytes from Charophyta and Angiospermae (Magnoliophyta) divisions are described in species or genus level, filamentous algae – in group level. The minimal size of organisms sampled and processed is 2-3 mm. The abundance of species/groups is estimated according to a 5-degree scale: 1 = very rare, 2 = rare, 3 = common, 4 = frequent and 5 = very frequent.

For calculation of index, abundance of indicator species that are listed in the Lithuanian list of indicator species (A – sensitive, C– insensitive and B – indifferent taxa; Table 9) is assessed. The abundance of plants in each group of indicator species is obtained by summing the abundance class of plants in that group of species calculated for each depth zone. To convert abundance of species to quantity (Q) of species, the estimated abundance is raised by power of 3 (abundance is being cubed). Then index is calculated according to the following equation:

$$MEI = \frac{\sum_{i=1}^{n_A} Q_{Ai} - \sum_{i=1}^{n_C} Q_{Ci}}{\sum_{i=1}^{n_g} Q_{gi}} * 100 \qquad (5)$$

Where:
MEI – Reference Index
Q_{Ai} – Quantity of the i-th taxon of species group A
Q_{Ci} – Quantity of the i-th taxon of species group C
Q_{gi} – Quantity of the i-th taxon of all groups
n_A – Total number of taxa in group A
n_C – Total number of taxa in group C
n_g –Total number of taxa in all groups Quantity= abundance.

The index is calculated for each transect. Depending on the situation, MEI correcting factors are also being applied (Table 10). The necessary conditions for MEI calculation for different lake types are described in Table 11. If these conditions are not fulfilled, the index cannot be calculated. MEI values are transformed into EQR values according to the formula: EQR = (LRI+100)*0.5/100. Index values per status class are presented in Table 12.

Table 9. List of indicator species for MEI calculation.

Species	Indicator species groups	
	Lakes average depth >3 m	Lakes average depth >3 m
Alisma gramineum	B	–
Batrachium circinatum	C	B
Butomus umbellatus	B	B
Callitriche hermaphroditica	B	B
Ceratophyllum demersum	B	B
Ceratophyllum submersum	B	–
Chara aspera	A	A
Chara contraria	B	A
Chara virgata	B	A
Chara filiformis	A	A
Chara globularis	B	A
Chara hispida	–	A
Chara intermedia	A	A
Chara rudis	A	A
Chara strigosa	A	A
Chara tomentosa	A	A

Species	Indicator species groups	
	Lakes average depth >3 m	Lakes average depth >3 m
Drepanocladus aduncus	B	B
Drepanocladus sendtneri	B	B
Eleocharis acicularis	B	B
Elodea canadensis	C	C
Fontinalis antipyretica	B	B
Hippuris vulgaris	B	B
Hydrilla verticillata	B	A
Hydrocharis morsus-ranae	C	B
Lemna minor	C	B
Lemna trisulca	C	B
Myriophyllum sibiricum	+	+
Myriophyllum spicatum	B	B
Myriophyllum verticillatum	B	B
Najas intermedia	B	A
Najas marina	C	C
Nitella flexilis	B	A
Nitella mucronata	B	A
Nitella opaca	A	A
Nitellopsis obtusa	B	B
Nymphaea alba	B	B
Nymphaea candida	B	B
Nuphar lutea	B	B
Persicaria amphibia	B	B
Potamogeton × nitens	B	A
Potamogeton × salicifolius	B	A
Potamogeton angustifolius	A	–
Potamogeton acutifolius	B	A
Potamogeton alpinus	A	A
Potamogeton berchtoldii	B	B
Potamogeton compressus	B	A
Potamogeton crispus	C	B

Species	Indicator species groups	
	Lakes average depth >3 m	Lakes average depth >3 m
Potamogeton filiformis	A	A
Potamogeton friesii	B	B
Potamogeton gramineus	A	A
Potamogeton lucens	B	A
Potamogeton natans	C	B
Potamogeton pectinatus	B	B
Potamogeton perfoliatus	B	B
Potamogeton praelongus	A	A
Potamogeton pusillus	B	B
Potamogeton rutilus	A	A
Ranunculus reptans	+	+
Rhynchostegium riparioides	B	B
Sagittaria sagittifolia	C	B
Scorpidium scorpioides	B	B
Sparganium emersum	C	B
Spirodela polyrhiza	C	B
Stratiotes aloides	B	A
Utricularia minor	–	+
Utricularia vulgaris	B	A
Zannichellia palustris	C	B

Table 10. MEI correcting factors.

Lake average depth	Correcting factors
>3 m	– if LRI > 0 and vegetation limit <5 m, MEI is reduced by 50; – if dominant stands of one of the following taxa occur, MEI is redused by 50: *Ceratophylum demersum, C. submersum, Elodea canadensis, Najas marina, Potamogeton pectinatus.*
<3 m	– if LRI > 0, maximum depth ≥3 m and vegetation limit <3 m, MEI is reduced by 50; – if dominant stands of one of the following taxa occur, MEI is redused by 50: *Ceratophylum demersum, C. submersum, Elodea canadensis, Najas marina, Potamogeton pectinatus.*

Table 11. The necessary conditions for MEI calculation.

Lake average depth	Necessary conditions
>3 m	– total plant quantity (abundance[3]) is ≥55; – species belonging to genus Nymphaea and Nuphar make less than 80% of total plant quantity.
<3 m	– total plant quantity (abundance[3]) is ≥35; – species belonging to genus Nymphaea and Nuphar make less than 80% of total plant quantity; species for which indicator value is not determined make no more than 25% of total plant quantity.

Table 12. National class boundaries for MEI EQR.

Class boundary	High/good	Good/moderate	Moderate/poor	Poor/bad
EMI EQR	0.76	0.50	0.25	0.01

2.2.3. Macrozoobenthos

Benthic invertebrates play an essential role in lake ecosystem key processes, like food chain dynamics, productivity, nutrient cycling and decomposition. In the food webs of lakes, they have an intermediate position between primary producers and destruents on the one side, and higher trophic levels (as fish) on the other side (Solimini et al., 2006). However, several studies have shown weak or no pressure-response relationships for benthic invertebrates in lakes, especially for littoral invertebrates and eutrophication pressure (Poikane et al., 2016). At the lake ecosystem level, benthic invertebrates can be an additional biological quality element for the assessment of lake ecosystem health, compared to the phytoplankton and macrophyte communities, which are considered better indicators.

The main pressures, which affect the lake integrity are eutrophication, acidification, alterations of hydrology and geomorphology (Young et al., 2005). Assessing the lake ecosystem health using benthic invertebrate structural and functional parameters, we must consider that the littoral, sub-littoral and profundal invertebrate communities are driven by different governing factors, which therefore probably indicate different human disturbances (Solimini et al., 2006). In the EU GIG intercalibration groups the use of littoral benthic invertebrate communities for the assessment of ecological quality was accepted (Böhmer et al., 2014). Using currently developed Latvian Lake Macroinvertebrate Multimetric Index (LLMMI) (Skuja and Ozoliņš, 2017), we can assess the lake littoral community structural characteristics and ecological status.

For the lake ecosystem health assessment using benthic invertebrates by Latvian researchers will be used:

1. Ecological state assessment using LLMMI (Skuja and Ozoliņš, 2017).
2. Structural and functional characterisation of littoral benthic invertebrate communities: analysis of community structural characteristics for different ecological quality classes (e.g., according to Skuja and Ozoliņš (2017)); species trait analysis (e.g., functional feeding groups, microhabitat and locomotion type preference). Especially endangered and protected species occurrence will be analysed, and the main anthropogenic factors assessed.
3. Assessment of benthic invertebrate alien species impact on lake ecosystems (Arbačiauskas et al., 2008).

Sampling. For the sampling, a 50 m long, representative lake littoral zone stretch is chosen. Habitat types are determined according to bottom substrate types. Sampling is done in proportion to coverage of dominant habitat types. If the water depth does not exceed 1.5 m in littoral zones of lakes and reservoirs, sampling is done using a hand-net. 10 replicates are taken or if bottom is very rich in detritus (large particles of detritus) only 5 replicates are taken. All replicates are merged in to one sample and analyzed also as one sample (Skuja and Ozoliņš, 2017).

Kick and sweep approach is used: Kick sampling in lake littoral zone: hand-net is vertically placed on the bottom and substrate is mixed with the toes pointing downstream 0.25 m before the frame of the hand-net. Suspended material is collected in the net. Sweeping technique: if the lake is deeper than > 1 m and the littoral zone is steep, samples are taken by sweeping the handnet vertically from bottom along the shoreline vegetation upwards (especially in dystrophic peatbog lakes) (Skuja and Ozoliņš, 2017).

Additional qualitative samples. Sampling by hands from stones and macrophytes in shallow water bodies: hand-net is vertically placed on the bottom and stones and macrophytes are turned over by hands in the front of hand-net. All organisms are also picked up from the stones and macrophytes. Also, the finer substrate is mixed by hand (Skuja and Ozoliņš, 2017).

Sample processing. Sample is washed in the net of a hand-net, till there are no fine particles of the bottom substrate. If the sample contains a lot of coarse sand, the organic material is suspended and separated from mineral particles by shaking the bucket. After careful washing, a sample is put into the bottle and preserved in 96% ethanol. Before sorting, the sample is washed in water using a sieve with mesh size of 0.5 mm to rinse the preservative (ethanol). Sample is sorted in the same day (Skuja, Ozoliņš, 2017).

During sorting, specimens are separated according to taxonomical groups: Bivalvia, Coleoptera, Diptera, Ephemeroptera, Gastropoda, Heteroptera, Hirudinea, Hydrachnidia, Lepidoptera, Crustacea, Megaloptera, Neuroptera, Nematoda, Odonata, Oligochaeta, Plecoptera, Trichoptera. Dominant groups are separated additionally. If only some individuals are found for particular groups, they could be placed in the same bottle (e.g., Bivalvia + Gastropoda, Coleoptera + Heteroptera, Ephemeroptera + Plecoptera, Hirudinea + Hydrachnidia etc.).

Number of individuals is determined separately for each taxa. Biomass is estimated using analytical balance (precision of 0.001). Before weighing, specimens are placed on filter paper to remove left moisture (water or preservative). Caddisfly larvae are removed from the cases before weighing.

Identification level. Benthic macroinvertebrates are identified to the best achievable taxonomic level (preferably species level; or genus, family level): Trichoptera, Ephemeroptera, Plecoptera, Gastropoda, Bivalvia (except genus Pisidium), Odonata, Coleoptera, Heteroptera, Hirudinea, Megaloptera, Turbellaria.

Oligochaeta, Chironomidae, Simuliidae, other Diptera families, Hydrachnidia, *Pisidium* sp., Nematoda are not identified further.

Data processing. Before the calculation of metrics, a taxonomical adjustment is recommended to apply to avoid the overlapping of taxa, especially for EPT taxa.

Calculation of the metrics. Number of taxa, Number of EPTCBO taxa, ASPT (Average Score per Taxon) index, Acidity index and Shannon – Wiener diversity index is calculated using ASTERICS 4.04 software.

2.2.3.1. Latvian Lake Macroinvertebrate Multimetric Index

LLMMI has been developed on the base of Estonian multimetric index (Birk et al., 2010), comprising five metrics: Number of taxa, Number of EPTCBO taxa, ASPT index, Acidity index and Shannon-Wiener diversity index, thus indicating taxonomic composition, abundance, ratio of disturbance of sensitive taxa to tolerant taxa and diversity (Table 13) (Skuja and Ozoliņš, 2017).

Table 13. Overview of the metrics included in the Latvian Lake Macroinvertebrate Multimetric Index (LLMMI).

MS	Taxonomic composition	Abundance	Sensitive / tolerant taxa	Diversity
LV	Number of taxa Number of EPTCBO taxa	Relative abundance (Shannon – Wiener diversity index)	ASPT index (Armitage et al., 1983). Acidity index (Henrikson and Medin, 1986)	Number of taxa Number of EPTCBO taxa Shannon – Wiener diversity index

Calculation of LLMMI (Latvian Lake Macroinvertebrate Multimetric Index). Values of Number of taxa, Number of EPTCBO taxa, Shannon-Wiener diversity index, ASPT and Acidity index are standardized according to Hering et al. (2006) and the LLMMI value are calculated as arithmetic mean of standardized EQR values of all five metrics (Table 14).

$$EQR = \frac{Metric\ result - Lower\ Anchor}{Upper\ Anchor - Lower\ Anchor} \quad (6)$$

Table 14. Upper and lower anchor values for calculation of LLMMI EQR.

Metric	**Upper anchor** (highest observed metric value in all dataset)	**Lower anchor** (lowest observed metric value in all dataset)
ASPT	6.3	3.5
Shannon-Wiener diversity index	3	1.4
Acid index	11	1
Number of EPTCBO taxa	19	2
Number of taxa	30	7

Table 15. National class boundaries for LLMMI EQR.

Class boundary	High/good	Good/moderate	Moderate/poor	Poor/bad
LLMMI	0.80	0.60	0.40	0.20

2.2.3.2. Description of benthic invertebrate communities for three ecological quality classes

Additionally, of assessment of ecological quality, benthic invertebrate communities at each sampling site will be analysed and compared with a previously in Skuja, Ozoliņš (2017) described communities of frequently common and abundant taxonomic groups, characteristic for high, good and moderate ecological status. Especially endangered and protected species occurrence will be analysed.

Description of communities at high status. Characteristic benthic invertebrate communities: Bivalvia – *Anodonta anatina*, *Unio tumidus*, *Unio pictorum*, *Pisidium* spp.; Crustacea – *Gammarus lacustris*; Ephemeroptera – *Cloeon dipterum*, *Ephemera vulgata*, Leptophlebiidae, Heptageniidae; Gastropoda – *Gyraulus albus*, *Radix balthica*, *Radix auricularia*, *Stagnicola* spp., *Valvata piscinalis*, *Viviparus viviparus*; Odonata – *Libellula fulva*, Libellulidae, Aeshnidae, Gomphidae; Trichoptera – *Halesus* spp., *Oecetis* spp., *Athripsodes cinereus*, *Mystacides azurea*, *Cyrnus flavidus*, *Ecnomus tenellus*, Phryganeidae, *Limnephilus flavicornis*, *Limnephilus* spp., Coleoptera – Elmidae (Figure 2) (Skuja and Ozoliņš, 2017).

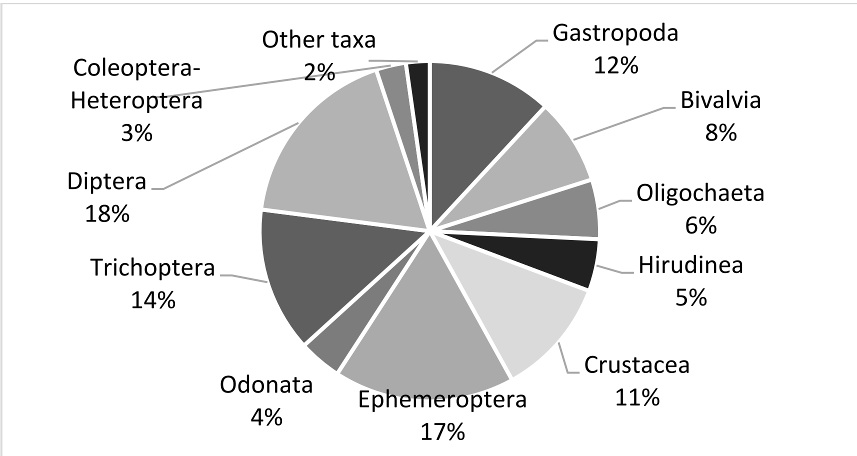

Figure 2. Average percentage of benthic invertebrate taxonomic group abundance for sites at high ecological status (n=14).

Description of communities at good status. Characteristic benthic invertebrate communities: Bivalvia – *Anodonta anatina*, *Unio tumidus*; Crustacea – *Gammarus lacustris*; Ephemeroptera – *Baetis* sp., *Centroptilum luteolum*, *Caenis horaria*, *Caenis luctuosa*; Gastropoda – *Viviparus contectus*, *Physa fontinalis*, *Gyraulus albus*, *Valvata* spp., *Radix auricularia*; Hirudinea – *Alboglossiphonia heteroclita*; Odonata – Libellulidae: *Somatochlora metallica*, *Cordulia aenea*, Gomphidae; Trichoptera –

Athripsodes aterrimus, *Mystacides azurea*, *Limnephilus nigriceps*, *Limnephilus* spp. (Figure 3) (Skuja and Ozoliņš, 2017).

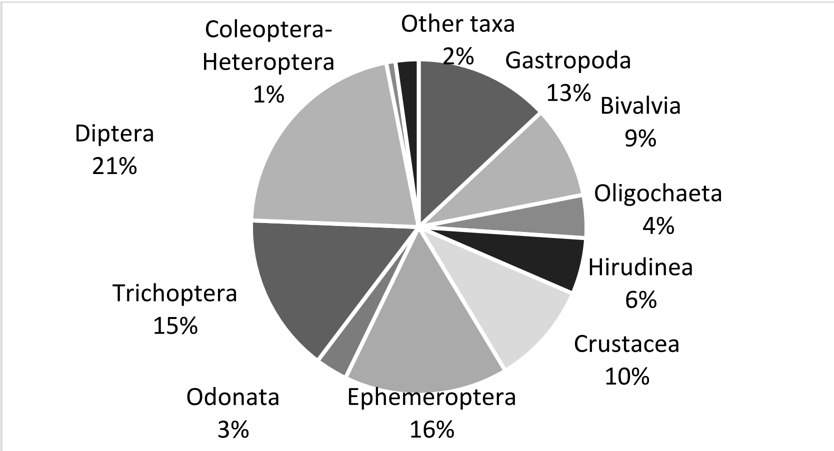

Figure 3. Average percentage of benthic invertebrate taxonomic group abundance for sites at good ecological status (n=33).

Description of communities at moderate status. Characteristic benthic invertebrate communities: Chironomidae (Diptera), Ephemeroptera – *Caenis horaria*, *Cloeon dipterum*; Trichoptera - Limnephilidae, *Limnephilus* spp.; Gastropoda – *Bithynia leachi*, Planorbidae; Crustacea – *Asellus aquaticus*; Odonata: *Erythromma najas*, Coenagrionidae, Hirudinea: *Helobdella stagnalis*, *Erpobdella octoculata*, Megaloptera – *Sialis* spp. (Figure 4) (Skuja and Ozoliņš, 2017).

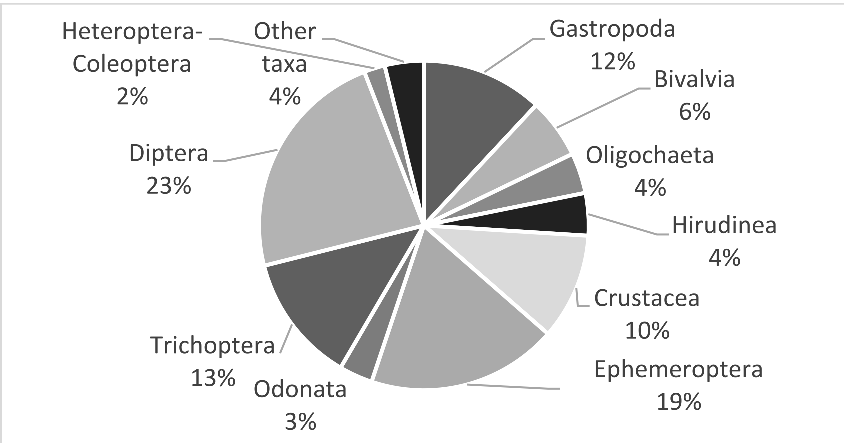

Figure 4. Average percentage of benthic invertebrate taxonomic group abundance for sites at moderate ecological status (n=24).

2.2.3.3. Assessment of ecological pressure of alien species

Additionally, to LLMMI, the ecological pressure of alien species will be assessed, using biocontamination assessment method, developed by Arbačiauskas et al. (2008): Site-specific biocontamination index (SBCI), which is derived from two metrics:

1. abundance contamination index (ACI) and
2. richness contamination index (RCI) at ordinal rank:

ACI = N_a/N_t, where N_a and N_t are numbers of specimens of alien taxa and total specimens in a sample, respectively.

RCI = T_a/T_t, where T_a is the total number of alien orders, and T_t is the total number of identified orders (Arbačiauskas et al., 2008).

SBCI is calculated using matrix (Table 16); five classes of biocontamination ranging from 0 to 4 are defined: 0 (no biocontamination, "high" ecological status, blue cell); 1 (low biocontamination, "good" ecological status, green cell); 2 (moderate biocontamination, "moderate" ecological status, yellow cells); 3 (high biocontamination, "poor" ecological status, orange cells); 4 (severe biocontamination, "bad" ecological status, red cells). Furthermore, these classes of SBCI directly correspond to five ecological quality classes in the Common Implementation strategy for the EU Water Framework Directive (WFD, 2000) (Arbačiauskas et al., 2008).

Table 16. Assessment of site-specific and integrated biocontamination indices (SBCI and IBCI, correspondingly) (Arbačiauskas et al., 2008)

RCI	**ACI**				
	none	**0.01 – 0.10**	**0.11 – 0.20**	**0.21 – 0.50**	**> 0.50**
none	0				
0.01 – 0.10		1	2	3	4
0.11 – 0.20		2	2	3	4
0.21 – 0.50		3	3	3	4
> 0.50		4	4	4	4

2.2.3.4. Lithuanian lake benthic invertebrate multimetric index (EMI)

In Lithuania for lake ecological status assessment using benthic invertebrates is used multimetric index, which is referred as "Ežero makrobestuburių indeksas" (EMI) in the national legislation (TAR, 2016-08-09, Nr. 21814) and is based on 4 metrics (Šidagytė et al., 2013):

1. First Hill's effective taxa number (Hi). It is calculated according to formula:

$$H_1 = e^{-\sum_{i=0}^{TS} p_i \ln p_i} \qquad (7)$$

Where: TS – number of taxa; p – relative abundance of i-th takson (Table 17)

2. Average Score Per Taxon (ASPT) (Armitage et al., 1983; *Pontogammarus robustoides* and *Obesogammarus crassus* attributed to Gammaridae).
3. Number of Coleoptera, Ephemeroptera and Plecoptera taxa (CEP).
4. Percentage of Coleoptera Odonata and Plecoptera individuals in respect of a total number of individuals (COP).

Table 17. The recommended identification level for the first Hill's number and CEP metrics.

Taxa	Identification level
Turbellaria	Species
Oligochaeta	Class
Hirudinea	Species
Mollusca	Species
Crustacea	Species
Plecoptera	Species
Ephemeroptera	Species
Odonata	Species
Heteroptera	Species
Megaloptera	Species
Neuroptera	Species
Coleoptera	Genus
Trichoptera	Species
Lepidoptera	Species
Diptera	Family

Transformation of calculated metrics values to EQR is done according to formula (like the one used in Latvia):

$$EQR = \frac{Metric\ result - Lower\ Anchor}{Upper\ Anchor - Lower\ Anchor} \qquad (8)$$

Values of metrics upper and lower anchors are given in Table 18, national class boundaries for EMI EQR are presented in Table 19.

EMI is an arithmetic mean of standardized EQR values of all metrics:

$$EMI = \frac{H_1 EKS + ASPT\ EKS + CEP\ EKS + COP\ EKS}{4} \qquad (9)$$

If the calculated LEMI value is greater than 1, it is set to 1.

Table 18. Upper and lower anchor values for calculation of EMI EQR.

Metric	Reference value	Lower anchor (lowest observed metric value in all dataset)
H_1	18	0
ASPT	5.8	1
CEP	12	0
COP	0.20	0

Table 19. National class boundaries for EMI EQR.

Class boundary	High/good	Good/moderate	Moderate/poor	Poor/bad
EMI EQR	0.80	0.60	0.40	0.20

Sampling method is a standard method of 12 kick or sweep replicates from different microhabitats. In addition, semi-quantitative sampling procedure is carried out using a standard dip-net (25x25 cm). Sampling can be performed in either of the two core eulittoral mesohabitats: a bottom (preferably hard) kick sample or a vegetation (preferably submerged) sweep sample. Within a stand of either mesohabitat, a stretch of about 15 - 20 meters long are sampled while moving along the shore in a trajectory of a zigzag curve (from the very shoreline to the depth of 1 m) in a way to result in 3 minutes of actual catching time. A semi-quantitative sample is supported by a qualitative (search) sample (duration 1 minute) within the same mesohabitat. Sampling time is from April to November, one occasion per sampling season and per lake (Šidagytė et al., 2013).

2.2.4. Fishes

The status of a fish population can reflect the overall condition of the aquatic environment. Fish population characteristics can be used as indicators of environmental health. This is a simple and not expensive method to assess fish population responses to environmental degradation and climate changes.

The WFD lists the following indicative parameters for the fish fauna: taxonomic composition, abundance, sensitive species, age structure. Fish communities are one of the indicators of ecological quality in freshwaters because fish occupy all trophic levels, and integrate inputs and the effects of pressures across the ecosystem (Carpenter et al., 1985). Despite the fact that the assessment of ecological quality in freshwater ecosystems is a key issue in many countries, assessment methods are country-specific (Blabolli et al., 2017). EQS (Ecological Quality Ratios) values can be calculated for all lakes.

In **Latvia** sampling of fish is performed with three different mesh-sized gillnets ranging from 20 mm to 35 mm knot to knot which are 30 m long (two 15 m long nets which are tied together) and 1.5 m deep. The gillnets are composed of 2 panels which are 15 m long with mesh size 20 mm and 30 mm, 25 mm and 35 mm, 27 mm and 33 mm. In some lakes which are marked as salmonid lakes (Laucesa in the TRANSWAT project) is used additionally 6.0 m hight gillnets 20 mm from knot to knot. Nets are placed in shallow and deeper zones near the coastline and up to 100 m far. Total sampling duration is standard effort of benthic gillnets set for approximately 12 hours including dusk and dawn.

Time period of sampling is June – September after perch, roach, and bream spawning. The catch within each gillnet is registered as total number of individuals and total weight for each species. For sampling strategy Latvia uses a modified version of European standard form EN 14757 and the main difference is that we do not regularly use gillnets with mesh sizes smaller than 20 mm.

Based on gillnet catches, fish assemblages are described. Fish structure was as total abundance expressed as weight (WPUE) per net. Furthermore, the abundance roach and bream percentage by weight of the Cyprinidae (Bream/RoachW%) and Percidae (PerchW%) fish families and the ratio of Cyprinidae to Percidae and roach average weight (g) were calculated.

Based on previous mentioned values, LVFI (Latvian Lake Fish Index) is calculated. LVFI is a method of lake ecosystem health assessment based on exploratory fisheries of commercially significant fish species.

LVFI is a multimetric index which includes four selected indices (Wavg – average weight, W% – percentage of total weight):

1. **WPUE** – weight per unit of effort.
2. **RoachWavg** – roach average weight (g) in a catch using nets with a mesh size of 20-35 mm.
3. **Bream/RoachW%** – roach and bream percentage by weight in a gill net with a mesh size 20-35 mm.
4. **PerchW%** – percentage of perch by weight in gill nets with mesh size of 20-35 mm.

EQR for different metrics are calculated separately:

$$EQR = \frac{\text{Lower boundary value} - \text{calculated index}}{\text{Lower boundary value} - \text{reference value}} \quad (10)$$

EQR of PerchW% and RoachWavg is calculated using the formula:

$$EQR = \frac{\text{Calculated index} - \text{lower boundary value}}{\text{Reference value} - \text{lower boundary value}} \quad (11)$$

Final LVFI EQR is calculated using combination of all four metrics:

$$EQR = \frac{\text{EQRsum} - \text{EQRmin}}{\text{EQRmax} - \text{EQRmin}} \qquad (12)$$

Where:
Lower boundary value is minimum value observed for selected metric.
EQRsum is a sum of individual EQR's of all four parameters for each individual lake.
EQRmin and EQRmax is the minimum and maximum EQR values for all lakes.

LVFI (Latvian Lake Fish Index) characterizes the ecological status of a lake. Totally five classes are described (high, good, moderate, poor, and bad), where value 1 means very high ecological quality and value 0 means very bad ecological quality. The best quality in Latvian lakes LVFI is 0.76, the worst LVFI is 0.17 (Table 20).

Table 20. Latvian national boundaries of quality of lakes.

Ecological status based on LVFI	LVFI
High/Good	0.76
Good/Moderate	0.57
Moderate/Poor	0.40
Poor/Bad	0.17

In **Lithuania**, national Lake Fish Index ("Ežero žuvų indeksas"; EŽI) is adopted for assessing the status of lakes (Virbickas et al. 2016; TAR, 2016-08-09, Nr. 21814).

To assess the status of the lake based on EŽI, fish are sampled with multimesh benthic gillnets, each of which is 40 m in length and 3 m in height. Mesh size vary every 5 meters and are 14, 18, 22, 25, 30, 40, 50, 60 mm. Fishing is carried out in the second half of summer – at the beginning of autumn with water temperature being >15° C. Depending on the lake area, at least 6 (<100 ha lakes), 8 (< 300 ha), 12 (< 600 ha), 16 (<1000 ha) or 20 (>1000 ha) benthic nets are used following the standardized method (TAR, 2018-05-15, Nr. 7783). Nets are positioned randomly to cover different parts and lake depths of each lake. In deep (>17 m maximum depth) lakes, 8–12 m height multimesh benthic gillnets for vendace Coregonus albula and smelt Osmerus eperlanus (14-, 18-, 22- and 26-mm mesh size) are also used as fish catches with standard height benthic gillnets fail to reflect the abundance of these pelagic fishes representatively. Nets are set in lakes for 10-12 hrs during the night covering sunset and sunrise periods.

Depending on the type of lake, from 5 to 6 fish metrics are used to calculate the index (Table 21).

Table 21. Fish metrics and their values per status class.

Type	Metric	Reference	High	Good	Moderate	Poor
1 (P)	*S_bream_W%* [1]	1.5	< 4	<11	<19	<26
	Benth_Sp_W% [2]	10	<20 (>0)	<35	<47	<61 (0)
	Perch_N% [3]	30	>25	>17	>9	>4
	Nb_Oblig_Sp [4]	6	6	5	4	<4
	Non-nat_W% [5] (only when Nb of ind. >1)	0	0	0	<1	<6
2 (S)	*Roach_Q_av* [6]	60	>50	>34	>23	>14
	S_bream_W%	1	<2.5	<9	<17	<26
	Benth_Sp_W%	7	<16 (>0)	<29	<45	<61 (0)
	Perch_Steno_W% [7]	35	>30	>17	>9	>4
	Nb_Oblig_Sp	6	6	5	4	<4
	Non-nat_W% (only when Nb of ind. >1)	0	0	0	<1	<6
3 (DS)	*Roach_Q_av*	60	>50	>34	>23	>14
	Benth_Sp_W%	4	<12 (>0)	<27	<41	<56 (0)
	Perch_Steno_W%	40	>35	>24	>14	>4
	Nb_Oblig_Sp	8	8-7	6-5	4	<4
	Non-nat_W% (only when Nb of ind. >1)	0	0	0	<1	<6

1 – relative biomass of silver bream.
2 – relative biomass of silver bream, bream, and ruff.
3 – relative abundance of perch.
4 – number of obligatory species. POLY lakes - bleak, rudd, pike, tench, perch, roach; S lakes - vendace, bleak, rudd, pike, perch, roach; DS lakes - vendace, smelt, burbot, bleak, rudd, pike, perch, roach.
5 – relative biomass of non-native and translocated species (common carp, gibel carp, silver carp, pikeperch).
6 – mean weight of roach individuals.
7 – relative biomass of perch, burbot, smelt, vendace and whitefish.

To calculate the index, the measured values for each metric are converted to EQR:

1. for metrics *S_bream_W%* and *Benth_Sp_W%* the values are transformed to EQR with formula: EQR = $(X-X_{MAX})/(X_{RC}-X_{MAX})$, where X – measured value, X_{RC} – reference value, X_{MAX} – theoretical maximum value.
   S_bream_W% metric X_{MAX}=30.
   Benth_Sp_W% metric X_{MAX}=70 in POLY and S lakes.
   X_{MAX}=65 in DS lakes.
   If calculated EQR values are <0 or >1, they are clipped at 0 and 1. If X=0 than EQR=0.
2. for metrics *Perch_N%, Perch_Steno_W%* and *Roach_Q_av* the values are transformed to EQR with formula: EQR = X/X_{RC}; if calculated EQR values are >1, they are clipped at 1.
3. for metric *Nb_Oblig_Sp,* the values are transformed to EQR with the formula: EQR = X/XRC
   Before transformation, X=4 values are multiplied by 0.3, and X<4 values are multiplied by 0.15.
4. for metric Non-nat_W%, the EQR values are adjusted as follows: if Non-nat_W% >0 <1%, EQR=0.5; if Non-nat_W% =1-5%, EQR=0.2; if Non-nat_W%>5%, EQR=0.

Total EŽI_EQR for the lake is the mean of the metric EQR values. If non-native and translocated species are not present in the lake, or only one individual has been recorded (occasional occurrence), metric *Non-nat_W%* is not used for calculation of total EQR. Status class boundaries according to EŽI are presented in Table 22.

Table 22. National class boundaries for EŽI EQR.

Class boundary	High/good	Good/moderate	Moderate/poor	Poor/bad
EŽI EQR	0.86	0.61	0.37	0.18

2.2.5. Zooplankton

Zooplankton organisms are not used in the ecological quality assessment scheme according to the WFD, despite they are an important component of the pelagic food web. Zooplankton reflects changes occurring in higher and lower trophic levels being eaten by fish and feeding on phytoplankton. Hence it has a strong indicator value that cannot be covered by existing WFD abiological quality elements (BQE). Numerous studies all around the globe have demonstrated the applicability of zooplankton as integrative and valuable indicators both for ecological quality of lakes (e.g., eutrophication, acidification) and recovery after restoration activities (e.g., nutrient loading reduction, biomanipulation). Its role has been continuously discussed and it is strongly recommended to include zooplankton as a central BQE in the assessment of lake water quality. Simultaneously, regional calibration and harmonisation of assessment methodologies are important to obtain relevant metrics (Duggan et al., 2020; Jeppesen et al., 2011; Josue et al., 2021; Karpowicz et al., 2020; Stamou et al., 2021).

Unfortunately, zooplankton exclusion from WFD, led to overall decrease of zooplankton species monitoring in Europe, including Latvia and Lithuania, consequently, trained experts in species identification are becoming rare and knowledge gaps exist.

This project will serve as a pilot study for evaluating zooplankton as a reliable trophic state (corresponding to total phosphorus values) indicator for lake ecosystem health assessment in Latvia and Lithuania.

There are several indicators based on contemporary zooplankton samples (Jeppesen et al., 2011) what will be used:

- **Crustacean (Copepoda + Cladocera) species richness** versus total phosphorus.
- ***Daphnia* spp., small cladocerans, calanoid copepods, cyclopoid copepods contribution** (%) in the sample versus total phosphorus.
- **Cladocera:Copepoda, Cyclopoida:Calanoida abundance** ratio versus total phosphorus in comparison with Secchi depth and Chlorophyll *a* values.

Besides, we will also verify previously studied Cladocera indicators of trophic state in Latvia (Čeirāns, 2007; Urtane, 1998), look for corresponding indicators representing Copepoda group and evaluate littoral species contribution to trophic state indicator development.

Previously described by Urtane (1998) Cladocera communities:

1. With dominance index decreasing as a response to eutrophication development: *Daphnia cristata, D. longispina, Bythotrephes longimanus, Disparalona rostrata, Ceriodaphnia pulchella, Scapholeberis mucronata, Limnosida frontosa, Bosmina (Eubosmina) coregoni.*
2. With dominance index increasing as a response to eutrophication development: *Bosmina (Bosmina) longirostris, Daphnia cucullata, Chydorus sphaericus.*

Methods for sampling and processing zooplankton samples are adapted following European Standard EN 15110:2006 “Water quality – guidance standard for the sampling of zooplankton from standing waters”.

For ecosystem health assessment several sampling sites in each lake should be chosen to cover different habitats. Data from pelagic and littoral zones will be combined. Both for qualitative and quantitative sampling, a conical plankton net of mesh size 90 microns is used. Optimal sampling frequency is three times per year (in May, July, and September). If sampling is carried out two times per year, then the best time is July and September.

Types of samples:

1. Quantitative sample – pelagic zone - 25 l of upper water layer.

2. Qualitative sample - vertical net haul (whole water column from the deepest part of the lake is sampled from bottom to the surface).
3. Qualitative sample – littoral net haul - habitat 1 – 2 to 6 m through vegetation is sampled.
4. Qualitative sample – littoral net haul - habitat 2– 2 to 6 m over sand, through stones or different vegetation than habitat 1 is sampled.

Samples are stored in 50 ml or larger volume plastic bottles or glass vials, preserved with 96% ethanol (samples type 3&4) or Lugol's Iodine (samples type 1&2).

In general, the entire sample is counted to species level (Cladocera, Copepoda) distinguishing individuals with eggs, males, females and copepodites for Copepoda. If there are more than 400 organisms in total, subsamples of 10 ml are examined until at least 200 organisms from each representing group (Cladocera, Copepoda) are counted. The rest of the sample is examined for absence/presence of rare species. Crustacean identification is following standard taxonomic treatises and taxonomic revisions (Einsle, 1993; Flössner, 1972, 2000; Sars 1903, 1918).

2.2.6. Phytobenthos

In Lithuania, the taxonomic composition and abundance of phytobenthos is also used for assessment of the status of lakes (https://circabc.europa.eu/sd/a/ae8b63f7-5364-4c2e-bd86-df5b7b89f4ef/LT%20-%20Phytobenthos_intercalibration_LT_report_updated.pdf).

Lithuanian lake phytobenthos index (EFBI) is the Trophic index (Rott et al., 1999). For index calculation, phytobenthos sampling is carried out in July - August. Samples are taken once per year, from the hard substrata, submerged in the littoral of lakes (ideally, from the stones). The periphyton covering the stones' surface is scratched off with a scalpel, scraper or similar device and is transferred into a labelled sampling container. If mainly sand or soft sediments are present, the upper millimeters are lifted off with a spoon. Samples of diatoms can also be collected from submerged macrophytes. The samples are preserved in the field by adding Lugol solution of a final concentration of 1%.

The suspension of sampled phytobenthos is mixed by shaking and a small amount is transferred with a pipette on to a cover slip. Diatom objects are determined to the species level with a 1000-fold magnification microscope. The slide is reviewed until new species are not found. More than 400 objects are evaluated**.**

Calculation of the Trophic index according to Rott et al. (1999):

$$TI = \frac{\sum_{i=1}^{n} TW_i * G_i * H_i}{\sum_{i=1}^{n} G_i * H_i} \qquad (12)$$

Where:
TI – Trophic Index
TW_i – Trophic value of species i
G_I – Weighting of species i
H_i – Abundance of species i in percent

Transformation of the Trophic Index:

$$TIEQR = 1 - (\frac{TI-0.3}{3.6}) \qquad (13)$$

Where:
TIEQR – Trophic index in EQR scale
TI – calculated Trophic Index.

Trophic and weighting values of species for calculation of TI are according to Rott et. al (1999) (also present in OMNIDIA database). Status class boundaries according to EŽI are presented in Table 23.

Table 23. National class boundaries for MEI EQR.

Class boundary	High/good	Good/moderate	Moderate/poor	Poor/bad
EFBI EQR	0.63	0.47	0.32	0.16

2.3. Physico-chemical indicators

Increased concentrations of nutrients (nitrogen and phosphorus) in lake water have caused eutrophication of many aquatic ecosystems. It is widely accepted that phosphorus concentrations limit primary production in most freshwater ecosystems, although nitrogen limitation or co-limitation is also possible under some conditions.

Physico-chemical parameters, including nutrient concentrations are supporting parameters in the assessment of ecological quality of surface waters. The WFD states that at good ecological status, nutrient concentrations must “not exceed the levels established so as to ensure the functioning of the ecosystem and the achievement of values specified (for good status) for the biological quality elements”.

Concentrations of total phosphorus (TP) and total nitrogen (TN) as well as Secchi depth measurements during summer are used in the ecological quality assessment of lakes both in Latvia and Lithuania (Tables 24 and 25). Yearly average nutrient concentrations at 0.5 m depth are used in Latvia, and average nutrient concentrations in vegetation season are used in Lithuania. Secchi depth characterises light conditions in the water column and it is used as an indicator of phytoplankton biomasses in oligohumic lakes. However, in humic lakes Secchi depth is not a reliable indicator of phytoplankton development, because high concentrations of coloured or chromophoric dissolved organic matter also decrease water transparency. Lithuania has included biochemical oxygen demand (BOD_7) in their assessment scheme. This parameter characterizes the availability of easily degradable organic matter in water. Boundary values of physical and chemical parameters for different classes of ecological quality are presented in Tables 24 and 25.

Table 24. Ecological status classes of lakes according to the physico-chemical parameters in Lithuania.

No	Quality element		Metric	Lake type	Boundary values for ecological status classes of lakes according to the values of indicators of physico-chemical quality elements				
					High	Good	Moderate	Poor	Bad
1	General elements	Nutrient conditions	N_{tot}, mg/l	1-3	<1,00	1,00-2,00	2,01-3,00	3,01-6,00	>6,00
2			P_{tot}, mg/l	1	<0,040	0,040–0,060	0,061–0,090	0,091 – 0,140	>0,14
3			P_{tot}, mg/l	2-3	<0,030	0,030–0,050	0,051–0,070	0,071 – 0,100	>0,10
4		Organic material	BOD_7, mg/l O_2	1	<2,3	2,3-4,2	4,3-6,0	6,1-8,0	>8,0
5			BOD_7, mg/l O_2	2-3	<1,8	1,8-3,2	3,3-5,0	5,1-7,0	>7,0
6		Water transparency	Secchi, m	1	>2,0*	2,0-1,3	1,2-0,8	0,7-0,5	<0,5
7			Secchi, m	2-3	>4,0	4,0-2,0	1,9-1,0	0,9-0,5	<0,5

* at a depth of less than 2 m, the transparency of the water is up to the bottom.

Table 25. Ecological quality classes of lakes according to the physical and chemical parameters in Latvia.

Type	Parameter	Unit	High	Good	Moderate	Bad	Very bad
1	TP	mg/l P	<0.025	0.025-0.050	0.05-0.075	0.075-0.100	>0.100
	TN	mg/l N	<1	1-1.5	1.5-2	2-2.5	>2.5
	Secchi	m	gr.>vid.dz.	1.5-2.2>vid.dz.	1-1.5	0.5-1	<0.5
2	TP	mg/l P	<0.025	0.025-0.050	0.05-0.075	0.075-0.100	>0.100
	TN	mg/l N	<1	1-1.5	1.5-2	2-2.5	>2.5
	Secchi	m	Not applicable due to high water colour				
3	TP	mg/l P	<0.025	0.025-0.050	0.05-0.075	0.075-0.100	>0.100

Type	Parameter	Unit	High	Good	Moderate	Bad	Very bad
	TN	mg/l N	<1	1-1.5	1.5-2	2-2.5	>2.5
	Secchi	m	gr.>vid.dz.	1.5-2.2>vid.dz.	1-1.5	0.5-1	<0.5
4	TP	mg/l P	<0.025	0.025-0.050	0.05-0.075	0.075-0.100	>0.100
	TN	mg/l N	<1	1-1.5	1.5-2	2-2.5	>2.5
	Secchi	m	Not applicable due to high water colour				
5	TP	mg/l P	<0.02	0.02-0.045	0.045-0.07	0.07-0.095	>0.095
	TN	mg/l N	<0.5	0.5-1	1-1.5	1.5-2	>2
	Secchi	m	>4	4.0-2.0	2.0-1.0	1.0-0.5	<0.5
6	TP	mg/l P	<0.03	0.03-0.055	0.055-0.08	0.08-0.105	>0.105
	TN	mg/l N	<0.8	0.8-1.3	1.3-1.8	1.8-2.3	>2.3
	Secchi	m	Not applicable due to high water colour				
7	TP	mg/l P	<0.015	0.015-0.035	0.035-0.055	0.055-0.075	>0.075
	TN	mg/l N	<0.5	0.5-1	1-1.5	1.5-2	>2
	Secchi	m	>4.5	4.5-2.5	2.5-1.5	1.5-1	<1
8	TP	mg/l P	<0.0225	0.0225-0.045	0.045-0.0675	0.0675-0.09	>0.09
	TN	mg/l N	<0.65	0.65-1.15	1.15-1.65	1.65-2.15	>2.15
	Secchi	m	Not applicable due to high water colour				
9	TP	mg/l P	<0.02	0.02-0.04	0.04-0.06	0.06-0.08	>0.08
	TN	mg/l N	<0.5	0.5-1	1-1.5	1.5-2	>2
	Secchi	m	>4.5	4.5-3	3-1.5	1.5-0.7	<0.7
11	TP	mg/l P	<0.025	0.025-0.050	0.05-0.075	0.075-0.100	>0.100
	TN	mg/l N	<1	1-1.5	1.5-2	2-2.5	>2.5
	Secchi	m	Not applicable due to high water colour				

Besides the parameters included in the ecological quality assessment scheme according to the WFD, there are many other physical and chemical parameters that are used as indicators of lake ecosystem health. In Latvia, there are established quality criteria for priority fish water. Priority fish waters are lakes and river stretch in which it is necessary to carry out water protection or water quality improvement measures to ensure favourable living conditions for the fish population. The list of priority fish waters has been specified in Annex 2. to the Republic of Latvia Cabinet Regulations, No 118 (adopted in 12.03.2002).

Priority fish waters shall be subdivided into:

- salmonid fish waters, in which salmon (Salmo salar), sea trout and brook trout (Salmo trutta), grayling (Thymallus thymallus) and whitefish (Coregonus) live or where it is possible to ensure the existence thereof.
- cyprinid fish waters, in which fish of carp family (Cyprinidae), as well as pike (Perca fluviatilis) and eel (Anguilla anguilla) live or where it is possible to ensure the existence thereof.

Water quality standards for some of the most measured parameters for priority fish water are summarised in Table 26.

Table 26. Guideline and limit values of most measured parameters for the priority salmonid and cyprinid waters (Cabinet Regulation No.118, adopted 2002).

No.	Parameter, unit	Salmonid waters		Cyprinid waters	
		Guideline value	Limit value	Guideline value	Limit value
1	Ammonium, mg/l NH_4^+	≤ 0.03	≤ 0.78	≤ 0.16	≤ 0.78
2	BOD_5, mg/l O_2	≤ 2		≤ 4	
3	Dissolved oxygen, mg/l O_2	50 % > 9 100 % > 7	50 % > 9	50 % > 8 100 % > 5	50 % > 7
4	Ammonia, mg/l NH_3	≤ 0.005	≤0.025	≤ 0.005	≤ 0.025
5	Nitrites, mg/l NO_2^-	≤ 0.01		≤ 0.03	
6	pH		6-9		6-9
7	Suspended solids, mg/l	≤ 25		≤ 25	

The **oxygen status** is of vital significance to any freshwater ecosystem. It may have large diurnal and seasonal variations, depending on trophic status of a lake, loading of organic matter, including humic substances from the catchment area. In the hypolimnion of stratified lakes, oxygen conditions are stable over a 24 h period. The worst oxygen conditions in hypolimnion are found at the end of summer and winter stagnation period when oxygen concentrations critical to many organisms may occur. Quality classes for oxygen concentrations in hypolimnion were developed by the Swedish Environmental Protection Agency (1991) Table 27. The lowest measured value during the year is used for the assessment of oxygen conditions in stratified lakes.

Table 27. Oxygen conditions in hypolimnion of stratified lakes (Swedish Environmental Protection Agency 1991).

O_2 in hypolimnion, mg/L	Designation
>7	Oxygen rich condition
5 - 7	Moderate oxygen condition
3 - 5	Weak oxygen condition
1 - 3	Oxygen poor condition
≤ 1	Anoxic or almost anoxic condition

The **lake sediments** act as a sink of polluting substances and thus studies of sediments can provide valuable information on how the ecosystem has changed. Pollutants accumulated in lake sediments (e.g., phosphorus, heavy metals, persistent organic pollutants) can be released under certain circumstances and are able to influence the lake ecosystem in a long run. Nowadays, when measures have been implemented to reduce pollution loads from external sources to lakes, the so-called 'lake internal loading' can become more important and prevent any improvement of water quality for a considerable period after the loading reduction. Analysis of nitrogen, phosphorus and carbon concentrations in sediments are needed to adequately address pressures to lake ecosystem health.

Sequential extraction (Hieltjes and Lijklema, 1980; Psenner et al., 1984) are used to analyse different forms in which phosphorus is found in lake sediments:

- Loosely sorbed or labile phosphorus, which is an estimate of easily available P in sediments.
- P bound to iron compounds (e.g., hydroxides) - become available under anoxic conditions.
- P bound to carbonates, apatite-P and P released by the dissolution of oxides (not adsorbed to the surface). P release occurs under acidic conditions.
- phosphates adsorbed to metal oxides (e.g., Al_2O_3) and other surfaces - P compounds are released if pH increases.
- Residual P - represented by organic and refractory P compounds.

Information on sediment quality and phosphorus speciation forms will allow us to assess the role of sediment on eutrophication processes and lake ecosystem health.

2.4. Indicators of Lake Hydromorphological Character

In Europe, hydromorphological pressures are recognized as the second most common type of pressure after eutrophication, impacting ecological quality of surface waters (Poikāne et al., 2020). Hydromorphological pressures are man-made alterations to the hydrological regime and morphological features of a lake and its surroundings. In Latvia, the Lake Habitat Survey (LHS) method developed by Rowan et al. (2004) is used for the assessment of hydromorphological pressures. However, the LHS method has been taken over from the United Kingdom and has not yet been fully adapted to Latvian conditions. In the future, it would be necessary to test this method in Latvian reference lakes, as well as in strongly affected lakes to determine the intensity of habitat modifications.

Schematic representation of survey sites according to the LHS method is shown in Figure 5.

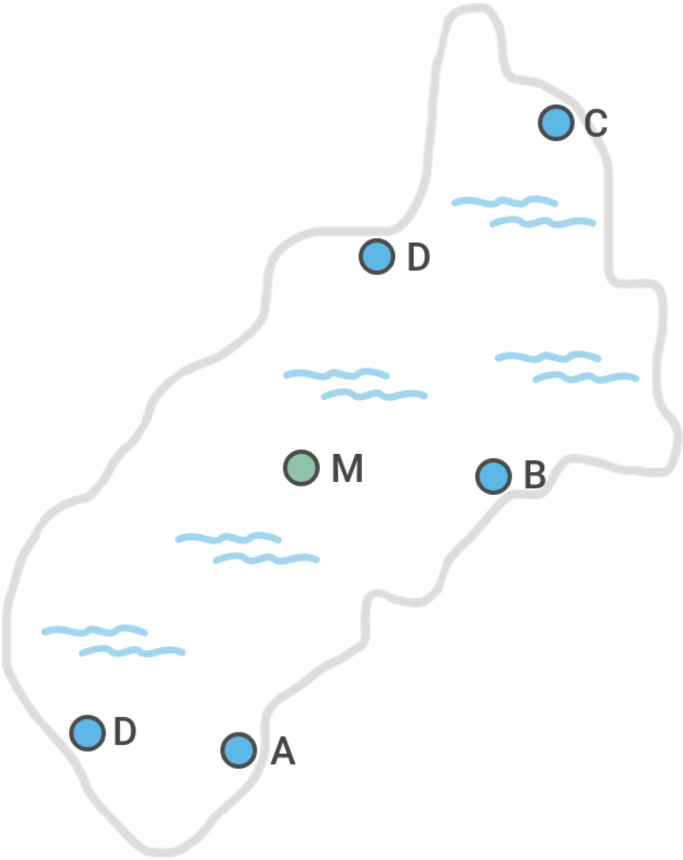

Figure 5. Possible location of sampling sites across the lake (A-E – survey sites of lake shores, M – survey site on the deepest part of a lake).

Seven components are considered when assessing hydromorphological pressures in Latvia:

1. **Hydrological regime:** diurnal, weekly and yearly changes of water level, occasions of water level changes per year (reversals), date when maximum and minimum water levels in a year were recorded, amplitude of lake water level daily or yearly changes and / or anthropogenic impact on hydrological regime such as presence of dams, alteration of inflowing or outflowing rivers, historically altered water level, presence of polders and amelioration ditches in the catchment, use of lake for energy and water supply, flood protection etc.

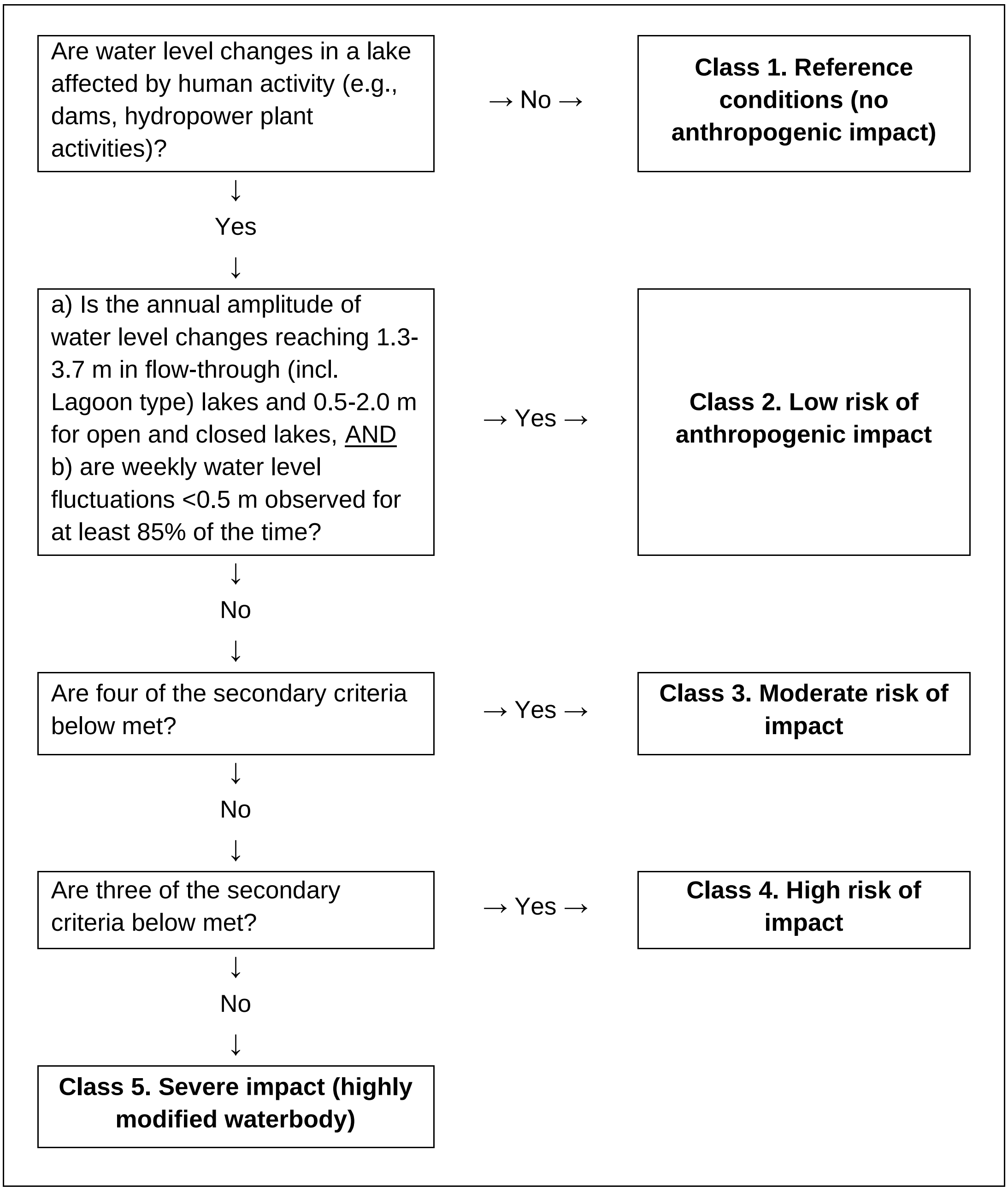

Figure 6. Assessment of anthropogenic impact on the hydrological regime.

Table 28. Secondary criteria of the assessment of anthropogenic impact on the hydrological regime.

No.	Criterion
1	Mean annual number of water level reversals is at least 50 (daily water level data are used).
2	At least 80% of dates of annual maximum water level fall between October 1^{st} and March 31^{st}
3	At least 80% of dates of annual minimum water level fall between March 1^{st} and October 31^{st}
4	Mean of annual maximum daily water level rises is between 0.6 and 1.0 m and a mean of annual maximum daily falls is between 0.2 and 0.55 m.
5	Mean annual amplitude of water level changes lies between 1.3 and 3.7 m for flow-through lakes (including lagoon type) or between 0.5 and 2.0 m for open and closed lakes.

Table 29. Significance of anthropogenic impact on lake hydrological regime.

Criterion	Significant impact	Moderate impact	Negligible impact
Presence of hydropower plant or dam	Presence of sluice or hydropower plant dam without fish pass	≥ 3 hydrotechnical constructions in the catchment	≤ 2 hydrotechnical constructions in the catchment
Regulation of inflowing and outflowing rivers	After 1980	Before 1980	Not regulated
Historic changes of water level	> 1m	≤ 1	Not changed
Share of polder areas in a catchment	≥ 5%	< 5%	No polders
Share of polder areas in a waterbody	> 10%	5-10%	< 5%
Network of amelioration ditches in a catchment		Present	Not present
Use for energy production, flood protection, water supply etc.	Substantial use of water and / or yearly fluctuation of water level between 0.5 - 5m		Not used

2. Modification of lake shoreline: the share of reinforced shoreline out of the total length of lake shoreline or number of surveyed sites with signs of shoreline reinforcement; share of shoreline impacted by polders.

Table 30. Assessment of lake shoreline modification.

Criterion	Significant impact	Moderate impact	Negligible impact
Shoreline reinforcement	≥ 50% of shoreline OR 5-7sites out of 10 surveyed sites OR 3 sites out of 4-5 surveyed sites.	≥ 30% - < 50% of shoreline OR 3-4sites out of 10 surveyed sites OR 2 sites out of 4-5 surveyed sites.	< 30% of shoreline OR ≤ 2 sites out of 10 surveyed sites OR ≤1 site out of 4-5 surveyed sites.

3. Intensity of use of lake shoreline: any non-natural land cover or land use (e.g., urban areas, roads, railways, beaches, recreational areas) or areas of agricultural land (e.g., arable land, pastures) within a 50 m distance from the waterline. Share of all these pressures is expressed as a percentage of total shoreline length or the maximum number of surveyed sites with the same kind of pressure.

Table 31. Intensity of lake shoreline use.

Criterion	Significant impact	Moderate impact	Negligible impact
Share of artificial and agriculture land cover types along the shore.	≥ 50% of shoreline.	≥ 30% - < 50% of shoreline.	< 30% of shoreline.
Presence of a single type of non-natural or agriculture land cover type in the surveyed stretches.	5-7sites out of 10 surveyed sites OR 3 sites out of 4-5 surveyed sites.	3-4sites out of 10 surveyed sites OR 2 sites out of 4-5 surveyed sites.	≤ 2 sites out of 10 surveyed sites OR ≤1 site out of 4-5 surveyed sites.

4. Human activities in a lake aquatory: activity of motorboats and rowboats, shipping, angling from a boat, angling from a shore, fish traps, swimming, presence of dams or barrier in a lake, bridges, military activities, harvesting of macrophytes, covering of water surface, dredging, liming, electric power lines across a lake. They are estimated as several activities in a lake.

Table 32. Significance of human activities in a lake.

Criterion	Significant impact	Moderate impact	Negligible impact
Number of human activities (pressures)	≥ 3 activities	2 activities	≤ 1 activity

5. Sedimentation regime: share of lake shoreline affected by coastal erosion (in %) or proportion of lake area affected by sedimentation or several surveyed sites where sedimentation processes have been observed over natural substrate in a littoral zone.

Table 33. Significance of sedimentation impact.

Criterion	Significant impact	Moderate impact	Negligible impact
Share of coastal erosion of total shoreline, %.	≥ 70% of shoreline.	≥ 50% - < 70% of shoreline	< 50% of shoreline.
Share of sedimentation zones compared to the total lake area (excluding vegetation islands), %.	≥ 70% of lake area.	≥ 50% - < 70% of lake area	< 50% of lake area.
Observed sedimentation over natural substrate in littoral zone.	≥ 7sites out of 10 surveyed sites OR 4 sites out of 4-5 surveyed sites.	5-6sites out of 10 surveyed sites OR 6 sites out of 4-5 surveyed sites	≤ 4 sites out of 10 surveyed sites OR ≤2 site out of 4-5 surveyed sites.

6. Physical and chemical conditions in the deepest part of a lake: transparency measured with a Secchi disk and /or oxygen stratification. Changes in oxygen concentration by depth is measured with a resolution of 1 m.

Table 34. Physical and chemical conditions in the deepest part of a lake.

Criterion	Moderate impact	Negligible impact
Transparency, m	<1.5 m	≥ 1.5 m
Concentration of dissolved oxygen, mg/L	< 4 mg/L	≥ 4 mg/L

7. Pressures related to the land use types in catchment: share of urban territories and / or anthropogenically impacted territories (including arable land and plantations) in the whole lake catchment area.

Table 35. Impact of land use types in catchment.

Criterion	Significant impact	Moderate impact	Negligible impact
Share of urban territories in catchment, %	≥ 8%	≥ 5% - < 8%	< 5%
Share of non-natural land cover (incl. cities) and / or arable land in catchment, %	≥ 40%	≥ 25% - < 40%	< 25%

Hydromorphological pressures are significant if the assessment based on the above-mentioned parameters and criteria shows that present conditions deviate from natural or reference conditions by 50% or more. Considering the significance of hydromorphological pressures, it is assumed that:

- score 0 corresponds to high quality (no hydromorphological pressure),
- score 2 corresponds to good quality (negligible impact or risk of hydromorphological pressures),
- score 4 corresponds to moderate quality (moderate impact or risk of hydromorphological pressures),
- score 6 corresponds to bad quality (high impact or risk of hydromorphological pressures),
- score 8 corresponds to very bad quality (extreme impact or risk of hydromorphological pressures).

Scores given for all seven components of hydromorphological assessment are summed up. Then the sum is multiplied by 100 and divided by the maximum possible sum of scores, which represents the worst-case conditions. The result is the deviation of the current lake hydromorphological conditions from the reference conditions expressed as percentages. Classification of lake hydromorphological status is summarized in Table 36.

Table 36. Classification of hydromorphological status of lakes.

Quality class	Deviation from reference conditions	Scores	Color code
1. high	<10%	0	blue
2. good	≥ 10% - < 30%	2	green
3. moderate	≥ 30% - < 50%	4	yellow
4. bad	≥ 50% - < 75%	6	orange
5. Very bad	≥ 75%	8	red

In **Lithuania,** the hydromorphological index for lakes (Ežero hidromorfologinis indeksas; EHMI) (TAR, 2016-08-09, Nr. 21814; Virbickas et al. 2016) is used to assess hydromorphological conditions of lakes. Index evaluates all three main elements of hydromorphological conditions:

1. Water level and water exchange. Lakes with a regulated water level can be determined from the data of the state cadastre of rivers, lakes and ponds. The degree of water level change is indicated in the technical documentation of the hydrostructures. Data on hydropower plants installed on lakes (and other lakeside water bodies) are collected by the Ministry of Energy.
2. Shore structure of the lake. The relative length of the strip of natural coastal vegetation and the extent of coastline changes due to reinforcement or erosion can be determined visually, based on aerial photographs and compared to the

shape of the coastline in the previous survey period (before lake level regulation), and can also be determined visually at the survey site.
3. Predominant substrate in the littoral zone. Determined visually at the survey site.

Variables of hydromorphological quality elements used for the calculation of Lithuanian lake hydromorphological index EHMI and their description are presented in the Table 37.

Table 37. Variables of hydromorphological quality elements and their description.

Variables		Description of lake ecological status according to parameters/indices/metrics of hydromorphological quality elements	*Score*
Water level and water exchange		There are no water level alterations caused by unnatural factors (water level is neither raised nor lowered, there is no water extraction, water flow is not regulated).	1
		Water level is raised, but water flow is naturalized.	2
		Water level is raised and stabilized (adjustments of water level are done to ensure safety of operation of hydro technical installation).	3
		Water level is raised and periodically alters due to operation of the electric power plant built on the lake outflow or water level and/or water exchange are periodically regulated because of other reasons. Or water level is lowered, but alteration is less than 1 m, lake area alteration is <10%.	4
		Water level is regulated, water level alteration exceeds 1m or an alteration in lake area is >10%.	5
Shore structure	Length of natural riparian vegetation belt	Not less than 70 % of the lake shoreline is covered by the belt of natural riparian vegetation (forest)	1
		70-30 % of the lake shoreline is covered by the natural riparian vegetation (forest) belt	2
		29-5 % of the lake shoreline is covered by the natural riparian vegetation (forest) belt	3
		<5 % of the lake shoreline is covered by the natural riparian vegetation (forest) belt	5
	Shoreline allterations	Shoreline is natural (neither straightened nor embanked) or < 5 % of the lake shoreline is altered	0
		5-25% of the shoreline is altered	1

Variables		Description of lake ecological status according to parameters/indices/metrics of hydromorphological quality elements	*Score*
		26-50% of the shoreline is altered	2
		>50% of the shoreline is altered	3
	Shore erosion	There is no shore erosion caused by unnatural factors (water level elevation/lift or water level alteration) or <5% of the shoreline is eroded	0
		5-25% of the shoreline is eroded due to unnatural factors	1
		26-50% of the shoreline is eroded due to unnatural factors	2
		>50% of the shoreline is eroded due to unnatural factors	3
Predominant substrate in the littoral zone		Clean, hard substrate (gravel and/or sand)	1
		Heterogeneous substrate: silty sand and/or gravel and/or clay, or hard substrate covered by a thin layer of silt	2
		Silt	3

EHMI is calculated as follows: EHMI=(sum of scores – maximal sum of scores)/(minimal sum of scores – maximal sum of scores).

Maximal sum of scores – 19

Minimal sum of scores – 3

The EHMI index values for the different ecological status classes are: > 0.90 – high; 0.90-0.80 – good; < 0.80 – less than good.

Testing the relationship between EHMI and indices based on fish and macroinvertebrates has shown that bio-indices are significantly correlated with EHMI; however, the EHMI relationship with the macroinvertebrate-based index is weaker than with the fish-based index.

III HYDROGEOLOGICAL INDICATORS FOR EVALUATION OF GROUNDWATER IMPACT

The ecological and chemical quality of SWBs associated with groundwater (GAAEs) or future deterioration in their ecological or chemical status is a key driver when considering assessment of GWBs. Another important factor is the determination of groundwater contribution - whether GAAE is critically dependent on groundwater or not. The 50% dependency criterion is often used. Also, it should be considered whether this dependency is continuous or seasonal and based on quantitative groundwater supply and/or chemical input. For instance, surface water aquatic species may be dependent on relatively non-polluted groundwater needed to maintain the ecology of polluted SWB as well. Thus, it is recommended to have a clear understanding of both quantitative and qualitative dependencies of GAAE prior undertaking any detailed WFD status assessments, as lack of information can lead to overestimation of problems and large investments to carry out PoMs (European Commission, 2015).

The identification of the level of groundwater dependency of an associated aquatic ecosystem may vary significantly. Some SWBs may have ecology that is critically dependent upon groundwater and may fail their WFD objectives when quality or quantity of groundwater input significantly change. Other SWBs may be able to withstand substantial changes in groundwater inputs and remain in good status. Permanently groundwater fed lakes are critically dependent and groundwater is the only source of water or contains chemicals that are critical for the ecology (e.g., Mazuika, Ummis lakes in Latvia). While not critically dependent are lakes where a significant component of their budget comes also from rivers and streams. Anyway, the estimation of dependency is a complicated yet so essential step (European Commission, 2015).

In this section we review approaches and methods we propose to use to at some extent solve above mentioned issues. We emphasize that many member states still struggle with identification of GDEs and their assessments due to multidisciplinary and complexity of GDE concepts.

3.1. Estimation of groundwater contribution to local water balance

Estimation of the water balance is a complicated task since some of its constituents, especially changes in groundwater storage, are difficult to measure directly and are often estimated indirectly through various water balance models or using analytic or empirical methods (Falalakis and Gemitzi, 2020). Estimation of groundwater recharge to a dependent surface water body such as a lake is often a challenge for hydrogeologists because direct measurements are costly, time consuming and difficult to carry out.

It is a challenge to estimate groundwater contribution to local water balance, especially in the areas of complex hydrological regime, as well as areas with insufficient monitoring data (Zacharias and Dimitrjou, 2003). Several estimation methods are described below which are mainly based on simple water balance calculations, using geographic information systems and data from topographic and geological maps. These methods can mainly be used in areas with a simple hydrological regime, where groundwater recharge takes place within the catchment area of an identified lake.

Simple water balance methods are widely used for quantifying groundwater recharge (Yin et al., 2011). The most common way of estimating recharge by the water balance method is the indirect or residual approach, whereby all the variables in the water balance equation except recharge (R) are either measured or estimated and R is set equal to the residual. The simple water budget is described by formula:

$$R = P - ET - R_0 - \Delta S \qquad (14)$$

Where:
P – precipitation, (mm/year)
ET – evaporation, (mm/year)
R0 – surface runoff, (mm/year)
ΔS – change in soil water, (mm/year).

The major limitation of the residual approach is that the accuracy depends on the accuracy of other components. Surface run-off may not be considered if most of the catchment area consists of sand and sand-gravel-pebble sediments with a high capacity to absorb precipitation. For long-term averaged steady state conditions, the soil-water content is constant. Therefore, ΔS could be assumed to be zero. However, it should be noted that the simplified water balance gives a rough estimation of groundwater contribution to the local water balance.

More accurate information can be obtained using a simple lake water balance described by following formula (World Meteorological Organization, 2008):

$$P + I = ET + Q + \Delta S \qquad (15)$$

Where:
P – precipitation, (km^3/year)
ET – evaporation, (km^3/year)
I – inflow, (km^3/year)
Q – outflow, (km^3/year)
ΔS – water storage changes, (km^3/year).

In this case water storage for years with low flow is the amount of groundwater. Precipitation is calculated using data of lake surface area (for annual water level) and yearly sum of precipitation from the nearest monitoring station. Inflow is calculated using surface water body area and analogue-river with flow measurements. In case of data lack might be used maps of average flow (M, l*sec/km^2). Evaporation is calculated using data of lake surface area and maps of Pastors (1987) "*Raionirovanie malih rek*

Latvijskoj SSP" and Q should be measured. It is important to note that this method will not provide accurate data on the contribution of groundwater to the local water balance, especially if it won't be possible to carry out the observations (monitoring) during the given period of time.

An **analytical method** based on the analytical elements of the balance calculation and the results obtained from the monitoring well levels can be used to describe the recharge of groundwater (Lebedev, 1976). Recharge rate is the result of the infiltration process (wt) described by following formula:

$$wt = \mu \frac{\Delta H_1 + \Delta H_0 R(\lambda)}{1 - R(\lambda)} \qquad (16)$$

$$R(\lambda) \text{ – function of } \lambda \frac{x}{2\sqrt{at}} \qquad (17)$$

Where:
μ – yield,
ΔH1 – level changes in the upstream well over a period of time t, (m)
ΔH0 – level changes in the downstream well over a period t, (m)
x – distance between boreholes (m)
a – alignment of the layer levels (m^2/d)
t – the time period corresponding to the specified level change (ΔH).

However, it should be noted that this analytical method can only be applied in the catchment areas where a balance monitoring station is located near the lake.

In case of insufficient data an alternative data source can be used to determine the groundwater recharge – a historic groundwater runoff map and information about groundwater recharge through infiltration (Prols and Delina, 1997). However, the scale of the map does not provide data precision, emphasizing on regional differences in groundwater discharge. Locally, the actual groundwater runoff can vary greatly and significant fluctuations in the groundwater runoff are possible, depending on the hydrogeological conditions at each site – e.g., permeability, drainage, the depth of the unsaturated zone. Also, it is important to note that the above-mentioned map is based on materials from the 1980s and may not appropriately describe the current situation.

The most precise assessment of groundwater contribution to local water balance can be achieved through area-based **water balance modelling** using measurement data (meteorological data, land surface parameters, e.g., soil moisture and land cover types, groundwater monitoring data etc.). It should be noted that creation of a model is an expensive, time consuming and complex process which requires infrastructure (boreholes) and long-term monitoring. Another efficient approach for estimating groundwater recharge into a water body is **hydraulic-head surfaces mapping** in a combination with water level measurements in boreholes and GIS techniques (Salama et al., 1995; Krogulec, 2010). However, this method is expensive, requires borehole infrastructure in the area and is difficult to implement under specific hydrologic conditions, especially if the aquifers are distributed in significant depths.

3.2. Groundwater flow and watershed

In moderate (humid) climate conditions when precipitation falls on the ground, a part of it evaporates back to the atmosphere and a significant part flows directly into surface water bodies by surface runoff. Finally, in Latvian conditions around 15% (rough estimate) infiltrate into the soil. Infiltrated water enters the vadose zone and some of it is used by vegetation. In moderate climate conditions surplus water percolates down to the water saturated zone and forms groundwater resources. During the recharge episodes the water table of shallow groundwater rises and slowly moves toward the closest drainage outlet - rivers, lakes, ditches etc. The areas where recharge occurs are called *recharge areas*, but where water flows into surface water bodies - *discharge areas.* The area between is called a transition zone, but the whole area where water flows are connected from recharge to discharge area is called *watershed*. Stream flow that persists in dry seasons is called *baseflow*. In most watersheds, groundwater discharge is the principal, and often the only, source of baseflow and during the dry seasons provides essential water input into surface water bodies (Poeter et al., 2020).

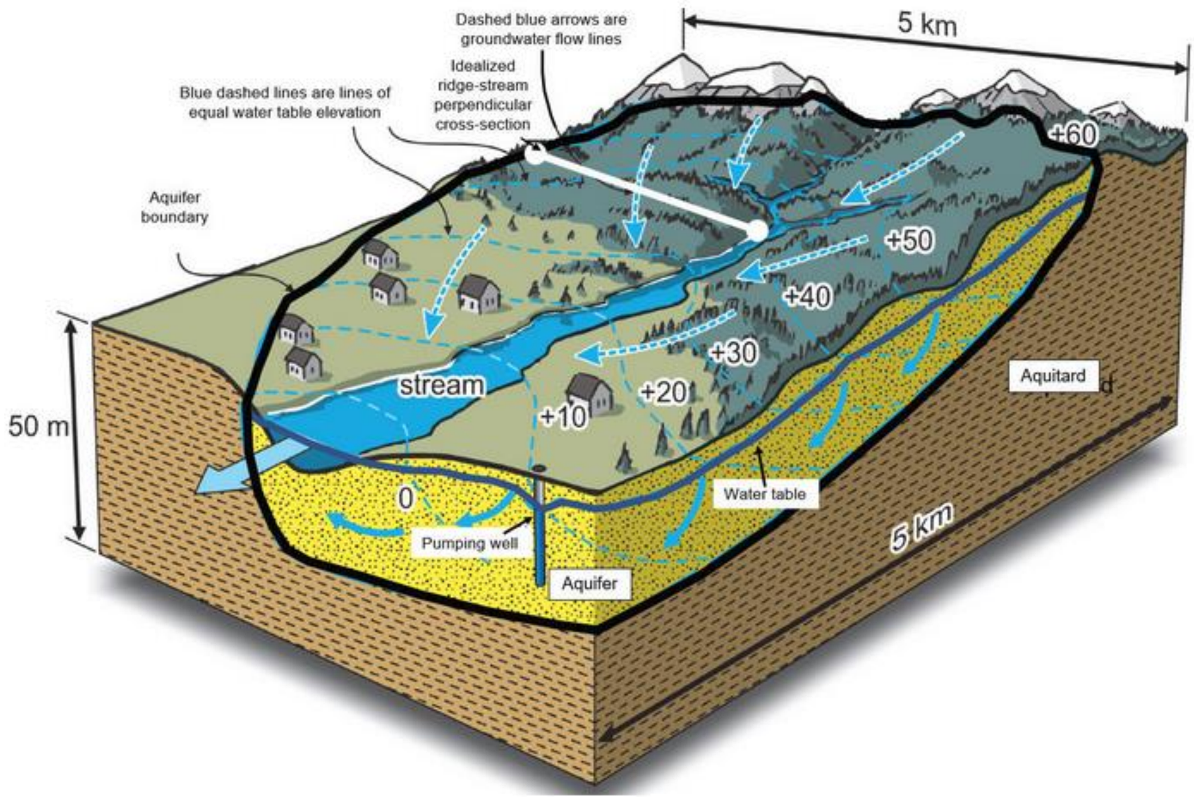

Figure 6. Groundwater flow in three-dimensional as shown in a groundwater basin/watershed (outlined in black, with the water table as a thick dark blue line, thin dashed blue lines of equal hydraulic head in three dimensions, and dashed blue arrows showing groundwater flow directions) (Modified Rivera, 2014, Poeter et al. 2020).

The elevation of the groundwater surface or water table in a piezometer (a well) relative to sea level is known as *hydraulic head*. The water table is higher in upland areas and the groundwater flows downward into the groundwater system and toward the lowlands due to gravitational force. The rate of water flow from the recharge areas in uplands to the discharge areas depends on the rate of recharge (water infiltrating to the water table), the elevation difference between recharge and discharge areas, and the permeability of the soils and rocks through which the groundwater is flowing. Groundwater flows from points with high hydraulic heads to points with low hydraulic heads (Poeter et al., 2020). Usually, shallow groundwater repeats terrain and its watershed boundaries coincide with surface water boundaries, while deeper aquifers may follow regional flows and have large, transboundary watersheds.

The water table in the bottom of a valley is closer to the surface than in the uplands and does not rise and fall as much as water tables under uplands. The hills may be perched even if there are no rain events for a longer time, but the valleys still receive groundwater from the uplands, because groundwater flow is relatively slow. In the dry season when active recharge does not occur, groundwater recharged before may travel many days, months, even years from the hills toward the valleys (Poeter et al., 2020), thus groundwater is available even at severe droughts, although it suffers from drought too. Delayed delivery of groundwater serves as a buffer, and it is often important to estimate this delay period length as it correlates with the local water system resilience not only to climate events, but also vulnerability to pollution migration.

A *watershed* is the land area that drains into a body of water (Figure 6). The boundary of a lake's watershed is defined by the highest points of the surrounding land around the lake. Precipitation (rain, snow melt) that falls in the watershed area flows by gravity over the ground into streams and shallow groundwater, finally entering the lake. Watershed is often also called a drainage basin/area or catchment. Delineation of watershed is important not only to estimate water balance, but also to implement proper risk assessment, as all the activities happening in the watershed (farming, urbanization, forestry, polluted sites, mining activities etc.) may influence the final recipient - the health of lake ecosystem.

In the case of a lake the watershed can be defined using terrain information (such as topography maps or DEM). Also, local anthropogenic activities such as ditches, pounds, amelioration etc. should be taken into account as it forms local boundaries. If available digitised information layers should be combined (e.g. in Latvia amelioration is available in system https://www.melioracija.lv/; Meliorācijas kadastrs, 2021). However, it is advised also to look at the latest available aerial photography to check for latest modifications nearby (e.g. variety of maps such as orthophoto or digital terrain are available in https://kartes.lgia.gov.lv/; LGIA, 2021).

Step by step delineation technique used to delineate surface water bodies in Latvia is described in Report X (project deliverable XXX). A simplified approach possible to carry out without any digital information is also described by EPA (1999).

3.3. Hydrochemical characteristic

Geology and water residence time are the major factors controlling natural water chemistry. Groundwater provides nutrients and electron receptors (e.g., sulphate) and usually creates specific physico-chemical conditions on GDEs. Water pH accounts for solubility and biological availability of nutrients and of heavy metals. At lower pH metals tend to occur in bioavailable forms, while nutrients are best taken at pH levels offering suitable adsorption conditions. Redox potential is important in the reductive dissolution of iron oxyhydroxides and the state of redox-sensitive elements (i.e., Fe, Mn, NH_4, NO_3). In GDEs linked to surface waters, seasonal and daily variations in photosynthesis can be a major natural cause of pH variations, thus modification of water chemical balance may result in irreversible changes in the entire ecosystem. Exposition of peat sulfides to oxygen due to groundwater lowering in dry periods can lead to oxidation of sulfides to sulphate and result in acidification of stream water (Kløve et al., 2011).

Elevated **nitrogen** compounds are one of the most common anthropogenic pollutants in Latvia and Lithuania (LV Report, 2020, LT Report, 2020; Retike et al., 2016a). Such pollution can be sourced by nitrogen fertilizers and manure, oxidation of organically bound nitrogen in soils, cattle feedlots, septic tanks, and sewage discharge. Usually, groundwater in Latvia has high denitrification potential, thus most of the pollution is reduced to N_2 and released in the atmosphere. Still, springs (especially outflow from fractured aquifers) and areas with high groundwater vulnerability might exceed Nitrate's directive (Nitrates directive, 2000) thresholds 50 mg/l for NO_3. Overall background level for NO_3 in Latvian groundwater is 4 mg/l (Retike et al., 2016b). Amount of baseline NH_4 in Latvian groundwater ranges from 0.043-0.85 mg/l (Bikše and Retiķe, 2019). Currently, there is a lack of knowledge on **phosphorus** compounds background levels in Latvian groundwater as the monitoring started only a few years ago. Still, the first results show negligible amounts of phosphorus in groundwater, however there is no threshold set for phosphorus compounds in Latvian groundwater.

In both Latvian and Lithuanian national groundwater monitoring programs are measure following parameters: field parameters (temperature, EC, pH, Eh, O_2), major ions, phosphorus compounds (TP and PO_4), nitrogen compounds (NH_4, NO_2, NO_3, TN, permanganate index), and heavy metals (Cd, Pb, Ni, Hg, As). Important difference is that Latvia analyzes total iron in field conditions, while Lithuania in the laboratory, so these parameters should not be interpreted together. Also, B-solutions (2019) project showed that Latvian Environment, Geology and Meteorology laboratory in Latvia which analyse all water samples in Latvian monitoring network is accredited according to LVS EN ISO/IEC 17025:2005 standards and it has a Latvian national accreditation bureau registration number LATAK-T-105-34-97, while Lithuanian Geological Survey laboratory is not accredited yet. There have been some significant deviations in intercalibration process carried out in 2019, thus a combination of data from both countries should be carried out with precaution. All possible chemical pollutants which may naturally occur in groundwater (e.g., NO_3, PO_4, As) have their baseline levels set at Latvian GWB scale (Bikše and Retiķe, 2019), for all synthetics parameters TVs are ½ of their maximum environmental criteria values.

IV PRESSURES AND IMPACTS ANALYSIS

4.1. Estimation of the lakes residual time

Lake residual time (also called the residence time or retention time of lake water) is the calculated average time of water mass renewal in a lake. At its simplest, this figure is the result of dividing the lake volume by the inflow or outflow of the lake. It describes the amount of time that a substance stays in the lake.

According to international hydrological praxis the residual time can be calculated according to the following formula:

$$Period\ of\ water\ retention\ (years) = \frac{lakes\ volume}{inflow + precipitation} \qquad (16)$$

The amount of inflow is calculated for each lakes tributary separately using the data from the nearest hydrological station on a particular river or (in case of missing data) using average runoff maps (A.Pastors, Raionirovanie malih rek Latvijskoj SSP", 1987) and following formulas:

$$Wp = Q_{yearly\ avg.} * T/10^9 \qquad (17)$$

Where:
Wp – yearly river inflow, km^3/year
T – number of seconds in a year, $T=31.56*10^6$

Q yearly avg. – yearly average water discharge in the cross-section of the river estuary, which is calculated as:

$$Q_{yearly\ avg.} = 31.7 * 10^{-6} * R * A \qquad (18)$$

Where:
A – area of a river basin, km^2
R – long-term average runoff layer according to isolines, mm (A.Pastors, 1987. g. "Gadā noteces slāņa kartē", mm). For large basins which cross more than one isoline, the covered areas by each isoline should be distinguished and finally the weighted average should be calculated.

The calculation of lake volume is based on a lake depth and water level measurements. Existing information of previous researches (Latvijas Valsts meliorācijas projektēšanas institūta ezeru baseinu shēmām ("Daugavas (Lielupes, Ventas, Gaujas) baseina ezeru un to apkārtējo platību kompleksās izmantošanas un aizsardzības shēma / Latvijas Valsts meliorācijas projektēšanas institūts, ЛАТГИПРОВОДХОЗ.- Rīga, 1972) can be used in case of measured data gaps.

In the lakes scheme the depth isolines (isobaths) divide the lakes body in layers with the thickness similar to the step of the isobath. Isobath is an imaginary line or a line on a map or chart that connects all points having the same depth below a water surface. The volume of the lake is calculated as follows:

$$V = h * (w0/2 + w1 + w2 + \cdots + wn/2) + 1/3 * wn * (hmaks. - hn) \quad (19)$$

Where:
V – lake's volume, m^3
w0, w1, w2, wn – the area of layers, m^2
h – the thickness of layers, m
hmaks – maximum depth of the lake (shown in lakes scheme)
hn - the depth of the largest isobath, m.

In cases when outflow from the lake is much greater than inflow, the residual time of a lake is calculated according to following formula:

$$Period\ of\ water\ retention\ (years) = \frac{Lake'swolume}{outflow+evaporation\ from\ open\ water\ surface} \quad (20)$$

The amount of outflow is calculated with the following formula (17), where Q yearly avg is a yearly average water discharge in the cross-section at the river outlet.

The amount of precipitation is calculated according to the following formula:

$$Pex. = P * Fsp * 10^{-6} \quad (21)$$

Where:
Pez. – the amount of yearly precipitation in the lake, km^3/year
P – precipitation amount in a year according to the data from the nearest meteorological station, mm
Fsp. – lake area, km^2

The amount of evaporation is calculated according to the following formula (using A. Pastors, Vidējās iztvaikošanas kartes (“*Raionirovanie malih rek Latvijskoj SSP”, 1987):*

$$Eez. = E * Fsp * 10^{-6} \quad (22)$$

Where:
Eez. – the amount of evaporation from the lake in a year, km^3/year
E - yearly evaporation amount according to evaporation maps by A. Pastors, mm
Fsp. – lake area, km^2

4.2. Lakes inflow and outflow discharge measurements

The discharge measurements of surface water bodies such as rivers and streams provide valuable information on their hydrological regime. Especially, these data are important for the evaluation of the ecological status of lake water bodies that have inflowing and outflowing streams. The hydrographic system of case study lake Ilgė (Garais) consists of the periodically flowing inlet of G-1 stream from the Apvalasai lake and outflowing Minava stream (Figure 7). Accordingly, Ilgė (Garais) lake falls within the type of semi drainage lakes.

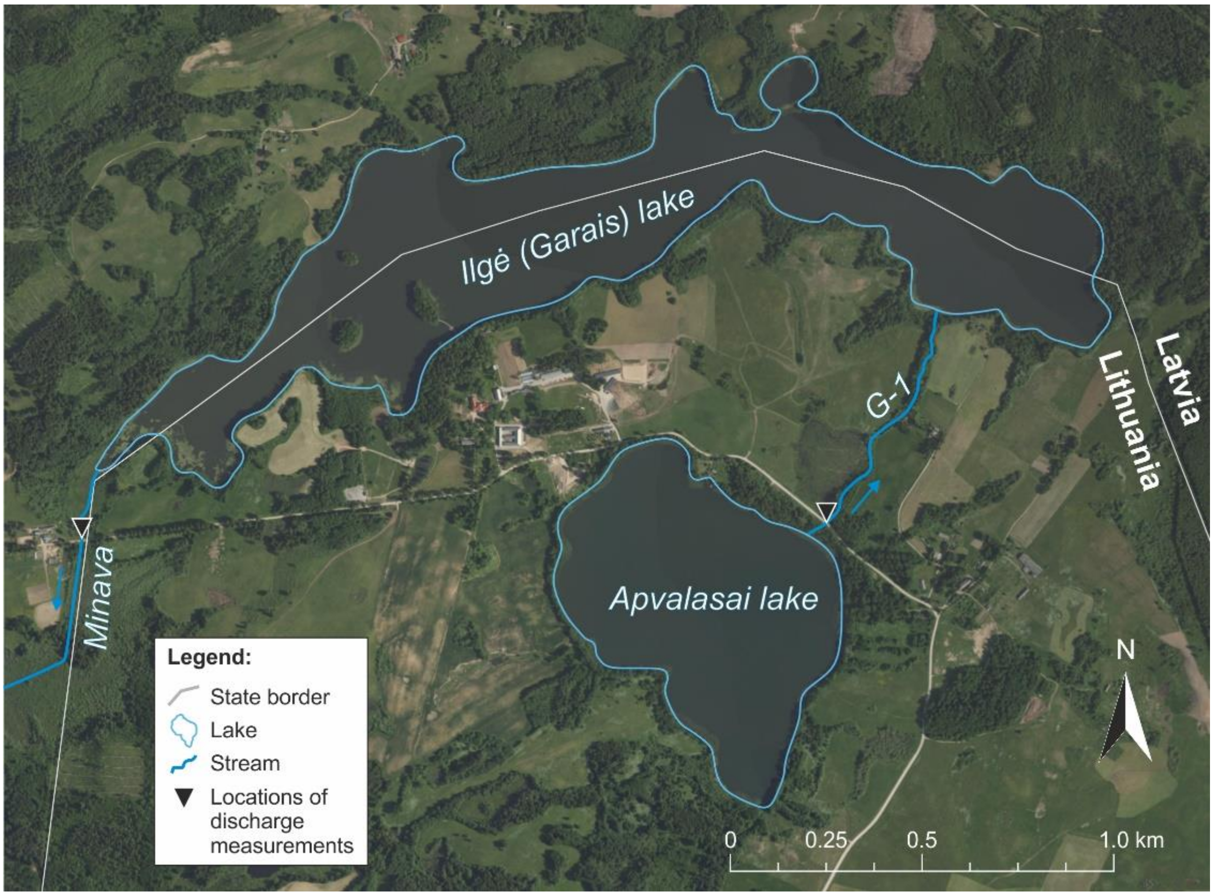

Figure 7. Scheme of inflow and outflow discharge measurements in Ilgė (Garais) lake.

To evaluate water exchange processes and total water balance of the selected lake, the discharge measurements of inflow and outflow are essential. Therefore, the discharge measurements of the inflow and outflow of Ilgė (Garais) lake will be carried out in selected locations as shown in Figure 7 once per three months. The observations will be done at road crossing culvert pipes which define a clear profile of supplying and draining water content. At these monitoring points, the diameter of the pipe (D), the height of water level (h), and flow velocity (u) at different depths will be measured (Figure 8).

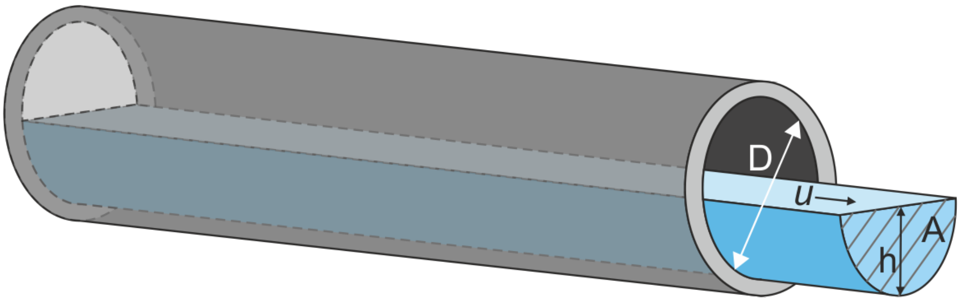

Figure 8. Discharge measurements at culvert pipe.

Flow velocity will be measured with electromagnetic flow meter *Valeport Model 801*. The average velocity (*v*) will be recalculated as follows according to the collected data (depending on the water level height and measured points per vertical), i.e., if three depths:

$$v = 0.25(u_{0.2} + 2u_{0.6} + u_{0.8}) \qquad (23)$$

if two depths:

$$v = 0.5(u_{0.2} + u_{0.8}) \qquad (24)$$

if one depth:

$$v = u_{0.5} \qquad (25)$$

The distribution of flow velocities in the pipe is not equal through the profile, since pipe walls cause some friction, therefore all calculated velocities will be multiplied by coefficient 0.9 (Karasev and Shumkov, 1985). According to pipe diameter and height of water level, the area of flow profile (*A*) will be estimated for the final discharge (*Q*) calculation:

$$Q = 0.9\overline{v}A \qquad (26)$$

During TRANSWAT project once per three months measured and calculated discharges of the inflow and outflow will be linked to the automatic water level measurements of Ilgė (Garais) lake collected by water level loggers of LEGMC. The combined data will provide an opportunity to create a water level-discharge rating curve for evaluation of daily inflow and outflow discharges of the case study lake.

4.3. Land use

Land use has become an important indirect parameter in the assessment of possible pressures from extra nutrient fluxes. It is one of the factors that correlates the most with the water quality in catchments. Depending on the land use type the catchment area of a water body can serve either as a protective barrier – promoting nutrient reduction or, conversely, to facilitate nutrient increase, for example, through deforestation.

The proportions of land use types in the catchment area helps to assess diffuse pollution loads, the risk of increase in eutrophication and sedimentation in the lake (Urtāns et al., 2017). Various land use activities can lead to increased risk of pollution - nutrients from fertilizers, pesticides, sediments, heavy metals, oil pollution and other chemical compounds can reach the water body with surface runoff, wastewater and, in some cases, pollution can reach the groundwater (EPA, no date).

Many studies have shown that urban and agricultural areas are responsible for water pollution, while forested and wetland areas show mostly natural loads. Impact of **forest land use** without human intervention would have no significant adverse effect on the water quality. Forests contribute to maintaining good quality of surface waters. However, nutrient runoff from anthropogenically affected forest areas (clear-cuts and ameliorated forests) is 20% higher than in natural forests (Abramenko, 2013).

Agricultural lands are one of the main diffuse nutrient pollution sources. In many parts of the world areas of agricultural land in river basins increase. Fertilizer use in agriculture leads to an increase of the nutrient, such as phosphorus and nitrogen, loads. Agriculture can also be a source of other chemical chemical pollutants, such as pesticides, herbicides etc. In catchment areas where arable lands are dominant, a greater adverse impact on water quality has been noted (Kändler et al., 2017). Agricultural activities such as livestock grazing can also contribute to changes in water quality by changes in runoff due to reduced vegetation (Randhir, 2007).

The proportion of **urban and semi-urban lands** and major roads in the catchment area can have a negative impact on water quality. Runoff from these areas is related to oil pollution from petrol stations and roads, nitrate compounds (from garden fertilizers, urine) and heavy metals. Nitrogen and phosphorus compounds, heavy metals, oil, etc. can enter the lake with surface runoff and dust particles from the areas with high density of major roads. If during the winter salt ($NaCl$, $CaCl_2$) is spread on the roads it can enter the lake with the meltwaters (Melluma and Leinerte, 1992).

To effectively identify land uses and assess the magnitude of diffuse pollution loads in the catchment area and their potential impact on the aquatic ecosystem, land uses can be divided into 8 groups (Table 38), a more detailed division is not necessary. If needed, it can be assembled into 5 large groups - agricultural, urban areas, forests, wetlands, and water bodies (GENESIS, 2015).

Table 38. Land use types by contribution to pollution loads.

Anthropogenic loads	Natural background loads
1. Arable lands 2. Grazing lands 3. Urban lands 4. Clearcuts	5. Other agricultural lands 6. Forests 7. Wetlands 8. Water bodies

If the proportion of land use types contributing to anthropogenic pollution loads in the catchment area accounts for more than 20% of the area, negative impact on water quality can be detected, in which case performance of in-depth research and data analysis is needed. Runoff from anthropogenic areas causes diffuse pollution, it is a complex process which depends on many factors and their interactions. The most important factors are climatic conditions, topography of the catchment areas, geology, vegetation types, soil properties, land use type and intensity of land management. The impacts of human activity affect the hydrological regime of water bodies and the chemical composition of waters (Dahm et al., 2013; Lagzdiņš, 2012).

To identify the distribution of land use types in the catchment area, it is necessary to use ArcGis software (or open sources GIS software such as QGIS) and available information, which can be collected from the following data sources:

1. CorineLandCover, land use types, 2018. Available: https://land.copernicus.eu/pan-european/corine-land-cover/clc2018
2. Rural Support Service (RSS), areas of arable land, areas of agricultural territories, 2018.
3. Agricultural Data Centre, animal units, 2018.
4. State Forest Service, forest types, clearcuts, 2018.
5. The Latvian Geospatial Information Agency, Orthophoto data, 2021. Available: https://kartes.lgia.gov.lv/

4.4. Nutrient source apportionment

Model description. Nutrient source apportionment modelling with FyrisNP model can be used to determine source apportioned gross and net transport of nitrogen and phosphorus in rivers and lakes. The time step for the model is one month and the spatial resolution is on the sub-catchment level. Retention, i.e., losses of nutrients in rivers and lakes through sedimentation, up-take by plants and denitrification, is calculated as a function of water temperature, nutrient concentrations, water flow, lake surface area and stream surface area. The model is calibrated against time series of measured nitrogen or phosphorus concentrations by adjusting two parameters (Hansson et al., 2008).

Input data. Data used for calibrating and running the model can be divided into time dependent data, e.g., time-series of observed nitrogen and phosphorus concentration, water temperature, runoff and point source discharges, and time independent data, e.g., land-use information, lake area and stream length and width. To perform simulations with the FyrisNP model, an Excel-file containing all input data is required. The Excel data file consists of eight to ten different worksheets depending on features used. It must contain data describing sub-catchments, such as land use data, data about stream lengths and lake areas etc., data about water temperature, N_{tot} and P_{tot} concentrations in runoff from different land use types, observed P_{tot} or N_{tot} concentrations, minor point sources and major point sources of nutrients. For minor point sources data about residents not connected to centralized sewerage systems will be used and for major point sources - nitrogen and phosphorus concentrations in wastewater treatment plant discharge (Hansson et al., 2008).

Results. Once the Excel file with is uploaded into the model, the data is automatically assigned to sub-catchments. The model determines the number of monitoring stations. Calibration is performed automatically, starting with the Monte Carlo method, and afterwards is completed with manual calibration. When complete, it is possible to analyze the calibration results - observed and simulated concentrations. Nutrient loads are calculated by months. In the result section of the model the incoming and the outgoing load in sub-catchment can be viewed and source apportionment of nutrients from all the land use types, minor and major point sources is available. The result data can be downloaded as an Excel file and used for further analysis or graphic depiction.

4.5. Groundwater abstraction

Global water use has increased in the last decades due to increase in global water demand. This comes as a result of an increasing number of global population as well as the changes in dietary that has a large effect on water consumption both as directly used water as well as virtually used water through products consumed (Bierkens and Wada, 2019). This demand is met by using both surface waters and groundwaters while the last one is of more importance due to having generally better quality. Greatly increasing groundwater trends can be observed not only in the world's megacities, but also in the agricultural sector who is a large consumer of groundwater for irrigation purposes. Even though smaller villages and households generally do not consume huge amounts of water, the attention must be paid if groundwater is the main source of water as even relatively small groundwater abstraction rate can cause changes in groundwater systems that can have negative impacts on other groundwater users with groundwater dependent ecosystems as an example.

Groundwater abstractions have an impact on groundwater dependent ecosystems through lowered groundwater levels and decreased discharge to streamflows, lakes, wetlands, springs. However, the impact of groundwater abstraction is strongly related to pumping rate, groundwater recharge rate and the storativity of the aquifer.

In natural conditions groundwater flows from recharge area to the discharge zone that can be either river, lake, spring, or a sea (Figure 9, a). When groundwater pumping starts to take place, some water that originally contributed to the lake (or other surface water body) is taken out from the system, causing lowered groundwater levels next to the abstraction site and slightly decreasing water level in surface bodies, although surface water balance still benefits from groundwater discharge (Figure 9, b). At higher pumping rate the groundwater system is affected to such a high degree that the lake is losing its water to groundwater to meet the pumping demand in the abstraction site (Figure 9, c). In such conditions the surface water body loses its water input from groundwater that might affect water chemistry, temperature, and other parameters, including lake water level. Decreased groundwater levels result in a thicker aeration zone, causing more soil to be exposed to oxygen that can yield other negative effects. If the groundwater pumping rate is extremely high (or aquifer water storage properties are poor), groundwater can be lowered to a such low level that lake water is disconnected from groundwater meaning that the soil under the lake (or any other water body) is not fully saturated (Figure 9, d). In these extreme situations lake water is freely infiltrating the groundwater at its maximum rate and the surface water body can lose a significant amount of water resources, leading to distinctively lowered surface water levels.

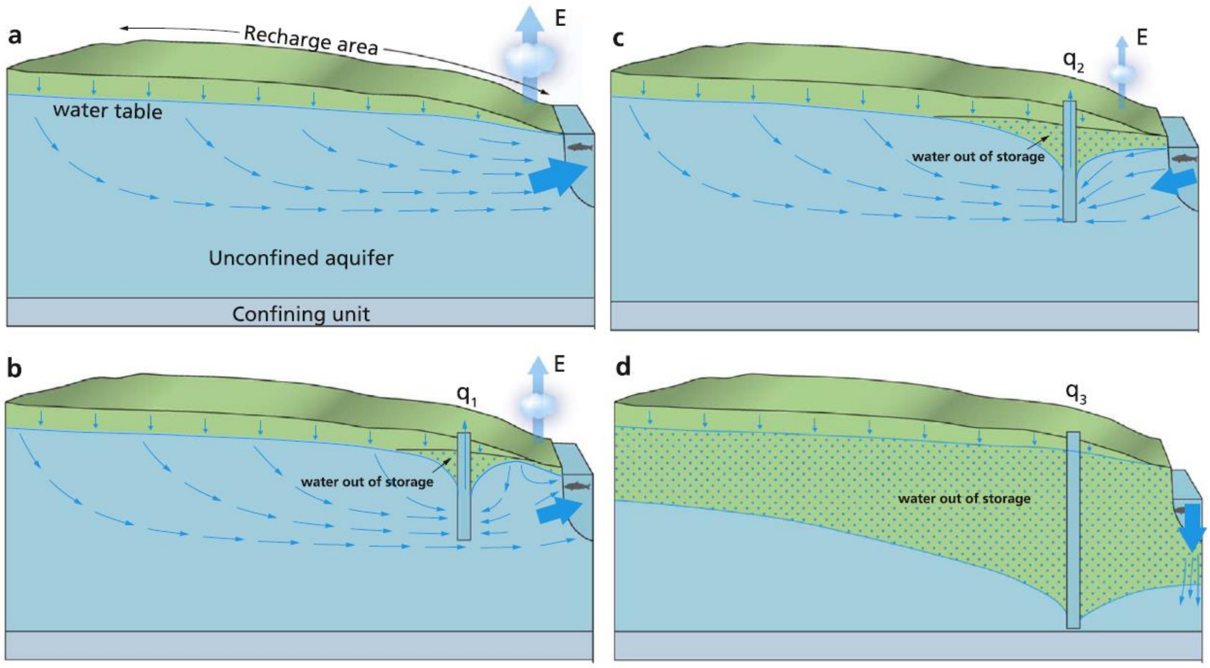

Figure 9. The impact on groundwater abstraction on surface water bodies according to the pumping rate (modified from Bierkens and Wada, 2019).

Groundwater level can have an impact on water chemistry and ecosystems. For example, a shallow groundwater table typically is dominated by nutrient poor alkaline rich groundwater that promotes low productive fen ecosystems with rare species. However, lowering of water table can cause drastic changes to soil water chemistry – oxidation of iron and sulphide can yield sulphuric acid, whereas increased dominance of rainwater in the soil zone can cause reduction of pH and, subsequently, cause mobility of potentially toxic metals. Furthermore, lowered groundwater level leads to oxygen increase that induces mineralization of organic matter which increases nutrient availability, in particular nitrogen (Bierkens and Wada, 2019). This example raises the importance of necessity to assess anthropogenic impact on groundwater levels.

The assessment of the groundwater abstraction impact on groundwater dependent ecosystems is a challenging task. There are many obstacles that make it difficult to do the assessment. A common problem is the availability of data to prove that groundwater level has decreased due to groundwater abstraction. To overcome insufficient data driven problems, a five-step scheme for the assessment of significant damage to GDE caused by quantitative pressures, including groundwater abstraction, has been developed and published by Retike et al. (2020). Such a scheme provides a logistic pathway to assess GDE starting from collecting any piece of evidence that water level has been lowered in the ecosystem to conducting an annual monitoring to reasonably make evidence-based conclusions. The step 2 in the scheme is devoted to investigating groundwater abstraction in the vicinity of the GDE. The impact of the abstraction on GDE is determined by the distance between the abstraction site and GDE. However, as groundwater level response to groundwater abstraction is depending on local hydrogeological conditions, it is hard to find one single approach to assess the relevant distance. Retike et al. (2020) provides a simplified approach to find the relevant distance from GDE to the abstraction site that can be used as an indicator if the abstraction can have a negative impact on GDE. Groundwater abstraction rate can be considered relevant to the GDE based on the distance from GDE according to the equation:

$$X = \sqrt{\frac{Q_{yr}}{\pi * R_{yr}}} \qquad (27)$$

Where:
X – distance from water abstraction site (m),
Qyr – abstraction rate (m^3/yr),
Ryr – average recharge rate (m^3/m^2/yr).

The value for Ryr preferably should be taken locally, from recharge maps, groundwater balance stations or gridded climate model data, but if nothing is available, a value of Ryr = 0.07 m^3/m^2/yr can be used (Retike et al., 2020). If GDE is closer to the abstraction site than the value of the x, groundwater abstraction is considered relevant to the GDE and it may have a negative impact.

V DEVELOPMENT OF CONCEPTUAL MODELS FOR GAAE MANAGEMENT

Groundwater provides baseflow, mostly good quality water with a stable temperature, and may buffer the temperature increase following climate warming. However, it also can transmit stressors to surface waters, for instance nitrate from agricultural fields to streams. Being a relatively slow system, these stressors are also lagged in time or transferred during the flow path. Groundwater both propagates and buffers stressors, but its effects depend on the local geology, climate, land-use, stressor combinations and scale (Kaandorp et al., 2018).

Whole GWB can be characterised as being in poor status if at least one of identified GAAEs is damaged because of anthropogenic alteration to connected GWB. Currently, the proposed approach is to set all GWB in bad status not considering GWB and damaged ecosystem size. This is a precautionary step but may lead to costly PoMs. However, it should be highlighted that PoMs should not be addressed to all GWB, but only to the negatively affected areas. All in all, it means that it is necessary to know the exact watershed or catchment where the negative influence is happening and understand involved mechanisms/processes i.e., how the pollution is transferred to the ecosystem (so called "source-pathway-receptor" approach) (Brkić et al., 2019; European Commission, 2015).

It is often recommended to use a modeling approach to understand the interactions between GAAE and GWB, while the most important step is to develop conceptual models of the whole system at first. Only after and if necessary, more detailed, and complicated numerical models can be developed (European Commission, 2015). Still, in most cases a conceptual model will have a satisfactory level of detail. It should be highlighted that numerical models require a good knowledge base and presence of various data usually missing, and of course time and knowledge how to build them. While the conceptual model can indicate where the main knowledge gaps are and what kind of information should be gathered at first, and if the numerical model is needed at all (Retike et al., 2020).

The term "conceptual model" is not set in the Groundwater Directive, nor is there a common definition by the Guidance Documents that recommend its use. A hydrogeological conceptual model usually describes and quantifies the relevant geological characteristics, flow conditions, hydrogeochemical and hydrobiological processes, anthropogenic activities, and their interactions. The degree of detail is based on the given problems and questions. Conceptual models can be applied under circumstances, from detailed assessments to a simplified scheme of interacting processes for communication purposes with stakeholders. Development of a conceptual model is one of the basic steps for the management of groundwater bodies (European Commission, 2010).

Main points during conceptual model setup (European Commission, 2010):

1. Main characteristics:
    a. scope and questions to be answered by the model to define the degree of detail and complexity of conceptual model,
    b. identification of vertical and horizontal boundaries,
2. Parameterisation and quantification:
    a. description and quantification of important hydraulic, geochemical and hydrochemical parameters,
    b. land use and other important pressure distribution.
3. Dealing with uncertainties - potential uncertainties, variability, and whether data are representative.
4. Iterative evaluation of a conceptual model. It is suggested to start with a simple model, then analyse its performance and make a more complex model if the simpler model is not sufficient.

It is essential to document all steps of development of the conceptual model and all the data sources and time periods used. It should be clearly shown, where improvements and iterations were made. The complexity of the visualisation depends on the aim of the study (European Commission, 2010).

5.1. Identification of the receptor and aim of the model

First, **the aim of the model should be agreed** as it affects the detail of certain parts of the model and data needed, and consequently, it will be able to answer only those questions which it was intended to answer. If the aim e.g., is to study agricultural impact on GAAE status then nutrient inputs and land use data will be essential. Still, such a model most probably will not answer the question about water abstraction impact, and so on.

It is important that in this step all stakeholders are involved, especially, end users. In assessment of GAAE the knowledge on local hydrogeology conditions and hydrology cycle is essential. Discussions should be held with surface water and groundwater ecologists and surface water managers to understand the location and groundwater needs of GAAE. If GAAE is a part of Natura 2000 network, there should be discussions with conservation ecologists too (European Commission, 2015). The time devoted to discussions and agreeing on what is the aim of the model will save the time in near future and more probably will deliver better quality results.

5.2. Delineation of spatial and temporal scale

After agreeing on the aim of the conceptual model, it is necessary to define its areal extent and boundaries (horizontal and vertical boundaries). The spatial boundaries of the model should be carefully considered and set in 3 dimensions to catch the effect and cause. In case of doubts, it is recommended first, to extend the model area far beyond the area of interest and then, when new data comes in, the area can be decreased. Iterative development of conceptual model will lead to a better understanding of the whole research system (European Commission, 2010).

In the case of lake ecosystem assessment first, the boundary of a GWB should be considered. Then the spatial boundaries may be decreased to specific lake ecosystem watershed or catchment (most likely SWB boundaries). If possible, water balance for the area covered by the conceptual model should be defined. Vertical boundaries (hydrogeological units) must be defined as well. Formations with comparable hydrogeological characteristics must be combined and important heterogeneous areas must be kept. In case of lake ecosystem assessment, it is necessary to know the depth of the lake and surrounding geological conditions. Temporal scale is also important as it describes system dynamics (like infiltration rates, geogenic changes of physical/chemical groundwater properties). Temporal aspects can be distinguished into natural variations (e.g., seasonal effects) and anthropogenic influences (like rising concentrations, decreasing groundwater levels) (European Commission, 2010).

The results of this step can be shown as cross sections, maps, block diagrams, providing: (1) spatial distribution/delineation of hydrogeological units in the area, (2) description of monitoring network and (3) information on groundwater flow directions (European Commission, 2010).

5.3 Quantification of parameters

Such data should/may be considered during the development of conceptual model (European Commission, 2010):

- **Typography** (morphology, surface waters, surface water catchment area);
- **Geology** (lithology, stratigraphy);
- **Hydrogeology** (groundwater catchment or body area, aquifer geometry, hydrogeological units - aquifer type, permeability, confinement, unsaturated zone, estimation of flow directions etc.). In the case of groundwater dependent lakes in Latvian-Lithuanin conditions representative conceptual cross-sections will be a) gaining lake or d) flow through the lake (Figure 10, 11, 12, 13).
- **Hydraulic data** (hydraulic conductivity, porosity, groundwater levels, hydraulic gradients, recharge, discharge etc.). The results of hydraulic assessment can be shown as cross sections, maps, block diagrams, providing: (1) quantified water balance, split to different components of discharge and recharge, (2) groundwater flow directions, (3) depth to groundwater table, (4) travel times of seepage and groundwater etc. (European Commission, 2010).
- **Hydrochemical data** (pH, temperature, conductivity, redox potential, alkalinity, dissolved oxygen, dissolved organic carbon, major ions etc.) should allow to identify baseline chemical composition. The results of this step can be shown as maps, diagrams, providing (1) groundwater chemistry characterisation in time and space, (2) natural background levels etc. (European Commission, 2010).

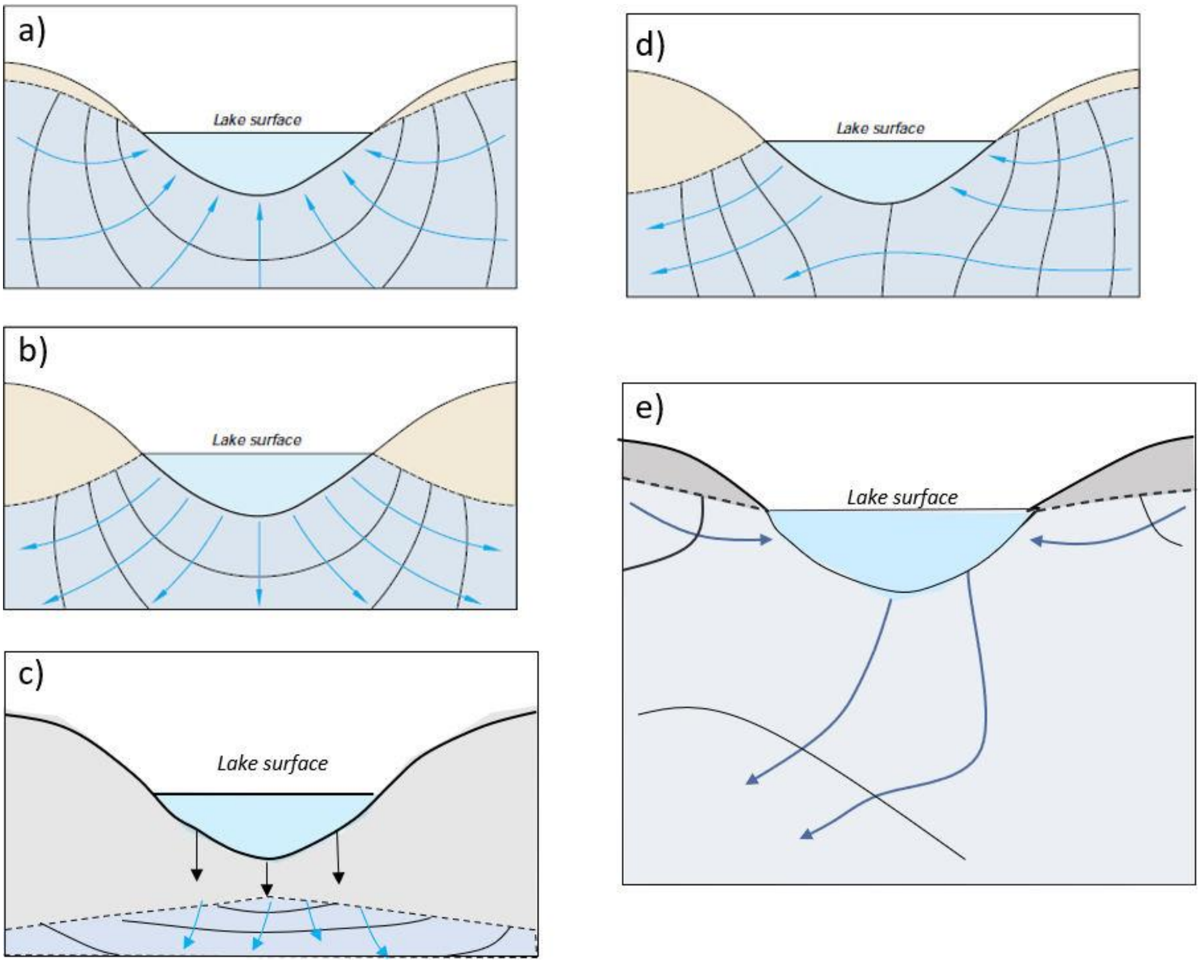

Figure 10. Conceptual cross sections of lake-groundwater exchange. Blue arrows represent groundwater flow. Black lines are equipotential lines. The water table is a black dashed line. a) Effluent or gaining lake. b) Influent or losing lake. c) Influent or losing lake perched above the water table (black arrows represent leakage). d) Flow-through lake. e) Mixed exchange lake (Winter et al. 1998; Woessner, 2020).

If a lake is in a groundwater flow system in which all groundwater flow is into the lake is an effluent or gaining lake (Figure 11). Then the lake surface is an expression of the water table. In this setting, flow discharging to the lake causes the lake level to rise unless it is balanced by loss of water from the lake by way of direct evaporation, evapotranspiration, or surface-water outflow. Water levels in the lake adjust in response to changes in the lake water budget (Woessner, 2020).

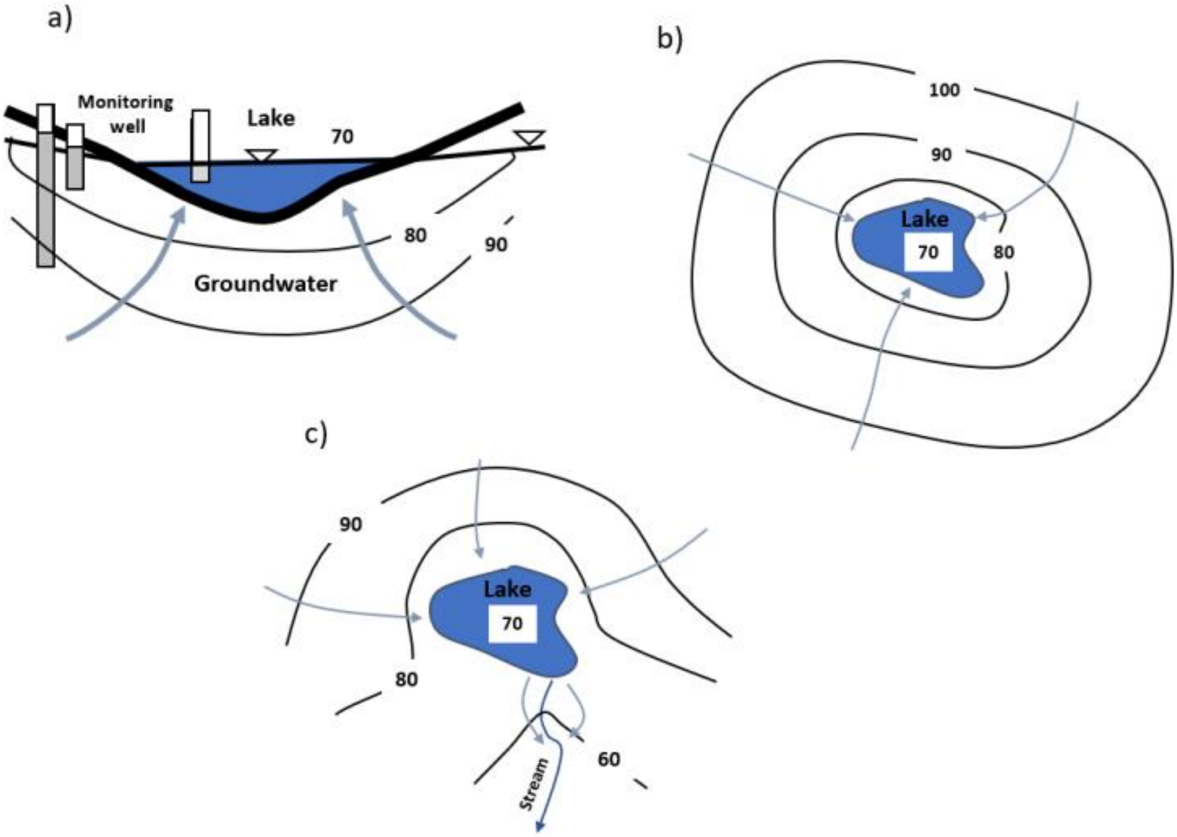

Figure 11. Cross section and map views of effluent (gaining) lake exchange. Equipotential lines and relative head values are shown in black. Groundwater flow is in the direction indicated by blue arrows. Aquifer conditions are assumed to be isotropic and homogeneous. Monitoring wells are open at the bottom. a) Cross sectional representation showing an upward groundwater gradient and groundwater discharging to the lake. Lake stage is shown as a water level on the vertical rectangle. b) A map view showing equipotential lines and groundwater flow converging at the lake. c) A map showing an effluent lake that has a stream discharge. Some groundwater may flow from the lake to the stream under these conditions (Woessner, 2020).

Flow-through lakes occur when the water table is higher on one side of the lake than the other, creating a gradient for groundwater to enter and leave the lake (Figure 12). In some settings these lakes have no surface-water outlet or inlet. The lake surface represents the elevation of the local water table (Woessner, 2020).

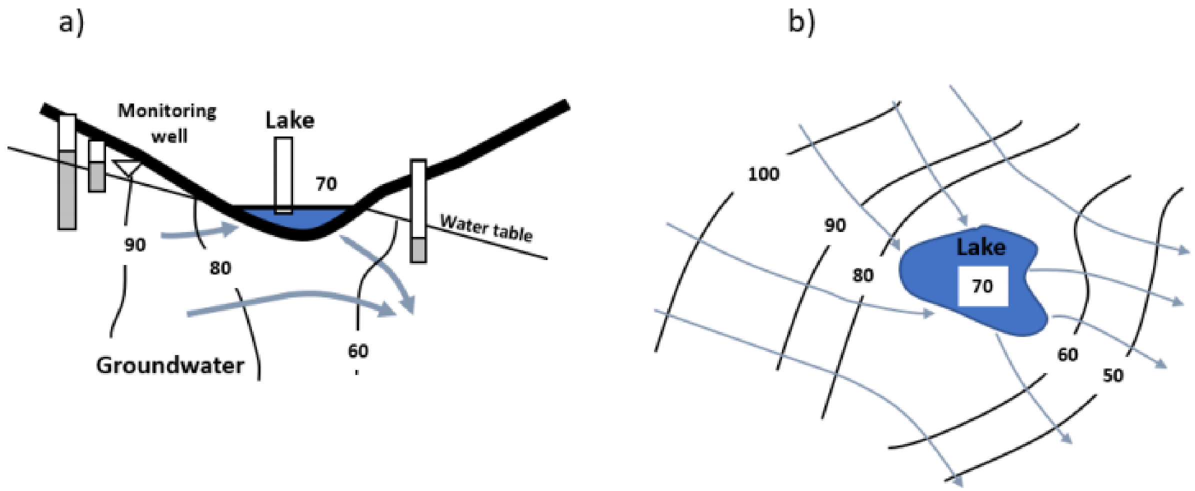

Figure 12. Cross section and map view of flow-through lake exchange. Equipotential lines and relative head values are shown in black. Groundwater flow is in the direction indicated by blue arrows. Aquifer conditions are assumed to be isotropic and homogeneous. Monitoring wells are open at the bottom. a) Cross sectional representation showing an upward groundwater gradient at the up-gradient side (left) and a downward gradient as lake water flows into the groundwater system (right). Lake stage is shown as a water level on the vertical rectangle. b) A map view showing equipotential lines and groundwater flow converging at the lake at the up-gradient side and diverging from the lake on the downgradient side (Woessner, 2020).

A mixed exchange lake suggests that the lake system is dominated by groundwater flowing into the lake, however, lake water flows through the bottom into the underlying groundwater system. Mixed exchange lakes usually occur where variations in lake bottom sediment properties and the presence of lower head values in earth material result in the loss of water from the lake. The mixed term is used here to suggest that exchange directions at the lake perimeter and lake bottom can be different (Figure 13). This condition is presented here to alert investigators to consider the possibility of complex exchanges in some settings (Woessner, 2020).

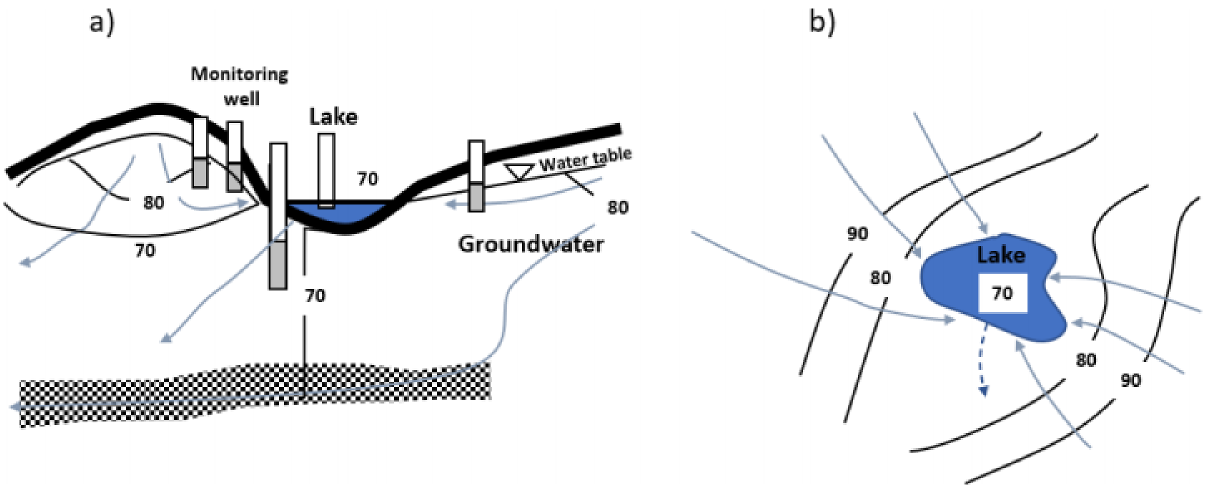

Figure 13. Cross section and map view of a mixed lake exchange. Equipotential lines and relative head values are shown in black. Groundwater flow is in the direction indicated by blue arrows. Aquifer conditions are assumed to be isotropic and homogeneous. The stippled pattern represents a zone of higher hydraulic conductivity. Monitoring wells are open at the bottom. a) Cross sectional representation showing an upward groundwater gradient near the shore (effluent conditions), and a downward gradient beneath the lake. This causes leakage from the lake bottom in this setting. Lake stage is shown as a water level on the vertical rectangle. b) A map view showing equipotential lines and groundwater flow for a mixed lake exchange. The dashed arrow represents the loss of water from the lake bottom to the underlying groundwater flow system (Woessner, 2020).

5.4. Risk assessment

The data included into further development of the conceptual model should provide more information about qualitative description of impacts on the research area. First, land use and potential stress factors and risks should be identified (agriculture, industry, infrastructure - e.g., amelioration, water abstraction points etc.). All factors must be related with the aim of the conceptual model. Also, receptors must be identified, in this case it will be a lake ecosystem, but in other cases could be wetland, well field etc. The results of this step can be shown as maps, providing (1) distribution of different types of land use, (2) distribution of different anthropogenic impacts and (3) distribution of different receptors (European Commission, 2010).

To describe quantitative impacts on the area of conceptual model it is advised to assess three main categories. First, identify existing and potential emissions of anthropogenic sources (e.g., agriculture and N surplus, industry, mining etc.). Second, identify the inputs to groundwater by anthropogenic sources (e.g., agriculture - TN or nitrates, TP or phosphates, pesticides, industry - BTEX, heavy metals etc.). The results of this step can be shown as maps and diagrams, providing: (1) the delineation of areas and receptors affected the reconstruction of the impacts from past events until today, (2) first predictions of the future impacts (European Commission, 2010).

Finally, all data and understanding need to be combined to interpret the system and improve the conceptual model. Groundwater quality data may reveal seasonality of the system or represent travel times of pollutants. With the help of time series analysis, the effects of existing measures can be described by estimations of travel times in the unsaturated and saturated zone and by delineating the impact on the kinetics of degradation and attenuation processes. The impact of measures addressing temporal and spatial development of past anthropogenic inputs can be described. While assessment of groundwater and surface water level time series may represent the surface-groundwater connection intensity, show impacts of water abstraction, or even allow to assess climate change impacts (hydrological droughts) (European Commission, 2010).

VI PROCEDURE FOR HEALTH ASSESSMENT OF GROUNDWATER DEPENDENT LAKE

6.1. Lake ecosystem health assessment procedure

Lake ecosystem health can be assessed by using several indicators:

- Hydrological and hydromorphological indicators, e.g., water residence time, water level changes, land-use etc.
- Physical and chemical indicators, e.g., concentration of nutrients in water and sediments, oxygen conditions, water transparency etc.
- Biological indicators, e.g., characterizing biodiversity or biomasses of fish, benthic invertebrates, phytobenthos, phyto- and zooplankton, macrophytes and other indices.

Many indicators can be applied to specific lake types, for example, Secchi depth is not used as an indicator of primary production in brown-water lakes. Lake types are also important for the ecological assessment of water quality according to the WFD requirements, because boundary values of, e.g., nutrient concentrations or biological indices are lake type-dependent. As the ecological quality assessment is an important part of the lake health assessment, we provide information on lake typology in Lithuania and Latvia.

The official **Lithuanian lake typology** separates lakes into three types with respect to their mean depth, as other criteria, such as altitude and geology, are very similar among most lakes (TAR, 2016-08-09, Nr. 21813).

Table 39. Lake typology in Lithuania (TAR, 2016-08-09, Nr. 21813).

Factors	**Lake type**			
	1 (Polymictic lakes)		2 (Stratified lakes)	3 (Deep stratified lakes)
Mean depth	≤ 3	> 3	> 3	n*
Max depth	n*	< 11	11-30	>30
Altitude (m)	< 200			
Geology	Alkalinity (>1.0 meq/lg (Ca >15 mg/l))			
Surface (km^2)	> 0.5			

n* - the parameter is not used.

The official **Latvian lake typology** separates lakes into eleven types with respect to their mean depth, mineralization, and watercolor (Table 40).

Table 40. Lake typology in Latvia (Cabinet Regulation No 858, 2004).

Type code	Mean depth	Water hardness	Color	Lake type
L1	Very shallow (< 2 m)	Hard-water (> 165 μS/cm)	Oligohumic (< 80 Pt-Co)	Very shallow clear-water lake with high mineralisation
L2	Very shallow (< 2 m)	Hard-water (> 165 μS/cm)	Polyhumic (> 80 Pt-Co)	Very shallow brown-water lake with high mineralisation
L3	Very shallow (< 2 m)	Soft-water (< 165 mkS/cm)	Oligohumic (< 80 Pt-Co)	Very shallow clear-water lake with low mineralisation
L4	Very shallow (< 2 m)	Soft-water (< 165 μS/cm)	Polyhumic (> 80 Pt-Co)	Very shallow brown-water lake with low mineralisation and pH>5,5
L5	Shallow (2–9 m)	Hard-water (> 165 μS/cm)	Oligohumic (< 80 Pt-Co)	Shallow clear-water lake with high mineralisation
L6	Shallow (2–9 m)	Hard-water (> 165 μS/cm)	Polyhumic (> 80 Pt-Co)	Shallow brown-water lake with high mineralisation
L7	Shallow (2–9 m)	Soft-water (< 165 μS/cm)	Oligohumic (< 80 Pt-Co)	Shallow clear-water lake with low mineralisation
L8	Shallow (2–9 m)	Soft-water (< 165 μS/cm)	Polyhumic (> 80 Pt-Co)	Shallow brown-water lake with low mineralisation and pH>5,5
L9	Deep (> 9 m)	Hard-water (> 165 μS/cm)	Oligohumic (< 80 Pt-Co)	Deep clear-water lake with high mineralisation
L10	Deep (> 9 m)	Soft-water (< 165 μS/cm)	Oligohumic (< 80 Pt-Co)	Deep clear-water lake with low mineralisation
L11	Very shallow (< 2 m) and shallow (2–9 m)	Soft-water (< 165 μS/cm)	Polyhumic (> 80 Pt-Co)	Very shallow and shallow brown-water lakes with low mineralisation and pH<5,5.

EU member states should assess the ecological quality of their lake and river waterbodies as required by the WFD. The term “ecological quality” can be considered as a part of the ecosystem health assessment. Ecological quality is primarily assessed according to the biological quality elements. Physical, chemical and hydromorphological indicators are regarded as supportive elements. The principal assessment scheme is presented in Figure 14.

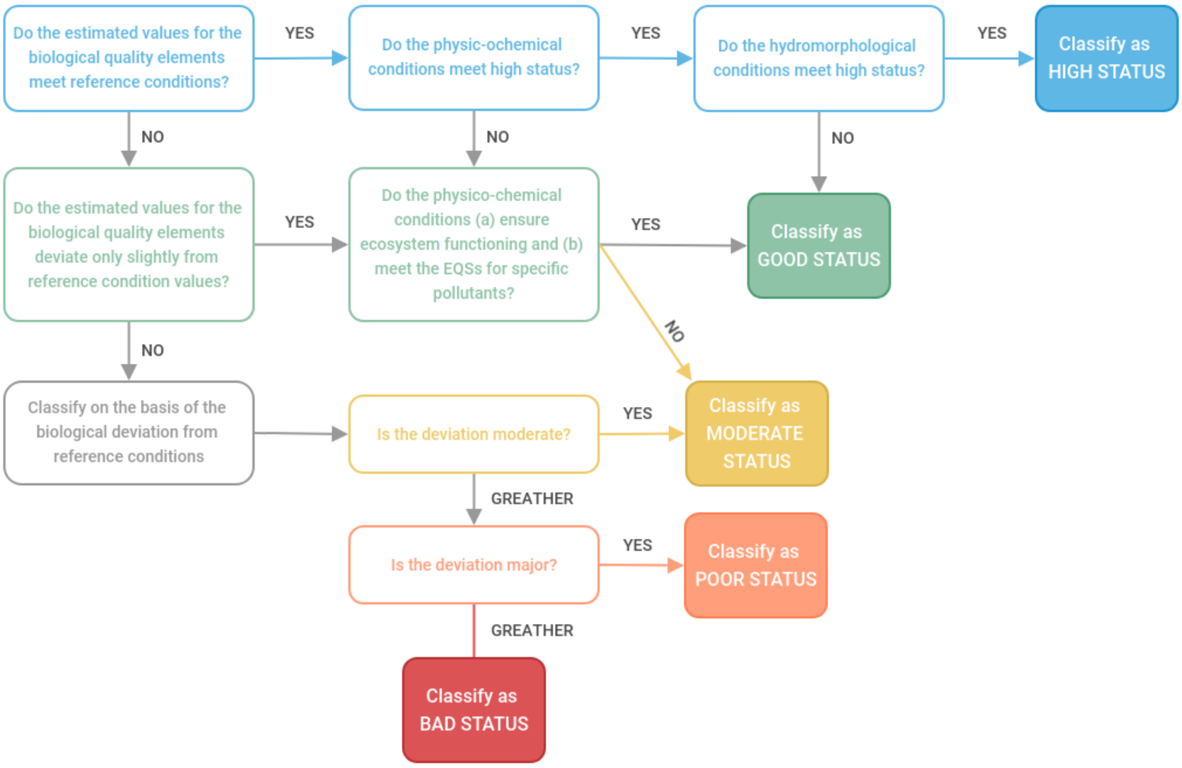

Figure 14. Indication of the relative roles of biological, hydromorphological and physicochemical quality elements in ecological status classification according to the normative definitions in the WFD Annex V:1.2. (after REFCOND, 2003)

Within this TRANSWAT project we propose following lake ecosystem health assessment procedure:

1. to analyse lake ecosystem structure and gathering information on hydrological, hydromorphological, physical, chemical and biological parameters described above in previous chapters.
2. to calculate quality assessment indicators.
3. to establish a conceptual diagram based on both assessment indicators ecological quality and other observed indicators, discuss possible thresholds.
4. to assess lake ecosystem health according to the established conceptual diagram. Expert judgement might be needed as for several elements there are no threshold values set. Comparison of lake ecosystem health assessment to the ecological status classification according to the existing requirements of the WFD.

6.2. Groundwater assessment as potential pressure on surface waters

GWB status from the view of GAAE is evaluated as good if the surface waters within it are evaluated as having a good status. A significant contribution implies that more than 50% of the contamination of analyzed surface waters comes from groundwater. If the GWDE is not significantly damaged or if the damage is low to moderate, then the status of GWB is also evaluated as good. If the GAAE is damaged then it is assessed if groundwater might be responsible for such deterioration (Brkić et al., 2019).

Considering that there are no nationally binding TVs for GAAEs and there is often missing information (especially on the proportion of groundwater contribution) we propose additional steps prior making costly investments into new monitoring stations. **First,** if GAAE is known to be damaged, then it should be assessed if any other (groundwater not related) factors are not to be blamed, i.e., amelioration of wastewater discharge. **Second**, if the factors responsible for GAAE damage are unknown they and investigation should be carried out to rule out surface impact. Meanwhile, desk studies for assessing potential impact from groundwater is encouraged, such as assessment of nearby water abstraction intensity or data from national groundwater monitoring networks (all so called direct data) and evaluation of indirect data (land use analysis in watershed areas, identification of polluting sites. All mentioned data at some extents are already present at country levels. **Finally**, only if groundwater is to be blamed for deterioration of GAAE status or it is assumed so by expert, then field investigations are mandatory. Proposed groundwater quantitative assessment and qualitative assessment procedures are presented in Figure 15 and 16.

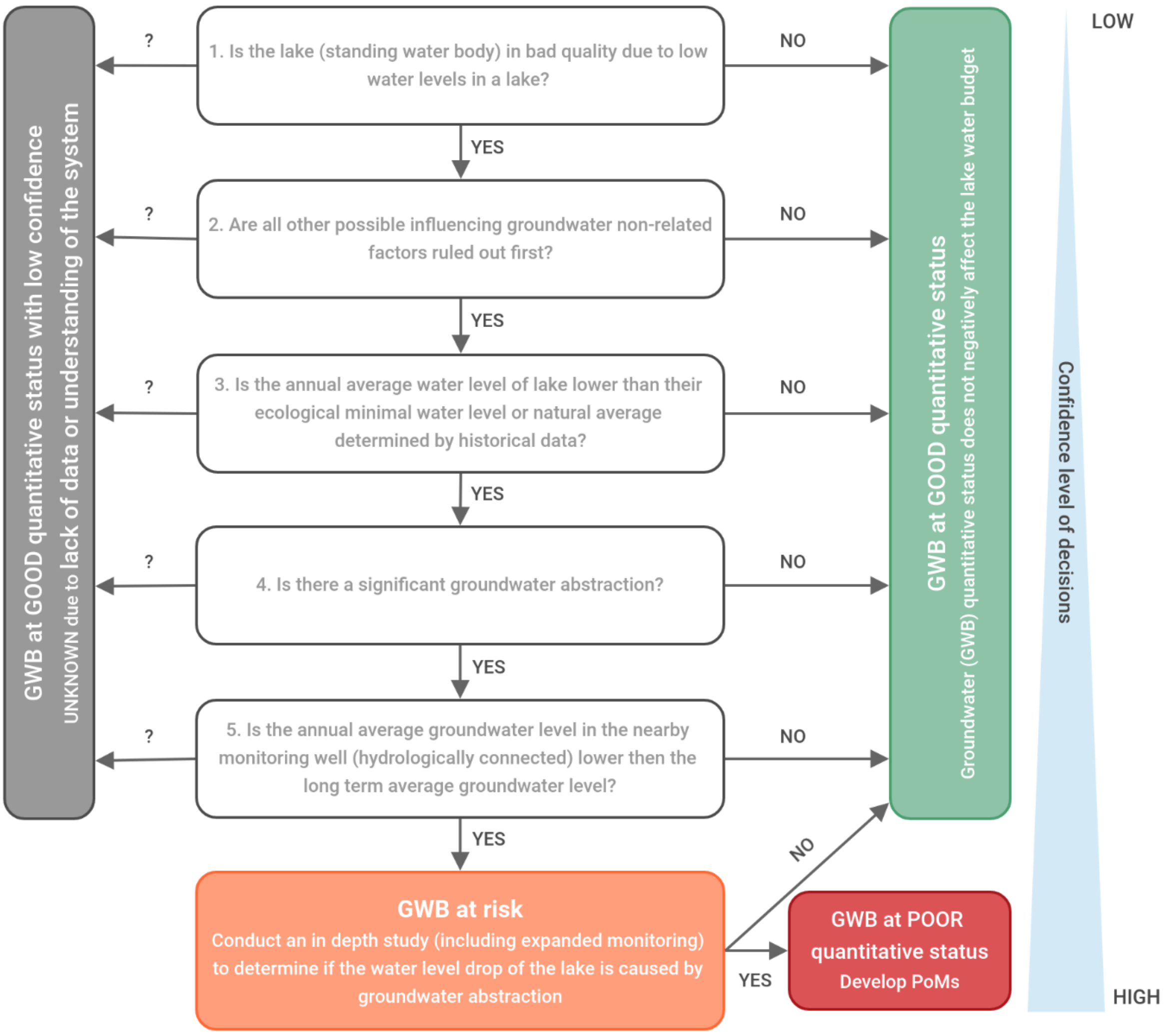

Figure 15. Procedure for the quantitative status assessment of groundwater (GWB) due to potential negative pressures on GAAE.

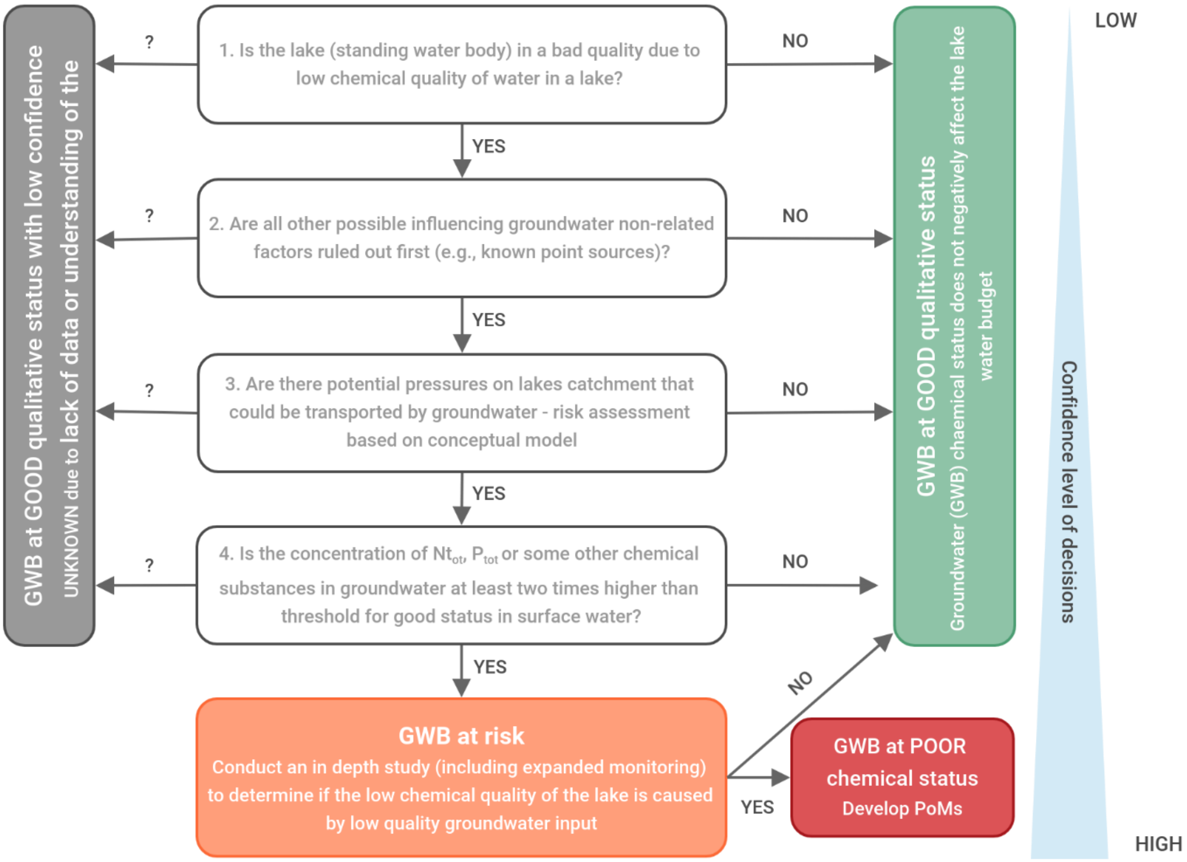

Figure 16. Procedure for the qualitative (chemical) status assessment of groundwater (GWB) due to potential negative pressures on GAAE.

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