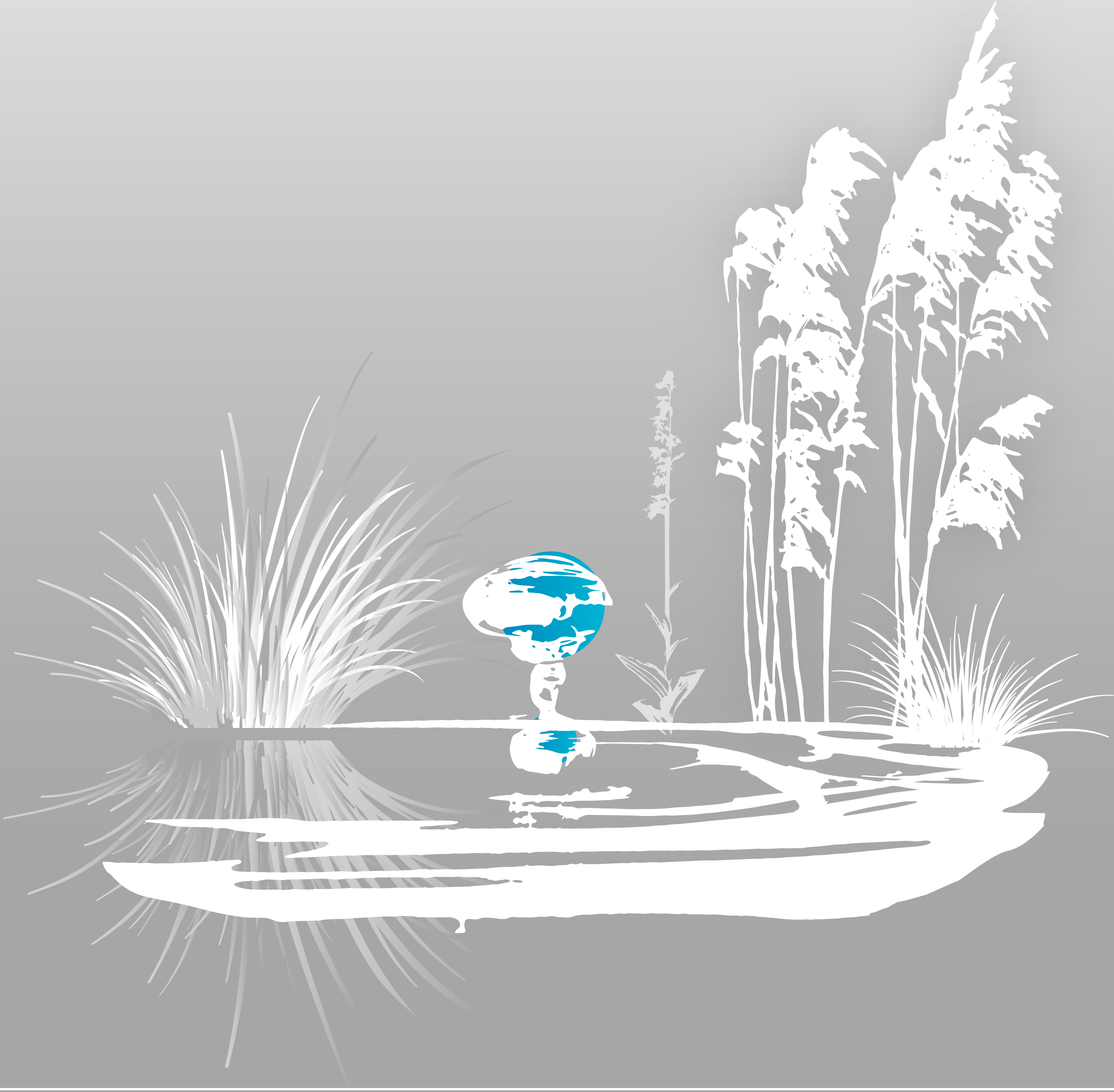
Handbook

Groundwater dependent ecosystems: conceptual understanding, threats and mitigation possibilities



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Book quotation example: Priede A., Strazdiņa L. (eds.) 2022. Groundwater dependent ecosystems: conceptual understanding, threats, and mitigation possibilities. Nature Conservation Agency, Latvia, Sigulda.

Chapter quotation example: Lode E. 2022. Groundwater on the surface. In: Priede A., Strazdiņa L. (eds.) Groundwater dependent ecosystems: conceptual understanding, threats, and mitigation possibilities. Nature Conservation Agency, Sigulda, Latvia, 22–27.

The book was prepared with the financial support of the Interreg V-A Estonia-Latvia cross-border cooperation programme 2014–2020 within the project "Joint actions for more efficient management of common groundwater resources" (Est-Lat 155, WaterAct).

The book is available electronically on the websites www.daba.gov.lv, https://videscentrs.lvgmc.lv/

ISBN 978-9934-8905-4-3

Groundwater dependent ecosystems: conceptual understanding, threats, and mitigation possibilities

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Sigulda, Latvia
2022

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Preface

So, still sauntering on, to the spring under the willows–musical as soft clinking glasses-pouring a sizeable stream, thick as my neck, pure and clear, out from its vent where the bank arches over like a great brown shaggy eyebrow or mouth-roof—gurgling, gurgling ceaselessly–meaning, saying something, of course (if one could only translate it)–always gurgling there, the whole year through–never giving out–oceans of mint, blackberries in summer–choice of light and shade–just the place for my July sun-baths and water-baths too–but mainly the inimitable soft sound-gurgles of it, as I sit there hot afternoons. How they and all grow into me, day after day–everything in keepingvthe wild, just-palpable perfume, and the dappled leaf-shadows, and all the natural-medicinal, elemental-moral influences of the spot.

Excerpt from Walt Whitman's prose poetry To the Spring and Brook
(from Complete Prose Works of Walt Whitman (1892))

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Water has always been a muse, a shelter or an object for artists, poets and their admirers. Whether in the form of a water body in landscape design, a rain cloud or an everyday water bottle, water is instinctively recognized as a key element in human life. Therefore, as soon as water quality deteriorates, it awakens social responsibility and the need for action. For conservationists, on the other hand, it is not a choice, but a professional duty to recognize and disrupt potential initiators of damage before the water source is affected, rivers are altered or lakes are polluted. It should not be forgotten that not only humanity, but thousands of other organisms, from the smallest microorganisms to centuries-old trees, not only enjoy water, but their existence depends on a continuous access to it.

The issues of access to and sustainable management of clean water, both groundwater and surface water, are very old. At the political level, the Water Framework Directive was set up more than 20 years ago to bring together local problems, create a systematic network of solutions and ensure healthy ecosystems across Europe. Latvia and Estonia, as Member States of the European Union, have followed the terms of the Directive and assessed their water resources, still much remains to be done.

The handbook "Groundwater dependent ecosystems: conceptual understanding, threats, and mitigation options" is a symbiont created by Latvian and Estonian hydrogeologists, biologists, and environmentalists–scientists, experts, and practitioners. Reflecting the latest research results and summarizing decades of knowledge and experience, the handbook unites the willingness to translate groundwater–its physical and chemical nature, the different forms and shapes, the ecosystems it nurtures, and its vulnerability to anthropogenic pressure. It was written with the conviction that by combining varied aspects of groundwater dependent terrestrial and aquatic ecosystems, the material included will improve the expert confidence and the decision-making process whenever a gurgling spring demands attention.

Līga Strazdiņa

Glossary

Annual recording period – one year period for which flow measurements are recorded. The recording period may be continuous or intermittent at intervals during which no data were collected.

Aquifer – a hydraulically continuous body of relatively permeable unconsolidated porous sediments or porous or fissured rocks containing groundwater. It is capable of yielding exploitable quantities of groundwater (International Groundwater Resources Assessment Centre, 2021)

Aquitard – groundwater-filled body of poorly permeable formations, through which still significant volumes of groundwater may move, although at low flow rates (International Groundwater Resources Assessment Centre, 2021).

Aquiclude – groundwater-filled bodies of poorly permeable formations, through which no or almost no flow of groundwater passes (International Groundwater Resources Assessment Centre, 2021).

Artesian well or artesian spring – the groundwater that is exposed to the ground surface under the artesian pressure in the ground.

Artesian pressure – the pressure that forces groundwater to the surface without pumping.

Baseflow discharge – the portion of the streamflow that is sustained between precipitation events, fed to streams by delayed pathways (it is also called drought flow, groundwater recession flow, low flow, low-water flow discharge, sustained or fair-weather runoff). It should not be confused with groundwater flow (Kendall & McDonnell 1998).

Birds Directive – the Directive 2009/147/EC. The Birds Directive aims to protect all of the 500 wild bird species naturally occurring in the European Union.

Cave springs – the groundwater emerge within a cave and flow into the surrounding landscape.

Closed basin lake – a natural lake without outlets and from which water leaves primarily through evaporation.

Cone of groundwater depression – it occurs in an aquifer when groundwater is pumped from a well. In an unconfined aquifer, this is an actual depression of the groundwater level. In confined aquifers, the cone of depression is a reduction of the groundwater pressure head in the surrounding of the pumping test well.

Ephemeral stream – a stream or part of a stream, which flows only in direct response to precipitation. Such flow is usually of short duration. Most of the dry washes of the region may be classified as ephemeral streams.

Equipotential line – lines of equal groundwater pressure.

Groundwater – water that exists underground in saturated zones beneath the land surface. The upper surface of the saturated zone is called the GW water table. It fills the pores and fractures in underground materials such as sand, gravel, and other rock, much the same way that water fills a sponge.

Groundwater aquifer – a body of rock and/or sediment that holds groundwater.

Groundwater aquitard – any geological formation of a rather semipervious nature that transmits water at slower rates than an aquifer.

Groundwater baseflow – the groundwater that is discharged to streams, and it integrates groundwater from multiple flow paths of varying scales. Also, it is called drought flow, groundwater recession flow, low flow, low-water flow, low-water discharge and sustained or fair-weather runoff.

Groundwater basin – a general term used to define a groundwater flow system that has defined boundaries (water divide) and may include permeable materials that are capable of storing or furnishing a significant water supply. The groundwater basin includes both the surface area and the permeable materials beneath it.

Groundwater body (GWB) – see Water Body.

Groundwater dependent ecosystem – ecosystems are considered groundwater dependent when the whole or major part of their water demand is supplied by groundwater (GW).

Groundwater Directive – Directive 2006/118/EC of the European Parliament and of the Council of 12 December 2006 on the protection of groundwater against pollution and deterioration.

Groundwater discharge area – a place on the surface where the groundwater comes out to the surface through springs or seeps, it is also referred to as *groundwater discharge*. Often groundwater discharges into wetlands, streams and rivers.

Groundwater flow – part of streamflow that has infiltrated the ground, entered the phreatic zone, and has been (or is at a particular time) discharged into a stream channel or springs; and seepage water (Chorley 1978).

Groundwater hydraulic gradient – the driving force that causes groundwater to move in the direction of maximum decreasing total head.

Groundwater level – the depth below the earth's surface that is saturated with water, or the level to which groundwater would rise in a well that is drilled in a confined aquifer.

Groundwater recharge area – area on the surface where the surface water infiltrates into the soil, thus feeding the groundwater unsaturated or saturated zones.

Groundwater table – upper boundary of an unconfined groundwater body, at which the water pressure is equal to atmospheric pressure

Habitats Directive – adopted in 1992, the Council Directive 92/43/EEC, ensures the conservation of over 1000 wide range of rare, threatened or endemic animal and plant species, as well as 200 habitat types.

Helocrene spring – emerges from low gradient wetlands; often indistinct or multiple sources seeping from shallow, unconfined aquifers.

Hillslope spring – occurs where the groundwater emerges on gently to relatively steeply sloping (15–60°) land, emerging from both confined and unconfined groundwater aquifers.

Hydraulic conductivity – a property of porous media to transmit a fluid (usually water) through pore spaces or fractures.

Hydraulic head (or piezometric head) – a specific measurement of groundwater pressure above the soil surface. It is usually measured as an elevation, expressed in units of length, at the entrance (or bottom) of a piezometer. In an aquifer, it can be calculated from the depth to water in a piezometric well (a specialised water well), and given information of the piezometer's elevation and screen depth. The hydraulic head can similarly be measured in a column of water using a standpipe piezometer by measuring the height of the water surface in the tube relative to a surface. The hydraulic head can be used to determine a hydraulic gradient between two or more points.

Hydrogeologic mapping – a method of gathering and evaluating geological information to create a three-dimensional depiction of the subsurface material in which groundwater occurs.

Hydrograph – a graph showing the rate of flow (discharge) versus time past a specific point in a river, channel, or conduit carrying flow. The rate of flow is typically expressed in cubic meters. It can also refer to a graph showing the volume of water reaching a particular outfall, or location in a sewerage network.

Hydrograph analysis – the most widely used graphical method for analysing different surface runoff components in different types of hydrographs, e.g. unit, flood, annual, long-term etc. hydrographs.

Lacustrine – pertaining to an environment of deposition in lakes, or an area having lakes. Because deposition of sediment in lakes can occur slowly and in relatively calm conditions, organic-rich source rocks can form in lacustrine environments.

Lentic ecosystem – also called the lacustrine ecosystem or the still water ecosystem; it is a part of freshwater ecology or aquatic ecology. The ecosystem entails a body of standing water, ranging from ditches, seeps, ponds, seasonal pools, basin marshes and lakes.

Limnocrene paleosprings – evidence of prehistoric groundwater presence and/or flow exists (e.g. paleotravertine, paleosols, fossil spring-dependent species, etc.), but no evidence of contemporary flow or aquatic, wetland, or riparian vegetation (Stevens et al. 2021).

Limnocrene springs – the main difference between the Rheochrine and Limnocrene is that with the Limnocrene exposed groundwater pools on the surface depression before channelizing into the stream flow.

Long-term period – the term is used in surface hydrology defining the norm of hydrological characteristics, i.e. the average value of the characteristics of the hydrological regime for a long-term period of such duration, with an increase in which the obtained average value does not change significantly. As a possible criterion for the duration of the specified multi-year period, the condition of including in this period an even number of multi-year cycles of change in the value under consideration is taken. In practice, the average value obtained from a series covering 40–60 years of observations is considered to be the norm for hydrological values. In this sense, we can talk about the rate of annual runoff, the rate of opening and freezing of water bodies, the rate of flood start and end dates, the rate of snow cover heights, the amount of water in the snow at the beginning of snow melting, etc. (Chebotarev 1970).

Lotic ecosystem – also called the riverine ecosystem. A lotic ecosystem can be any kind of moving water, such as a run, creek, brook, river, spring, channel or stream. The water in a lotic ecosystem, from source to mouth, must have atmospheric gasses, turbidity, longitudinal temperature gradation and material dissolved in it. Lotic ecosystems have two main zones, rapids and pools. Rapids are the areas where the water is fast enough to keep the bottom clear of materials, while pools are deeper areas of water where the currents are slower and silt builds up. The lotic ecosystem is an important part of freshwater ecology or aquatic ecology.

Madicolous habitats – are the habitats that are characterized by a thin layer of water that frequently flows over rocky surfaces, and for this reason, they are also known as *hygropetric habitats*, i.e. requiring a habitat of a thin layer of water covering a rock surface.

Minerotrophic – refers to environments that receive nutrients primarily through groundwater that flows through mineral-rich soils or rock (e.g. Environment Canada 2014), or surface water flowing over land (e.g. Wang et al. 2019)

Ombrotrophic – refers to soils or vegetation which receive all of their water and nutrients from precipitation, rather than from streams or springs. Such environments are hydrologically isolated from the surrounding landscape, and since rain is acidic and very low in nutrients, they are home to organisms tolerant of acidic, low-nutrient environments. The vegetation of ombrotrophic peatlands is often bog, dominated by *Sphagnum* mosses. The hydrology of these environments are directly related to their climate, as precipitation is the water and nutrient source, and temperatures dictate how quickly water evaporates from these systems (e.g. Davies et al. 2018).

Paludification – the process characterized by peat initialization on previously drier and vegetated habitats over inorganic soils, with no fully aquatic phase (e.g. Vitt 2006).

Palustrine (wetlands) – relating to a system of inland, nontidal wetlands characterized by the presence of trees, shrubs, and emergent vegetation (vegetation that is rooted below water but grows above the surface). Palustrine wetlands range from permanently saturated or flooded land (as in marshes, swamps, and lakeshores) to land that is wet only seasonally (as in vernal pools).

Perennial stream – a stream or part of a stream where the water flows continuously around the year as a result of groundwater discharge or surface runoff.

Phreatophytic ecosystems – formed by the deep-rooted plants that obtain a significant portion of the water from the phreatic zone (zone of saturation) or the capillary fringe above the phreatic zone. They can usually be found along streams where there is a steady flow of surface or groundwater in areas where the water table is near the surface.

Potentiometric groundwater surface – a synonym of "piezometric surface" which is an imaginary groundwater surface that defines the level to which water in a confined aquifer would rise were it completely pierced with wells.

Rheocrene springs or flowing springs – emerges into one or more stream channels, without pooling in the surface depression. Springs fed by the unconfined groundwater aquifers, located on the hillside, forming vegetation poor spring area from which the water flows away quickly.

Saturated hydraulic conductivity – a quantitative measure of a saturated soil's ability to transmit water when subjected to a hydraulic gradient. It can be thought of as the ease with which pores of a saturated soil permit water movement.

Seep or flush – a moist or wet place where water, usually groundwater, reaches the earth's surface from an underground aquifer (The Wildlife Trusts 2018).

Semi-lotic fountain spring – an artesian upwelling of groundwater in a fracture or tubular geologic structural setting which forces the flow to rise higher than the surrounding landscape.

Spring – a point at which water flows from an aquifer to the Earth's surface. It is a component of the hydrosphere.

Steady-state hydrologic budget – equilibrium condition for the water balance, where inflows (I) and outflows (O) are balanced, i.e. $I = O$, and changes in storage $\Delta S = 0$. Often, groundwater systems are considered to be at a steady state if inflows and outflows balance over a yearly or decadal timescale.

Streamflow hydrographs – see Hydrograph.

Subterranean ecosystems – ecosystems of especially fragile groundwater-dependent cave habitats. In caves of the temperate climate zones, where, besides eternal darkness and high humidity, especially constant low temperatures and a scarce food base are exerting a great influence on the population of the subterranean habitat.

Surface water (SW) – a water located on the Earth's surface, forming above ground water bodies, including streams, rivers, lakes, wetlands, reservoirs, and creeks. The ocean, despite being saltwater, is also considered surface water. In water management concepts the SW may also be referred to as blue water.

Surface water balance or water balance (SWBc) – the cyclical movement of water between the atmosphere and the ground surface, considering precipitation, evaporation, and runoff. It described by the partitioning of precipitation (*P*) into runoff (*Q*) and evapotranspiration (*ET*) (see also *Water balance equation*)

Surface water basin, also a drainage basin – any area of land where precipitation collects and drains off into a common outlet, such as river, bay, or other body of water. The drainage basin includes all the surface water from rain, snowmelt, hail, sleet and nearby streams that run downslope towards the shared outlet, as well as the groundwater underneath the earth's surface.

Surface water body (SWB) – see Water body.

Throughflow – the sporadic horizontal flow of water within the soil layer. It normally takes place when the soil is completely saturated with water. This water then flows underground until it reaches a river, lake, or wetland. Rates of water movement via throughflow are usually low. Rates of maximum flow occur on steep slopes and in previous sediments. The lowest rates of flow occur in soils composed of heavy clays. Rates of throughflow in these sediments can be less than 1 mm per day.

Travertine – a sedimentary rock formed by the chemical precipitation of calcium carbonate minerals from freshwater, typically in springs, rivers, and lakes; that is, from surface and groundwaters. Travertine is considered sensu stricto to be formed in hot springs and to be less porous than tufa (e.g. Blatt et al. 1980; Allaby 2013; Leeder 2011).

Tufa – a variety of limestone formed when carbonate minerals precipitate out of ambient temperature water. Geothermally heated hot springs sometimes produce similar (but less porous) carbonate deposits, which are known as *travertine*. Tufa is sometimes referred to as (meteogene) travertine.

Unsaturated groundwater zone – the portion of the subsurface above the groundwater table. The soil and rock in this zone contain air as well as water in its pores. Hydrologically, the unsaturated zone is often the main factor controlling water movement from the land surface to the aquifer.

Vienna Standard mean ocean water (VSMOW) – a sea-water sample that comprises the international standard for D/H and $^{18}O/^{16}O$ ratios. Differences in isotopic composition are expressed as parts per mille deviations from the isotopic composition of this standard. The D/H ratio is the ratio between deuterium (heavy hydrogen, ^{2}H) and hydrogen (^{1}H) in natural waters and other fluids, and in water combined in hydrous minerals. This ratio yields information about the origin and geologic history of the fluid, and about fluid/rock interactions. The $^{18}O/^{16}O$ ratio is an abundance ratio between two of the three isotopes of oxygen. They have similar chemical properties because they have the same electronic structure, but because of the differences in mass between their nuclei they have different vibrational frequencies which cause them to behave slightly differently in physical-chemical reactions. These differences can provide information, e.g. regarding the source of water in a past environment or the temperature at which various interactions have taken place.

Water balance equation – presenting elementary parts of water balance: precipitation (P), evapotranspiration (ET), and runoff (Q), expressed as volume of water.

Water body (WB) – according to Water Framework Directive, the WB means a discrete and significant element of lakes, reservoirs, rivers or canals, a transitional water or a stretch of coastal water, and a distinct volume of GW within aquifer or aquifers.

Water Framework Directive (WFD) – Directive 2000/60/EC of the European Parliament and of the Council of 23 October 2000 establishing a framework for Community action in the field of water policy.

Vadose zone – aeration zone above the water table in the ground. It varies widely in thickness, depending on the type of earth material present. Water within this zone, which is moving downward under the influence of gravity, is called vadose water, or gravitational water.

Abbreviations

EC – European Commission
ES – ecosystem services
EU – European Union
GAAE – groundwater associated aquatic ecosystem
GDAE – groundwater dependent aquatic ecosystem
GDE – groundwater dependent ecosystem
GDTE – groundwater dependent terrestrial ecosystem
GW – groundwater
GWB [1] – groundwater body
SMOW – Standard Mean Ocean Water
SWBc – surface water balance
SW – surface water
SWB1 – surface water water-body
WB – water body
WFD – Water Framework Directive
VT – threshold values

[1] The abbreviation to the "water body" i.e., GWB or SWB is derived from the "body of water" definition in the Water Framework Directive (WFD). To understand it more clearly: according to the WFD one terrestrial water body, i.e. river, lake, could be divided into several surface water management water bodies, i.e. SWB in this guideline

1. Concept of groundwater dependent ecosystems

1.1. The concept of groundwater dependent ecosystems

Agnese Priede (Nature Conservation Agency, Latvia), Elve Lode (Tallinn University)

Ecosystems are considered groundwater dependent if the whole or major part of their water demand is supplied by groundwater (GW). There are six types of ecosystems known worldwide as groundwater dependent ecosystems (GDEs): springs, wetlands, rivers, lakes, phreatophytic and subterranean ecosystems; they are all a function of their hydrological, geological, and climatic setting, whereas the springs are the only ecosystems that are fully associated with GW alone. The water supply of other GDEs partly may be ensured also by the surface water (SW) and precipitation (e.g. Brown et al. 2010). In disturbed or altered GW conditions the GW-fed ecosystem functions are damaged, impairing their structure and functions (Orellana et al. 2012). Thus, GDEs can be divided into ecosystems living in underground GW aquifers and ecosystems that depend on GW exposing to the surface, such as spring-fed or GW-fed lakes and rivers, wetlands, swamp woods, humid dune slacks (Eamus et al. 2006; Orellana et al. 2012; Retike et al. 2020). However, in some cases, GW-fed wetlands, swamp woods and dune slacks can be considered as GW-dependent terrestrial ecosystems (GWDTEs) (EC 2012) and GW-fed lakes and rivers as GW-fed aquatic ecosystems (e.g. Brown et al. 2010). In addition, the supply of GW to GDTEs can be periodic, intermittent, episodic or perennial (Kløve et al. 2011).

As terrestrial ecosystems, fauna and flora, as well as the human population rely heavily on GW, there is a need for adequate protection and management of GW-fed ecosystems and the services they provide. In the European Union (EU), the Water Framework Directive (WFD) 2000/60/EC provides a framework for water management and protection of all types of water resources, including GW. It has been in force since 2000 and has been integrated into national law (EU 2000). According to the WFD, at least good status of water body (WB) must be achieved through a good biological community status, WB hydrology and water quality, while the distinct volume of the groundwater body (GWB)[2] should be defined via certain GW flow criteria, sufficient amount of GW to serve the needs of people, and limited water abstraction.

The concept of ecosystems directly depending on GW incorporated in the WFD needs to be understood in a broader sense, as the status of biological communities needs to be assessed. These communities may present the status of both the aquatic and terrestrial ecosystems. Although there are many ecosystems that are only partially fed by GW and their quality and structural diversity can be significantly improved by GW supply, the EU guidelines (EC 2011a) focus only on those ecosystems that are highly dependent on GW supply, both for the whole year or for a significant part of the year.

The WFD indicates a clear relationship between GWB and GDEs. The GWBs defined by each EU Member State, as a GW management unit, may have an impact on the ecological quality of GW-fed surface WBs. In terms of WFD, the whole GWB is considered to be of poor quality status if

[2] The term "groundwater body" is defined here as a distinct volume of groundwater within an aquifer or aquifers, whereas the determination of an "aquifer" requires two criteria to be considered: a) significant groundwater flow (e.g. >10 m^3 per day as an average, or sufficient to serve the needs of 50 people), and b) the volume of groundwater abstraction that could cause significant diminution of ecological quality of groundwater dependent surface water body or terrestrial wetland ecosystem.

anthropogenic pressure on the GW causes significant damage to GDEs. Poor quality of GDEs can be caused by anthropogenic changes in the quality and quantity of GW. This means that, with regard to the WFD and the mandatory reporting by EU Member States, the status assessment of GDEs is part of the assessment of GWBs, including the identification of GDEs and their quality assessment. If signs of GDEs deterioration are found, rehabilitation measures should be planned and implemented to improve the condition of the GWB and thus restore the quality of the damaged GDEs.

1.2. Links between EU Water Framework Directive and nature directives

Agnese Priede (Nature Conservation Agency, Latvia)

The EU WFD has established a framework for sustainable water management and the protection of surface and groundwater WBs throughout the European Union (EU), while the so-called nature directives – the Habitats Directive 92/43/EEC[3] and the Birds Directive4[4] – provide a framework for implementation of biodiversity conservation requirements. Both the nature directives and the WFD aim to ensure healthy ecosystems, both the surface water and groundwater dependent terrestrial ecosystems (EC 2011b). All of the above mentioned directives focus on a healthy environment, and therefore measures implemented under the nature directives or the WFD can benefit the overall health of the ecosystem, ecosystem functionality, maintenance of ecosystem services, and biodiversity.

Although there are synergies between all three directives, they use different terms and are usually implemented at national level by different authorities using different instruments. Therefore, in practice, the need for integrated management is still poorly understood and, moreover, weakly linked to the implementation of specific actions. In fact, cooperation between sectoral bodies working with the WFD and nature conservation is often insufficient and can be limited to a general, often formal reference to each other's legislative acts and guidance documents. However, it is clear that a more integrated approach and closer cooperation, including funding sources for concrete management activities, planning and monitoring would benefit both sectors, which would have a positive impact on the quality of the environment and human well-being.

The concept of GDEs in the context of the WFD needs to be understood more specifically, focusing on the assessment of GWB status in their water management conditions. There is no definition in the WFD and suggested guideline approaches for assessment of GDEs status within the SWB. At the same time, GDEs may overlap spatially or partially with habitat types protected under the Habitats Directive and may include species protected under the Habitats Directive. They can occur both inside and outside the Natura 2000, the network of protected nature areas of European importance (EC 2011b).

[3] Council Directive 92/43/EEC of 21 May 1992 on the conservation of natural habitats and of wild fauna and flora.
[4] Directive 2009/147/EC of the European Parliament and of the Council of 30 November 2009 on the conservation of wild birds.

Terms "ecosystem" and "habitat" in the EU directives

The WFD includes the concept of ecological status as a measure for reflection of SWB conditions. It is largely evaluated by analysing the structure of biological communities, including habitat, physicochemical elements and the functionality of ecosystems. Measuring ecological status makes it possible to evaluate the impact of human activities on aquatic ecosystems, and is considered an essential tool for the sustainable management and protection of freshwater resources, including GW resources (EU 2000).

The use of terms "ecosystem" and "habitat" in the context of the WFD and the EU nature directives can be confusing, as they partially or sometimes completely overlap and may seem difficult to distinguish. The WFD uses the term "ecosystem" without a strict definition, thus implying a broad interpretation using some reference constituents. The term "habitat" in the WFD is used only in the context of the Habitats Directive.

The purpose of the Habitats Directive is to take the necessary measures to maintain or restore natural habitats and populations of species of wild fauna and flora to a favorable status. Conservation measures target *natural habitats*, which are defined in the Directive as terrestrial or aquatic areas that differ in geographical, abiotic and biotic characteristics, whether completely natural or semi-natural. Annex I of the Habitats Directive lists *habitat types* of Community interest – habitat types which are in danger of extinction in their natural range, have a small natural range due to their regression or their intrinsically restricted area, or present outstanding examples of typical characteristics of one or more of the nine following biogeographical regions in the EU.

However, the directives leave room for interpretation of the difference between the two terms, which is provided in various EU guidance documents but is generally based on ecology.

The concept of WFD ecosystem is broader, including the whole biological and physical entity with certain functionality. Organisms find proper habitats and live in a suitable environment within the ecosystem.

The concept of habitats in the Habitats Directive is narrower, focusing more on the species and environmental conditions necessary for the survival of certain species and communities, with an emphasis on endangered species and their requirements. Nevertheless, a habitat has a favorable conservation status only if it provides suitable conditions for the target species and communities.

1.3. Links with national legislation in Latvia and Estonia and recommendations for the future

Agnese Priede (Nature Conservation Agency, Latvia), Kristiina Ojamäe (Estonian Environment Agency), Elve Lode (Tallinn University)

In Latvia, the requirements of the WFD and the Groundwater Directive have been incorporated into the national Water Management Law and the resulting regulations of the Cabinet of Ministers. Water Management Law aims to establish a protection system for SW and GW that would also prevent the deterioration of GDTEs, but does not emphasize GW-related aquatic ecosystems. Aquatic ecosystems are mentioned only from the SW perspective. The purpose of the regulations of the Cabinet of Ministers No. 92 (17.02.2004) "Requirements for the Monitoring of Surface Water, Groundwater and Protected Areas and the Development of Monitoring Programmes" is to determine the requirements for the monitoring of SW, GW and protected areas and the activities to be carried out if the water body has not achieved the environmental quality objectives. However, none of the laws or regulations provide clear definitions of GDEs, GDTEs and GAAEs, nor is there a legally binding methodology for identifying such ecosystems in Latvia, so all existing regulations allow for a fairly broad interpretation. Moreover, no thresholds have been set to protect such ecosystems as well.

In practice, GDEs in Latvia are not much assessed from the point of view of GW impacts. The first study on GDTEs was initiated during the Interreg Estonia-Latvia project "Joint management of groundwater dependent ecosystems in transboundary Gauja-Koiva river basin" (GroundEco). In 2021, a state-funded project was launched to identify all GDTEs in Latvia in accordance with the GroundEco methodology based on the Estonian experience and to identify and assess the quality of both GDTEs and GAAEs at the national level. At present, the GDEs assessment and monitoring is not included in the national GW monitoring programs, and the GDEs were not considered for the assessment of GWB status in accordance with the requirements of the WFD.

In Estonia, the requirements of the WFD and the Groundwater Directive are incorporated into the national Water Act and into its implementing regulations. One of the general objectives of the Water Act is to prevent the deterioration and improve the condition of aquatic ecosystems, terrestrial ecosystems, including wetlands.

The ecological status of a SWB describes the quality of the structure and functioning of aquatic ecosystems, as well as the physical, chemical and hydromorphological quality indicators that are important for the functioning of aquatic ecosystems. The regulations for assessing SWB are regulated in accordance with the Estonian Cabinet of Ministers Regulations No. 19 and Regulation No. 28. The assessment of proportion of GW supply shall take into account to achieve at least good ecological status of GW dependent SWB, to prevent significant deterioration of SWB ecological status and to prevent significant damage to GDTEs. When determining the status of GWB, the status of groundwater-dependent aquatic and terrestrial ecosystems is taken into account, among other parameters. In accordance with the Estonian Cabinet Regulation No. 48, the chemical status of GWB is assessed as good status if the concentration of pollutants does not impede the achievement of the GWBs environmental objectives and does not cause significant damage to the ecological or chemical status of the SWB or GDTE. The quantitative status of GW is assessed as good if there is no downward trend in GW levels and the intrusion of saline or other waters that would lead to a significant deterioration of GDTEs.

In practice, Estonia has established a list of terrestrial and aquatic ecosystems that are directly dependent on GWBs under the WFDs Section 2.1 of Annex II. Conceptual models are described for 31 GWBs including the GWBs relationship with GW dependent SW and terrestrial ecosystems. From here, the status of GWB was assessed to determine whether GW chemical quality indicators can lead to unfavourable status for SWB or terrestrial ecosystems feeding on GW.

In Estonia, it is currently planned to include the GDEs assessment and monitoring into the national SW and GW monitoring programs in accordance with the river basin management plans. Although there are several objectives for monitoring hydrogeological conditions, the impact of GW chemical and quantitative status and effects of GW abstraction affecting the GDEs in the Cabinet Regulation No. 9, it is still being elaborated. The implementing rules set chemical limit values for both GWB and SWBs, but there is currently no approach to harmonized limit values or a methodology to establish them for GDEs. This topic requires more monitoring data and research.

It should also be emphasised here that the division of springs into lotic (flowing water) and lentic (standing water) ecosystems in Chapter 2.3 of this book is an example of a structured spring or GDE database that can be created in both Latvia and Estonia. All information provided in Chapter 2.3, including generalised diagrams of various spring formations, corresponding detailed descriptions of terrain and fluidum morphology, the ecological significance of these spring ecosystems, and the predominant disturbances were based on the literature (mainly: Springer & Stevens 2009; Stevens et al. 2016a,b; SSI Webpage) reviewed during the compilation of this book.

Therefore, the illustrative photographs provided in Chapter 2.3 should be seen as an attempt to confirm that the establishment of such a system would be possible in Estonia and Latvia. And it is also important to emphasize that all illustrative photographs and corresponding brief comments are found in various literature and references, largely based on descriptions from experts or, in some cases, simply by interested people.

2. Understanding groundwater and groundwater-fed systems

2.1. Groundwater below the ground

Elve Lode (Tallinn University), Agnese Priede (Nature Conservation Agency, Latvia), Krišjānis Valters (Latvian Environment, Geology and Meteorology Centre)

Groundwater is water that exists mainly in subsurface pore spaces, but also in fractures and channels of rock formations. The dissolution of soluble rocks, such as limestone, dolomite and gypsum, forms karstic underground drainage systems with sinkholes and caves. After the polar ice caps, GW is the next largest freshwater reservoir on Earth, containing more than 100 times the volume of streams and freshwater lakes (Shiklomanov 1993). GW plays an important role in the hydrological water cycle, plant growth and soil formation, as well as in providing water for human activities (Brands et al. 2016).

2.1.1. Formation of groundwater below the ground

More than 68% of freshwater is found in polar ice caps and glaciers and is therefore largely unavailable to public use (Shiklomanov 1993). About 30% of the world's freshwater is GW, but only 1.2% is found in rivers and lakes of surface water.

Geological formations that provide significant amounts of water to wells or springs are called *aquifers*. The aquifer consists of two or more water-permeable rock layers in the subsurface, separated by intervening aquitard layers, which may impede groundwater movement locally but not significantly regionally. With the exception of fossil aquifers, GW is replenished mainly by precipitation falling within the recharge area of an aquifer (Figure 2.1.). When infiltrated into the soil, atmospheric precipitation moves either horizontally or vertically within the subsurface. Part of this water remains in the unsaturated or vadose zone (also often referred to as "*soil water*"), while the other part reaches the saturated aquifer zone where the pore spaces are completely filled (Brands et al. 2016).

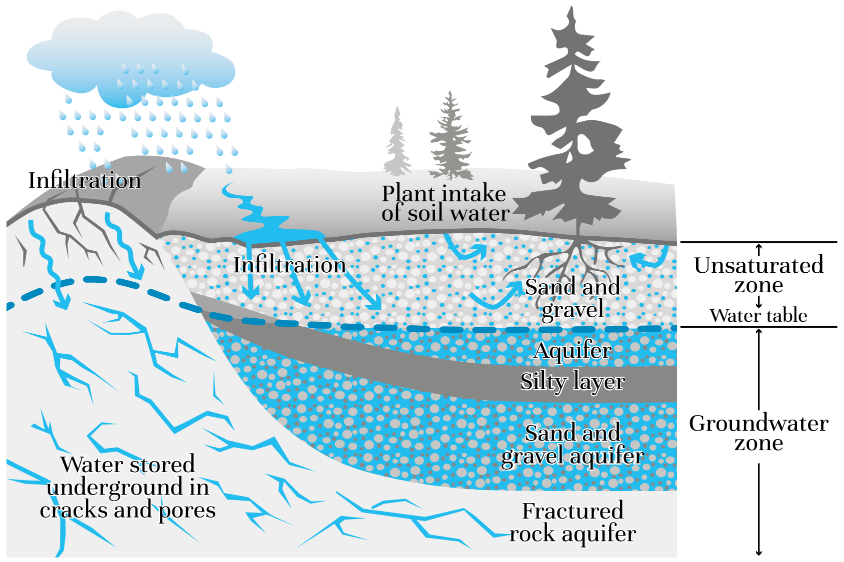


Figure 2.1.
Groundwater in the ground (modified from Natural Resources Canada)

2.1.2. Types of groundwater aquifers

The unsaturated zone of the GW aquifer is located just below the earth's surface, where the soil or rock pores contain both water and air but are not completely *filled* with water. It differs from the zone below which is saturated with water. The top of the saturated zone is called *the GW table* (Winter et al. 1999). Aquifers can be classified according to whether they are *confined* or *unconfined* and according to their geologic composition. *Confined aquifers* exist between two impermeable layers, often composed of clay or clay-derived rock. *Unconfined aquifers* lack a confining layer above the GW table and are therefore vulnerable to pollution from the surface. *Unconsolidated aquifers* consist of sand, gravel and other materials that are not cemented together and where water fills the gaps between the particles. In contrast, water-bearing formations that are cemented together are called *consolidated aquifers*. Relatively large pore spaces in unconsolidated aquifers and solution channels in carbonate aquifers can contain significant amounts of GW with high hydraulic conductivity. In contrast, only the presence of fractures and joints in igneous and metamorphic rocks allows water to enter and move through formations (Brands et al. 2016).

Sand and gravel aquifers are generally unconfined. They are made of alluvial or glacial deposits and are found in valleys, depressions and lowlands. Alluvial aquifers formed over hundreds to thousands of years by water deposition of sand, silt, and clay particles. These alluvial deposits can contain large GW reserves that are quite accessible and easy to withdraw. Sand and gravel aquifers of glacial origin can also have large GW reserves that are relatively easy to withdraw, but the hydraulic conductivity in such aquifers can be quite variable due to the lack of sorting in glacial till (Brands et al. 2016).

Carbonate rock aquifers are primarily limestone or dolostone formed in ancient marine environments. Weak carbonic acid in rainwater dissolves carbonate rock and over thousands of years has formed karst landscapes in many areas, which are now characterized by many solution cavities, caverns and sinkholes through which water can move rapidly. Where carbonate layers are exposed to the surface, landed precipitation and running water in the streams can be infiltrated directly into the GW, thus significantly increasing the GW contamination potential. Sandstone aquifers and igneous and metamorphic rock aquifers typically store and transmit water only along bedding planes, joints, and other cracks and fractures. Hydraulic conductivity in such aquifers is low, but large regional aquifers can yield high total volumes of GW (Brands et al. 2016).

2.1.3. Groundwater flow and well

The hydro-physical properties of the aquifer affect the water storage capacity as well as the hydraulic conductivity or the ability of the aquifer to transmit water. Factors that affect hydraulic conductivity and storage capacity include *porosity* (proportion of pore space) and *permeability* (connectedness) of pore spaces. The GW flow has a down-gradient from high to low pressure, which often corresponds with moving from high to low elevation in the aquifer. An exception occurs in cases of confined aquifers contained within tilted rock formations where considerable pressure builds up in the lower reaches of the aquifer. In such cases, a well or a spring located at a point where the *surface of the GW potentiometric pressure* is higher than the land surface, and GW can upflow freely to the ground surface without the need for pumping; such upflows are termed as *artesian* wells or springs (Brands et al. 2016).

Wells drilled into unconfined or confined aquifers with insufficient pressure to upflow naturally require pumping to bring water to the surface. Some level of *drawdown* (or reduction in the height of the potentiometric pressure surface) occurs with pumping of water from aquifers. The GW pumping forms *a cone of GW depression* (Figure 2.2) around wells, where the shape, size, and depth of the cone of GW depression as well as the rate at which the GW level recovers from the pumping depends upon the pumping rate and duration, aquifer characteristics such as hydraulic conductivity, and the presence or absence of "active" wells nearby. Usually, the pumping tests is one of the most important measures to determine whether the aquifer will provide an adequate yield (measured in litres per minute) for the intended purpose of the GW well installation (Brands et al. 2016).

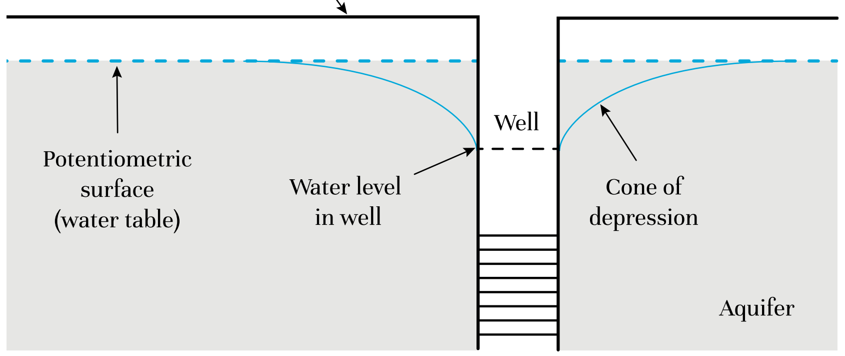


***Figure 2.2.** A schematic representation of the cone of GW depression formed by the GW pumping test in the well (copied from Brands et al. (2016)).*

2.2. Groundwater on the surface

Elve Lode (Tallinn University)

By digging a pit in the ground under normal conditions, it can be seen that the soil is moist. This means that some of the subsurface pore space, also called the *unsaturated GW zone*, is occupied by water, and some of the pore space is occupied by air (Figure 2.1.). By digging down far, we reach the water in the hole, i.e. the pore spaces in the soil are 100% filled with water, the soil is in saturated conditions, and the water level in the borehole now represents the *GW level*, formed by the *GW water surface* of *saturated zone* (Earle 2019).

In general, the GW surface of the upper-most saturated soil zone forms a subdued replica of the surface topography from the hills toward the valleys (Figure 2.3.). The area of land in which the SW infiltrates into the ground is called the *GW recharge* area, and the exposed GW area or point on the land surface is called the *GW discharge* area.

In this case, hydrogeologic mapping is the most attractive tool to create a three-dimensional depiction of GW recharge-discharge flow models (Figure 2.4.). Unfortunately, the preparation of such a map requires quite complex and various measurements in the field, e.g. GW level recordings, measurements of saturated hydraulic conductivity of geologic formations, measurements of GW hydraulic gradients along GW level measurement transects, SW discharges, etc. Thus, the addition of all the GW table configurations and the recharge patterns is largely controlled by spatial and temporal patterns of precipitation and evapotranspiration at ground surface (Freeze & Cherry 1979).

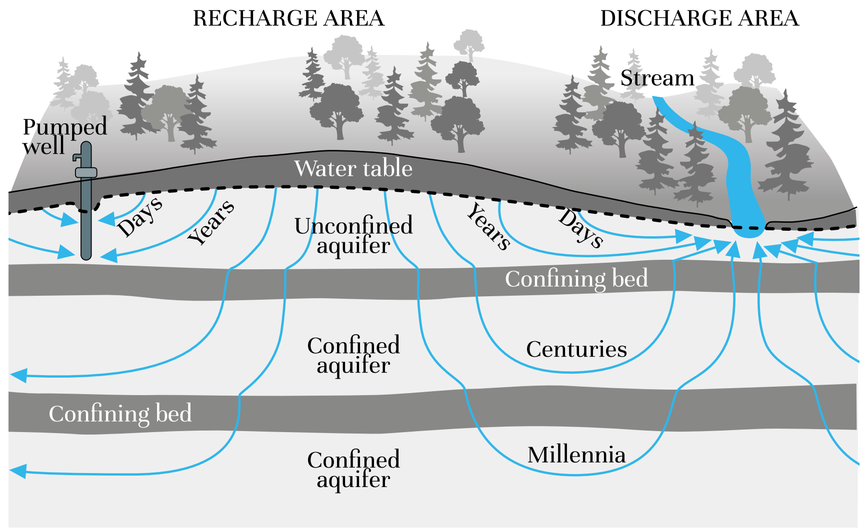


***Figure 2.3.** Conceptual scheme of groundwater recharge and discharge areas on the land surface (modified from Winter et al. (1999)).*

2.2.1. Groundwater recharge-discharge reflections in hydrologic water budget

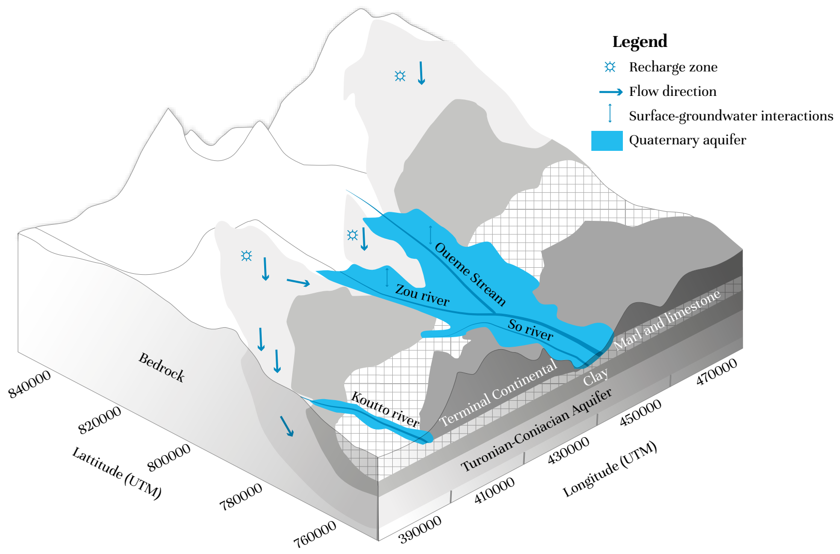


***Figure 2.4.** Example of three-dimensional conceptual model of groundwater recharge-discharge interconnections with the land surface flowing system (scheme is copied from Kpegli et al. 2018).*

The GW recharge-discharge regime has a significant interrelationship with the components of the SW hydrologic cycle. The basic tool for quantifying the various components of the SW hydrologic cycle is the SW *hydrological budget or surface water balance (SWBc)*. In the case where both the *SW* and *GW basins* are spatially compatible and there are no external inflows to or outflows from the basins, the SWBc equation for a given SW basin over a given time interval Δt could be expressed as (Eq. 2.1.)

$$\boldsymbol{P = Q + ET + \Delta S_S + \Delta S_G} \qquad (2.1.)$$

where P is the precipitated amount of water on the SW basin, Q the SW discharge or runoff out from the SW basin, ET the evapotranspiration from the SW basin, ΔSs the change in water storage of the SW reservoir, and ΔS_G the change in water storage of the GW reservoir (both saturated and unsaturated conditions) during for instance *annual* recording *period* (Freeze & Cherry 1979).

For a longer, i.e. the *long-term* period the storage changes in both the SW and the GW reservoirs assumed to be stable, and therefore Eq. 2.1. transforms to a well-known simplified Eq. 2.2. form,

(2.2.) $$\bar{P}=\bar{Q}+\overline{ET}$$

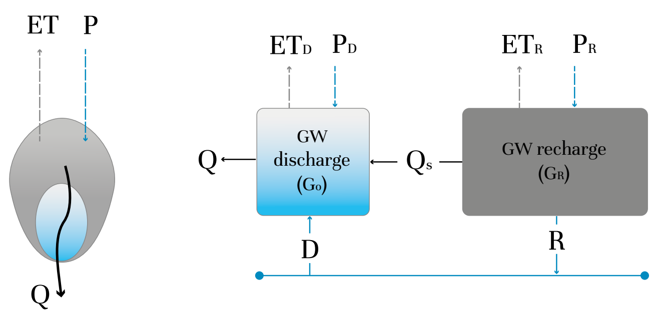
where $\bar{P}$ is the long-term precipitation amount, $\bar{Q}$ the long-term SW runoff, and $\overline{ET}$ the long-term annual evapotranspiration from the SW basin. The values of Eq. 2.2. are usually reported in mm-s over the basin.

However, if it is assumed that the major part of the SW basin acting as a GW recharge area and there is a very $\bar{P}$ small part of the SW basin acting as a GW discharge area (Figure 2.5.), then the SW basin recharge equation is turning into the Eq. 2.3.

(2.3.) $$P_R = Q_S + G_R + ET_R$$

where P_R is the recharge area precipitation, Q_S the SW runoff, G_R the recharge to the GW and ET_R the evapotranspiration of the GW recharge area. The values of Eq. 2.3. are usually reported in mm-s over the drainage basin.

At the same time (Figure 2.5.) the GW discharge component in the SW discharge could be expressed via Eq. 2.4. as following



(a) (b)

Figure 2.5. Conceptual schemes of the steady-state hydrologic conditions with groundwater (GW) inflow component (b) for the small surface water (SW) basin (a), where the GW recharge area (the green polygon on the scheme) covers a comparably large percentage of the SW basin area; the blue polygon on the scheme is the GW recharge area. The used symbols: P – precipitation, ET – evapotranspiration, Q – SW runoff with GW inflow component, Qs – SW runoff without GW component, R – SW inflow to the GW from the GW recharge area, D – GW outflow from the GW discharge area (scheme modification based on Freeze & Cherry (1979)).

(2.4.) $$Q = Q_S + D - ET_D$$

where Q is the SW runoff together with the GW discharge component, Q_S is the direct SW runoff, D is the GW discharge and in principle equals to R, i.e. to the GW recharge or infiltration, and ET_D is the evapotranspiration from the GW discharge area. Equating the GW discharge components in the following (Eq. 2.5)

(2.5.) $$D - ET_D = Q_G$$

the Eq. 2.4. becomes Eq. 2.6.

(2.6.) $$Q = Q_S + Q_G$$

where Q is the SW runoff together with the GW discharge component, whereas the GW discharge Q_G forms the so-called GW baseflow in the SW runoff (Figure 2.6.). Based on Eq. 2.6. the *hydrograph* of the SW runoff consists of two, SW and GW components (Figure 2.6.).

It also means that increased GW baseflow is the result of increased hydraulic gradients in saturated zones near the SW water body, and this is itself a consequence of increased up-basin gradients created by a GW table rise (Freeze & Cherry 1979). The time lag between recharge area infiltration event and increase in stream baseflow is therefore directly related to the time required for infiltration components to induce a widespread GW table rise (Freeze & Cherry 1979).

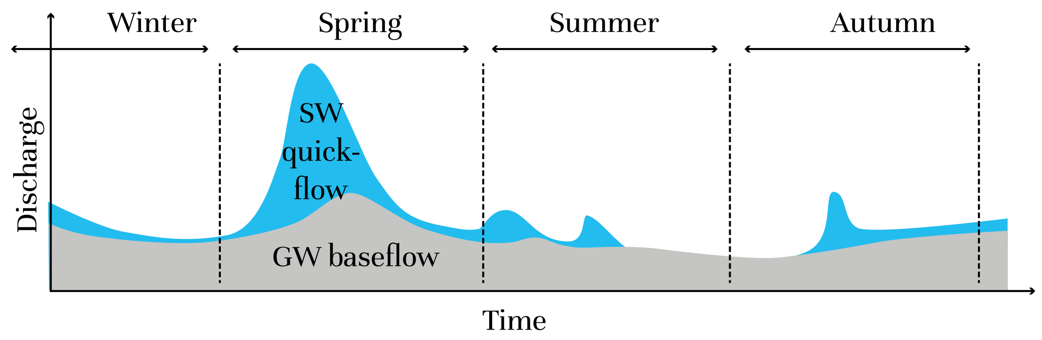


Figure 2.6. Conceptual scheme of the surface water (SW) runoff hydrograph with the SW and groundwater (GW) components, i.e. quickflow and baseflow correspondingly The hydrograph is based on visualisation of recorded SW runoff values at the SW gauging station over the certain time period.

There are different types of SW hydrographs (e.g. unit, flood, annual, long-term hydrographs) used for analyses of different SW runoff components (Figure 2.7.). Since in hydrological systems interactions of SW and GW components are important for efficient water resources management incl. protection, there are different approaches for determining their interactions. The most known approach is the hydrograph analysis, whereas the recession curve, the constant slope or the straight line separation methods (Raghunath 2006) are the most simplified and known approaches for the GW component separation (Figure 2.7.).

In addition to the above described, the temporal share between the GW baseflow and SW discharges in hydrographs may vary significantly, depending on hydro-topographical-climatological conditions of the GW recharge area. Components of the GW baseflow formation depend on the type of precipitation (snow, rain), air temperature, freezing state of the soil and SW infiltration conditions in the GW recharge area. Hereby, it is important to notice that due to the needed lag time for the GW

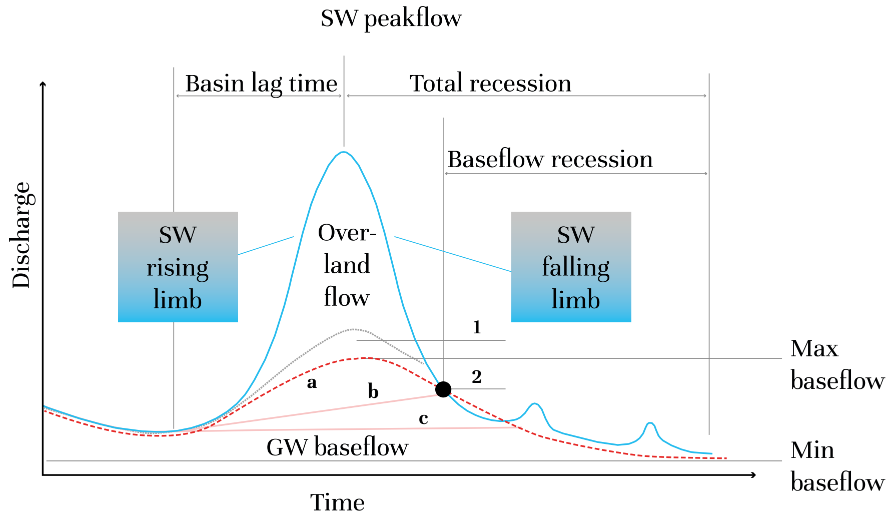


Figure 2.7. Example of the SW hydrograph delineation to the SW and the GW components during the SW flood period. Where: 1- throughflow; 2 – SW inflection point; a, b, c – GW components separated by the (a) – recession curve, (b) – constant slope, and (c) – straight line methods (e.g. Raghunath 2006).

transport from the GW recharge area to the discharge there is always the GW baseflow hydrograph peaks shifted to the right from the SW peaks (Figure 2.6. and 2.7.), i.e. the baseflow peaks of the discharge recording profiles always formed later in comparison with overland flow peaks (Luo 2015). Also, the flatter baseflow hydrographs with smaller recession constants refer to the stronger water flow regulation effects of the lake or stream aquifer systems in the basin (Luo 2015).

2.2.2. Chemical and isotopic indicators for groundwater-fed systems

One of the widely known tools for analyses of the SW-GW components in GW dependent discharge profile is the chemical mass-balance for a specified recording place and time (Figure 2.8.). Expressed it for the steady-state conditions of dissolved chemical constituent (e.g. Na^{2+}, Ca^{2+}, Mg^{2+}, Cl^{-}, SO_4^{2-} or HCO_3^{-}) the chemical mass balance for the SW components may be compiled as follows (Freeze & Cherry 1979)

(2.7.)

$$CQ = C_P Q_P + C_O Q_O + C_S Q_S + C_G Q_G$$

where C is the concentration of dissolved constituents in SW sampling. The C_P, C_O, C_S, and C_G are the chemical constituents in direct rainfall Q_P, overland flow Q_O, subsurface stormflow Q_S, and groundwater Q_G. respectively.

However, for practical reasons, derivation of the GW component of Q_G from the chemical mass-balance equation (2.7.) based on assumption that $Q_O + Q_S = Q_D$, i.e. the Q_D is the direct discharge into the stream SW discharge and physically represents the component of rainfall that moves rapidly across or through the ground into the stream and is obtained by sampling surface drainage or soil-zone seepage near the stream during the storm-runoff period (Freeze & Cherry 1979). Based on that the GW component could be estimated from

(2.8.)

$$Q_G = Q\left(\frac{C - C_D}{C_G - C_D}\right)$$

where values of C_G are normally obtained by the water samplings from the GW wells or piezometers near the stream or by sampling the stream baseflow prior to, or after, the storm. If the C and Q are measured in the stream at various times during the storm-runoff period, the variation of Q_G can be computed, as shown schematically in Figure 2.8.

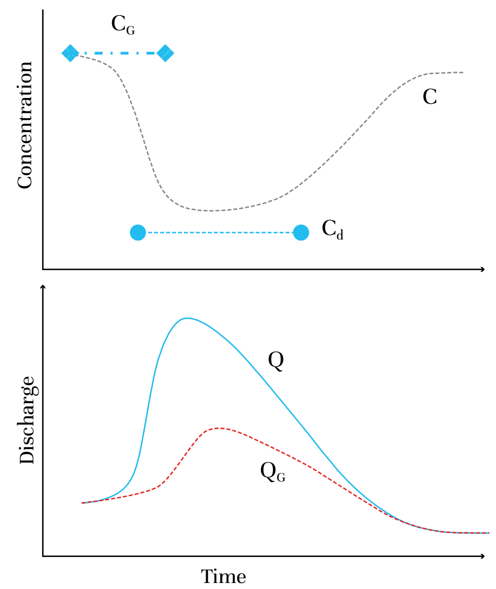


Figure 2.8. Graphical visualisation of calculated groundwater (Q_G) component in the SW discharge, Q (Eq. 2.8.), where: C – sampled chemical constituent in the stream Q over the certain sampling period Δt, C_d and C_G – sampled chemical constituent in direct (Q_d) and Q_G components accordingly (scheme modification based on Freeze & Cherry (1979)).

One of the main limitations of the chemical mass-balance method is that the chemical concentrations used for the shallow GW, to represent the direct runoff, are lumped parameters that may not adequately represent the water that actually contributes to the stream discharge. Therefore, the direct runoff Qd is a very ephemeral entity that may vary considerably in concentration in time and space. The chemistry of shallow GW obtained from wells near streams is commonly quite variable spatially as well (Freeze & Cherry 1979).

To avoid some of the main uncertainties inherent in the chemical mass-balance method, the naturally occurring isotopes 18O, 2H, and 3H can be used as indicators of the GW component of streamflow during periods of storm runoff (Freeze & Cherry 1979). The 18O method is suited for the type of rainfall event in which the 18O content of the rain is relatively constant and is very different from the shallow GW or baseflow. In this situation, 18O of the rain is a diagnostic tracer of the rainwater that falls to the SW basin during the storm, and the part of the GW discharge could be evaluated according to Eq. 2.9. (Freeze & Cherry 1979).

(2.9.)

$$Q_G = Q_S\left(\frac{\partial^{18}O_S - \partial^{18}O_P}{\partial^{18}O_G - \partial^{18}O_P}\right)$$

where $\partial^{18}O$ denotes the ^{18}O content in per mille relative to the VSMOW standard, the Q_G and the Q_S is the GW and SW discharges, correspondingly, subscripts of S, G, and P for the $\partial^{18}O$ indicate the ^{18}O content for the SW water, shallow GW and precipitation accordingly.

2.3. Types of groundwater exposed springs

Elve Lode (Tallinn University)

The term “spring” is used to describe the source of a stream: the point where the water emerges from the ground wherever the water table is high and/or a layer of impermeable substrate prevents the water from percolating away underground, ensuring that the ground stays waterlogged for most of the year. Spring usually is characterized by a continuous flow of cold water, uniform in temperature and rich in oxygen and minerals (Averis 2003; Blaus et al. 2020).

Spring flushes on the other hand mark out places where water flows over the ground more diffusely. Flushes occur on gently sloping ground, or often linear or triangular and may include small watercourses. The presence of a well-developed bryophyte ground layer and the lack of dominant grasses distinguishes flush habitats from wet acid, neutral and calcareous grasslands.

Spring mires or soligenous fens develop mainly in the contact zones of units with different geological structure which creates favourable conditions for the occurrence of large outflows of groundwater rich in calcium carbonate leached from glacigenic sediments. The formation of spring mires and their size is also connected with the manner in which they are supplied (descending or ascending outflows) and the groundwater resources of drained local and transitory aquifer systems. Usually, the habitat tends to be small in area (<1 ha) (Dobrowolski et al. 2010, 2012).

There are at least 2.5 million springs on Earth. They are physically diverse, complex, highly individualistic, highly interactive ecosystems, and an important source of biodiversity. Many springs are recognized also by their socio-cultural value. Ecohydrologists have identified eight suites of variables with potential value in springs ecosystem classification, including (1) the geology of aquifer (such as karstic aquifer or metamorphic strata); (2) springs discharge (e.g. ephemeral springs or deep-aquifer springs with highly steady discharge); (3) water quality (temperature, geochemistry); (4) landscape location; (5) biota, e.g. vegetation, aquatic macroinvertebrates, fish; (6) anthropogenic use and management; (7) source geomorphology (lentic, lotic and paleosprings), and (8) combinations of those variables (Stevens et al. 2021).

2.3.1. Springs

Hydrologically, the places on the earth where the GW is exposed to the surface are called *springs*. The term *spring* comes from the German word *springer* which means “to leap from the ground” (Friedl 2013).

The surface water bodies created by the springs, i.e. by exposed GW, may be called also spring-fed or GW-fed water bodies, e.g. spring-fed stream, spring-fed lake, seasonal spring-fed pool, spring-fed mire or fen. Hereby, from an eco-hydrological point of view, springs could be named *ecosystems* where GW is exposed at the Earth’s surface and typically flow on the Earth’s surface (Stevens et al. 2016a). As a result of the latter, *springs as ecosystems* on the surface could be divided into the *lotic* (flowing water) and *lentic* (standing water) ecosystems (Stevens et al. 2016a).

The term lotic (from the Latin *lotus*, meaning *washing*), refers to running water, i.e. fluvial or fluviatile habitats such as rivers and streams, while lentic (from the Latin lentus, meaning slow or

motionless), refers to standing water bodies such as lakes and ponds (lacustrine), or swamps and marshes (paludal) (Marsh & Fairbridge 1999). Like any ecosystem, lotic and lentic ecosystems can be destroyed through natural or human interaction. Lotic and lentic systems may succumb to such things as climate change, being dammed, drained, filled or degraded by non-native species invasion.

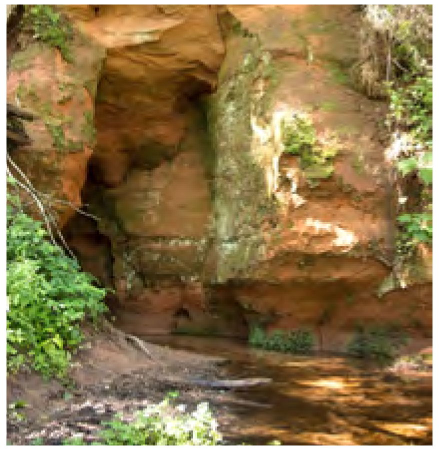
The most recognised lotic spring ecosystems are: caves, hillslopes, rheocrenes, geysers, gushets and hanging-garden springs, and the lentic spring ecosystems are: helocrenes, exposures, hypocrenes, limnocrenes, (carbonate) mound-forming springs and (semi-lotic) fountain springs (Springer & Stevens 2009; Stevens et al. 2016a, b; SSI Webpage).

Lotic springs

a) ***Cave springs*** of the karst terrain (Figure 2.9.) emerge entirely within a cave formation and they are the most common in the *mature to extreme* karst formation conditions with large conduits that allow for GW emergencies in *free draining or dammed* type karst springs. However, emerging within the cave they flow out into the surrounding landscape. The cave spring ecosystems can support a wide range of biota. In this guideline, these ecosystems are later named also as aquatic spring ecosystems. The common stressors: GW extraction, pollution, and recreation.

In the Baltic region, cave springs can also occur in sandstone formations, where caves are formed by suffosion processes.

Example site from Estonia: Merioone cave spring located in south-east of Estonia; discharge of



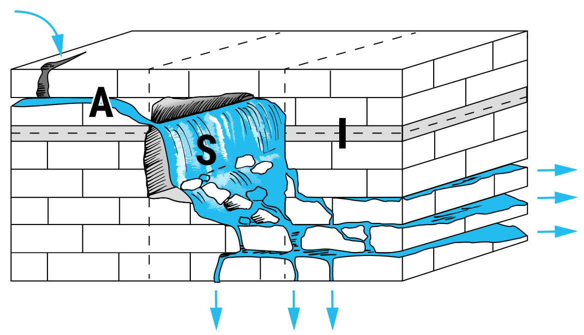


Figure 2.9. Generalised diagram of cave spring formation bottom left (modified from Springer &Stevens (2009), Stevens et al. (2016a, b), SSI Webpage), Merioone, non-karstic cave spring (Estonia) top left (photo: Eesti geoloog 2016), Rūcamavots, non-karstic cave spring (Latvia) top right (photo: A. Priede). In the diagram: A – groundwater aquifer, I – impermeable stratum, S – spring.

the spring is 16–23 l/s; stable water temperature around 6°C (data source here and further: Estonian Nature Information System EELIS).

Example site from Latvia: Rūcamavots spring is located in Gauja National Park, in Sarkanās Cliff, near town Cēsis; stable water temperature around 7.2°C, chloride content around 18.7 mg/l, pH 7.1–7.5.

b) ***Hillslope springs*** of sloping land (Figure 2.10.) are most often diffuse at the source of the springs and more canalised downstream from the source of the springs. They often support a wide array of wetland and riparian vegetation associations; when hillslope springs are travertine-forming, there is often an associated bryophytes (moss) community. Hillslope spring discharges support a diverse array of microhabitats. The slope gradient is usually negatively related to floral diversity. Aspect also strongly influences diversity, although those relationships have yet to be rigorously quantified. By formation of emerging discharges, the hillslope springs can be perennial or ephemeral, and topographically they can be rheocrenic and upland hillslope springs.

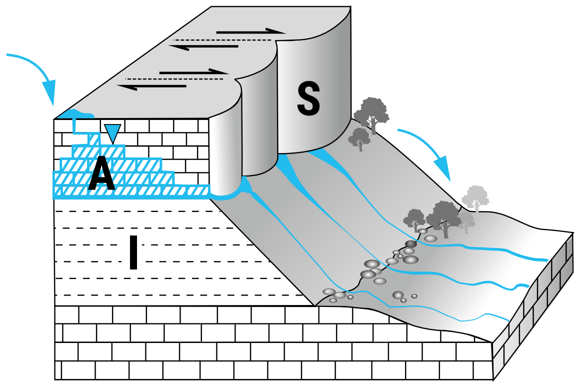


Figure 2.10. Generalized diagram of hillslope spring formation bottom left (modified from Springer & Stevens (2009), Stevens et al. (2016a, b), SSI Webpage), Viidumäe hillslope springs (Estonia) top left (photo: M. Vainu), Kazu leja hillslope springs (Latvia) top right (photo: J. Bikše). On the diagram: A – groundwater aquifer, I – impermeable stratum, S – spring.

Rheocrenic or floodplain/riparian hillslope springs emerge from the bank or terrace of a river or stream (distinguished from a true rheocrene spring, which sources on the stream bed). Rheocrenic hillslope springs are subject to regular stream or river flooding and usually contain widespread, flood-tolerant species. Upland hillslope springs are located outside of a riparian setting, they are not subject to stream flooding and commonly support rare species. Anthropogenic subtypes of hillslope springs are also possible; these can be created by pipe or ditch leakage. In this guideline, these ecosystems are later named aquatic spring ecosystems. Common stressors: GW extraction, recreation, non-native species introduction and climate change.

Example site from Estonia: Viidumäe hillslope springs located in the south-western part of the Western Saaremaa Upland on Estonian Saaremaa Island. The springs are exposed on terraces of Ancylus Lake either as point sources or as seepage springs. Most of the springs are small with a discharge rate of 0.1–1.0 l/s). These springs are also named rheocrenic springs emerging from an unconfined GW aquifer, located on a slope in sandy or gravelly soil.

Example site from Latvia: springs on a hillslope of Kazu leja valley located in the Gauja National Park, near Cēsis. The springs emerge from fractured dolomite aquifer with an impermeable clay layer beneath, thus several individual springs and spring seepage in colluvium deposits can be observed. Precipitation of freshwater tufa occurs in individual springs and extensive tufa mining has occurred within the vicinity in the past. Spring water temperature is 7–7.7°C, EC around 550 µS/cm and low chloride content of about 1.5 mg/l, pH 7.1.

c) ***Rheocrene springs*** or flowing springs (Figure 2.11.) generally occur because of geologic structural constraints on the GW flow path. This is often visible as the narrowing of a bedrock canyon, which forces GW out of floodplain alluvium and into the stream channel. Rheocrene springs are most visible when they emerge into otherwise dry channels, but they can also emerge into perennial streams, i.e. to permanent spring-fed streams including spring-brooks or spring-runs. They have been described as areas with relatively uniform temperature and the de-oxygenated GW contribution to the stream.

These springs could form a surface water continuum with springs discharge dominated and those dominated by surface water discharges. Surface water dominated spring channels are influenced by flood-related disturbances, whereas spring flow-dominated channels tend to provide stable habitat that allows for evolutionary micro-adaptation. In this guideline, these ecosystems are later named also as aquatic spring ecosystems.

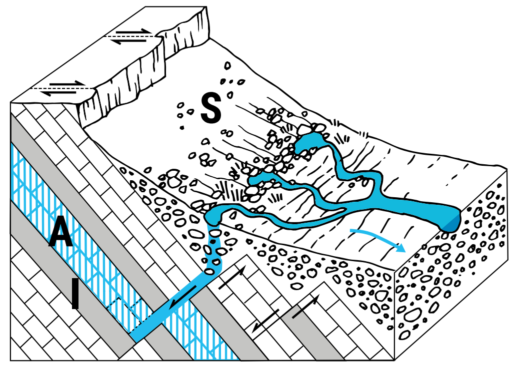


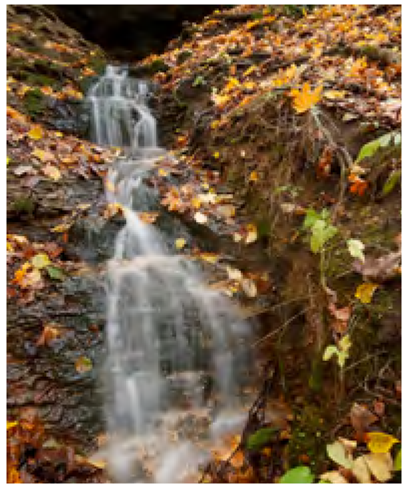
Figure 2.11. Generalised diagram of rheocrene spring formation bottom left (modified from Springer & Stevens (2009), Stevens et al. (2016a, b), SSI Webpage), Lavi reocrene spring (Estonia) top left (photo: M. Vainu), Aclejas springs (Latvia) top right (photo: A. Priede). In the diagram: A – groundwater aquifer, I – impermeable stratum, S – spring.

Common stressors: GW extraction, livestock water supplies, agricultural hay-mowing, urbanization, road construction (may dewater or divert water from the downslope portion, or alter channel margins), recreation, non-native species invasion, climate change.

Example site from Estonia: Lavi rheocrene spring located at the start of Lavi stream in the north-eastern part of Estonia. The GW is exposed from the limestone fractures and gives a start to the Lavi stream. The flow rate is relatively constant and the water quality is good. Water temperature is varying between 3.6–7.5 °C. The measured flow rate is varying between 110–440 l/s (Kink 2004), the water pH 7.1–7.5 (Ca-HCO_3).

Example site from Latvia: Aclejas springs located on the valley slope of River Abava near Renda in Western Latvia. Springs flow through a 250 m wide and 400 m long valley hidden in spruce forest, in total there are about 6–12 springs. The largest spring is 6 m wide and 3 m deep resembling a pool. The GW level in spring area is 24–25 m a.s.l., the water flow rate is slow with low pressure, it emerges from sand-gravel layer (Dēliņa 2021). Around the spring area, a micro reserve for a rare and protected bryophyte species *Trichocolea tomentella* has been established.

d) ***Gushet springs*** of nearly vertical cliffs (Figure 2.12.) emerge as focused GW flow cascades from nearly vertical cliffs. Gushet springs emerge from discrete sources in cliff faces. They typically discharge from perched, unconfined aquifers, often along fractures. Gushets often support madicolous habitat, thin sheets of water flowing over rock faces. They frequently contain many different microhabitats, supporting diverse ecosystems. Although they occur primarily in areas dominated by steep escarpments, they are also found in regions with more modest topographic relief.



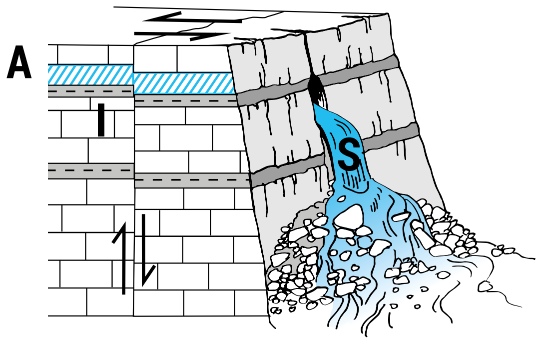


Figure 2.12. Generalized diagram of gushet spring formation bottom left (modified from Springer & Stevens (2009), Stevens et al. (SSI Webpage), gushet spring of Salumäe Silmaallikas (Estonia) top left (photo: A. Arnwald 2007), Dauģēnu cirka spring (Latvia) top right (photo: J. Bikše). In the diagram: A – groundwater aquifer, I – impermeable stratum, S – spring.

In this guideline, these ecosystems are later named also as aquatic spring ecosystems. Common stressors: GW and surface water extraction, livestock water supplies, recreation, non-native species invasion, climate change.

Example site from Estonia: gushet spring of Salumäe Silmaallikas located in Western Estonia. The GW discharges from between the layers of limestone and flows down the slope as a spring brook. The flow rate reaches up to 2 l/s in the spring. Salumäe terrace is a wintering place for vipers.

Example site from Latvia: Dauģēnu cirka spring is located in Northeastern Latvia, at Dauģēnu cliffs. The spring emerges at the contact of GW bearing sandstone and low permeability clay layer beneath and the lower part of the vertical outcrop. The spring is characterized by low mineralization and temperature around 7.6°C.

e) ***Semi-lotic fountain springs*** (Figure 2.13.) can be ephemeral or perennial; an anthropogenic subtype can be created by drilling into an artesian aquifer. At fountain springs, cold groundwater is forced out of the Earth by stratigraphic head pressure or CO_2. Discharge is caused by a confined, pressurized aquifer, rather than by heat as is the case with geysers. Fountain springs may be cold water, submarine seeps of hydrocarbons, carbonates or brine. Alternate names: semi-terrestrial; palustrine or rarely, lacustrine. In this guideline, the latter named aquatic spring ecosystems. Common stressors: GW extraction, pollution, livestock water supplies, recreation, non-native species invasion, climate change.

Example site from Estonia: the man-made artesian Purskav spring in Central Estonia. The spring formed due to geological drilling around the Oostriku River in 1980. Initially, the water erupted to a height of almost a meter, now the water pressure has dropped, but during high water, the spring erupts to a height of 0.7 m (Vilbaste 2013). The spring is located in the Endla Nature Reserve, central Estonia.

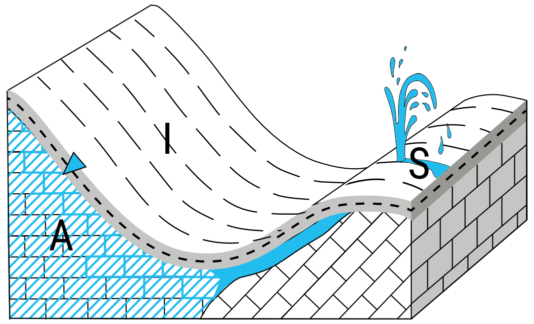


Figure 2.13. Generalised diagram of semi-lotic fountain spring formation bottom left (modified from Springer & Stevens (2009), Stevens et al. (SSI Webpage), man-made Purskav spring in Estonia top left (photo: A. Ansberg), Lūžņu grāvis spring (Latvia) top right (photo: J. Bikše). On the diagram: A – groundwater aquifer, I – impermeable stratum, S – spring.

Example site from Latvia: Lūžņu grāvis springs are located in the central part of Latvia, near Ķemeri resort. Up to five springs emerge at Lūžņu grāvis stream channel with a total discharge rate varying seasonally from 8.9 to 14.5 l/s (Prols 2010). The spring water contains hydrogen sulfide and sulfur precipitation can be observed on the stream bed.

Lentic springs

g) ***Helocrene springs*** (Figure 2.14.) emerge diffusely in a marshy, wet meadow setting rather than having a discrete source. GW discharge from these helocrenes in regions outside the Baltics is *typically saline*, but in the Baltic countries, the helocrenes have *freshwater* and low oxygen concentrations that sustain many wetland species. Helocrene spring ecosystems typically are *low-gradient marsh-forming or gravity-driven or spring-fed wet meadows*. The GW emerges to the surface diffusely from a contact or seepage geologic setting, without discrete sources on the surface.

In Fennoscandia, they are forming *mineral-rich spring fens* or mires of multiple subclasses. This spring type can be ephemeral or perennial. Many helocrene springs are alkaline. In this guideline, these ecosystems are later named also as terrestrial spring ecosystems. Common stressors: GW extraction, livestock water supplies (creation of open water), agricultural hay-mowing, urbanization, road construction (may dewater the downslope portion), peat mining, recreation, non-native species invasion, climate change.

Example site from Estonia: Avaste fen covering 8702 ha belongs to the Lihula-Lavassaasere mire complex (37 810 ha) on Western Estonia Lowland. It is a typical GW-feed mire massif in lowland conditions, formed via paludification of the carbonate bedrock. The main distributed peat is the fen peat with the thickness of 2.6–4.1 m, overlying the loam, clay, and moraine deposits. The mire is very wet and its water level is often 0.2–0.3 m above the soil surface. However, in its development, this once nutrient-rich fen massif is becoming a poorer transitional bog: this is indicated by large, few meters in diameter hummocks with the bog and forest plant species.

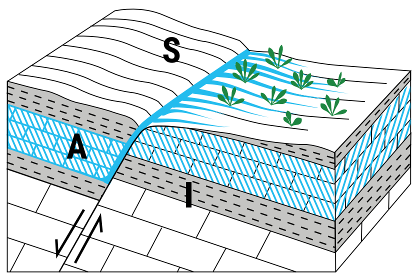


Figure 2.14. Generalized diagram of helocrene spring formation bottom left (modified from Springer & Stevens (2009), Stevens et al. (SSI Webpage), GW-fed Avaste fen (Estonia) top left (photo: I. Kruusamägi, 2015), Diļļu Meadows (Latvia) top right (photo: A. Priede). On the diagram: A – groundwater aquifer, I – impermeable stratum, S – spring.

Example site from Latvia: Diļļu Meadows in Western Latvia hosts an alkaline fen (nowadays largely transformed into meadows) located on the hillslope foot of the shore of the ancient Baltic Ice Lake. Spring emergence is supported by low permeability glacial till layers in shallow depth below the meadow, while GW seeps from the shore of the Baltic Ice Lake with an elevation of up to 30 m above the meadow.

h) ***Limnocrene springs*** (Figure 2.15.) emerge into an open pool of water as confined or unconfined GW aquifers. Limnocrene springs can be perennial or ephemeral: for example, the disappearing lakes found in limestone settings in Ireland. Limnocrene paleosprings (i.e. springs that flowed in prehistoric times, but no longer flow) can sometimes be recognized. Anthropogenic limnocrenes include springs formed in GW depending on livestock watering tanks, mine pits, quarries, etc. Due to their relatively uniform temperature and chemistry, the sources of these springs may support aquatic species that are different from surrounding habitats influenced by surface water. Alternate names are depressions, sinkholes, lacustrine wetlands or aquatic bed wetlands; GW-fed ponds, pools, tanks, quarries (anthropogenic), or lakes; acid limnocrenes. In this guideline, these ecosystems are later named aquatic spring ecosystems. Common stressors: GW depletion, agricultural and mining pollution, urbanization, pond margin habitat alteration, livestock grazing, recreation, non-native species invasion, climate change.

Example site from Estonia: the limnocrene Äntu Sinijärv Lake is located in northeastern part of Estonia. It is an alkalitrophic lake with the clearest water in Estonia. Exposed GW springs ensure high water transparency, up to 15 m in the lake, although the maximum depth in the lake is only 8 m. The summer maximum water temperature is between 10–15^{0}C (Wikipedia 2018). Due to very low level of nutrients, the lake is species-rich, with unique vegetation. The shores are quaking, the bottom is covered with lake lime, on which lush vegetation grows (e.g. *Charophyta* spp., *Fontinalis* spp., *Hippuris* spp.).

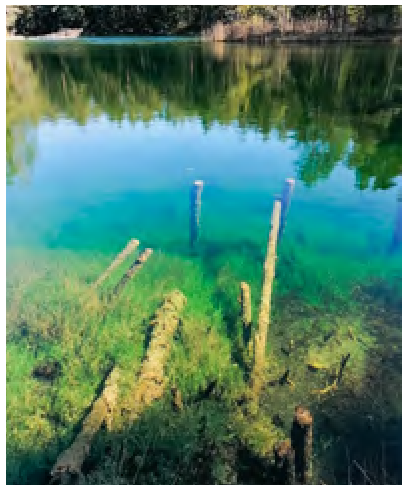
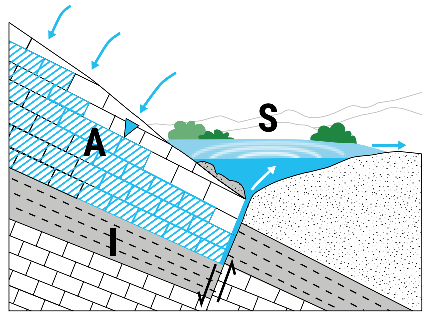


Figure 2.15. Generalized diagram of limnocrene spring formation bottom right (modified from Springer & Stevens (2009), Stevens et al. (SSI Webpage), limnocrene Äntu Sinijärv Lake (Estonia) top left (photo: Ritassilla 2020), sulfur pond in Raganu Mire (Latvia) top right (photo: L. Strazdiņa). On the diagram: A – groundwater aquifer, I – impermeable stratum, S – spring.



Example site from Latvia: Raganu Mire with sulfur pond is located in the central part of Latvia, near Ķemeri. Sulphide-rich GW seeps underwater and precipitates sulfur sediments on the bottom of ponds. Sulfide is formed due to the presence of gypsum sediments in the Devonian aquifer that releases high sulphate content and organic-rich shallow GW. Sulfide concentration can exceed 50 mg/l.

h) ***(Carbonate) mound-forming springs*** (Figure 2.16.) form where elevated calcium carbonate concentration or organic peat mound matter creates a dome form, from which GW emerges and usually flows. Mound-form springs can be perennial or ephemeral. Subtypes include collapsed mound, organic mound, carbonate mound, and ice mound. Secondary springs types often associated with mound-form springs are geyser, fountain, helocrene, limnocrene, and paleosprings. Due to their unique water chemistry, and sometimes due to their isolation in arid regions, these springs often support high biodiversity and endemism. In this guideline, these ecosystems are later named aquatic spring ecosystems. Common stressors: GW depletion; agricultural and mining pollution; urbanization, pond margin habitat alteration, livestock grazing/soil compaction, recreation, non-native species invasion, climate change.

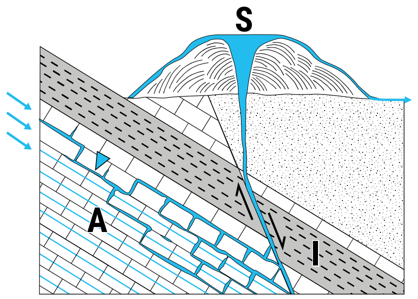


Figure 2.16. *Generalized diagram of (carbonate) mound-forming spring formation bottom left (modified from Springer & Stevens (2009), Stevens et al. (SSI Webpage), Varimõisa carbonate mound spring (Estonia) top left (photo: M. Vainu), Dzērves Bērziņu springs (Latvia) top right (photo: J. Matuko). On the diagram: A – groundwater aquifer, I – impermeable stratum, S – spring.*

Example site from Estonia: Varimõisa carbonate mound spring located in the southeast part of Estonia. There is a dome formation higher than the surrounding ground, with a watery hole on the plateau of the dome. There was no direct drainage from the dome at the end of June 2021, but it was seen that the seep puddle has been larger recently. Apparently, the water seeps into the mosses surrounding the dome. Apparently, the whole dome is formed from the carbonate spring. A swampy spruce forest grows around the dome. There is yellowish-brown sediment in the bottom of the seep, probably precipitated there from the iron-rich water (https://allikad.info/springs/EE01842).

Example site from Latvia: Dzērves Bērziņu Springs are petrifying springs with freshwater tufa formation located in the western part of Latvia, near Cīrava. Several smaller and larger springs can be identified in the central part of the spring system which lies on the foot of a hillslope. Tufa has formed a flat dome with spring ponds and streams on top. Most of the spring water outflowing from the dome is forming a stream with a discharge rate of around 50 l/s. There are few small waterfalls on the stream made of tufa. Tufa formations have been mined in the past and signs of vast excavation works are still visible in the form of ponds and channels. The spring water temperature is around 7–8°C, pH around 8 with a TDS 260 mg/l.

2.3.2. Seeps

Derived from Chorley (1978), it should be noticed that the GW outflows to the ground could take place as a part of the streamflow "...discharged into a stream channel or springs; and seepage water". Thereby the contemporary seep research defines that the "surface seeps are locations where upwelling groundwater saturates the surface. The GW may be transported to nearby surface waters along the surface and shallow subsurface flow paths. Seeps are generally considered to be springs with lower discharge magnitudes" (Springer & Stevens 2009; O'Driscoll et al. 2019).

Seeps may differ from springs, as they often emerge over a diffuse area and generally have low flows that do not form channels. Groundwater seeps often flow diffusely through soils and vegetation (Williams 2016; O'Driscoll et al. 2019), therefore seep discharge may be more difficult to measure relative to larger springs. Williams (2016) provided a classification of seeps into three general classes: helocrene (emerges from wetlands/marshy substrate); limnocrene (discharge into a pool); and rheocrene (flowing spring that emerges into channels) (Figure 2.17.). Seeps and springs can also be categorized based on their magnitude of flow and flow permanence. However, since flow permanence assessment requires monitoring, many studies may not have enough data for accurate flow characterization. Williams (2016) recommended a flow characterization system for low flow: <0.01 m^3/s; medium flow: 0.01–0.5 m^3/s; and high flow: >0.5 m^3/s. Flow is an important variable for

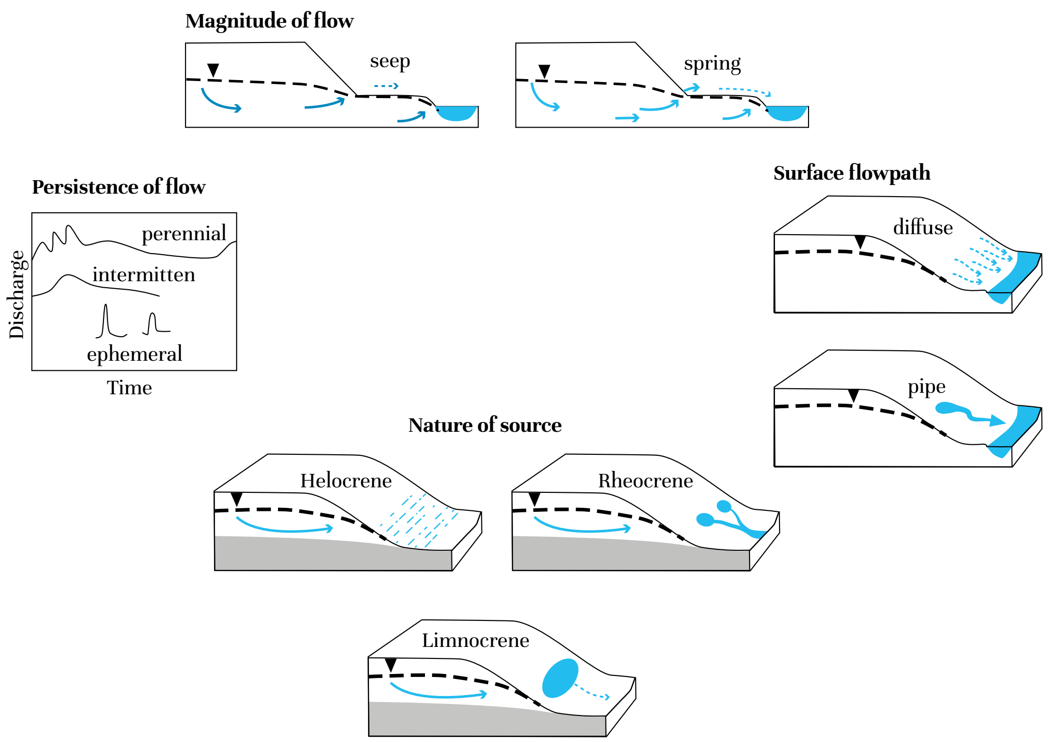


Figure 2.17. *Generalized diagram of groundwater induced seeps (modified from O'Driscoll et al. (2019)).*

characterizing seeps and springs because of its influence on temperature and habitat (O'Driscoll et al. 2019).

Seep discharge can influence the local ecology due to its controls on primary productivity, food supply (leaves and detritus), and influence on spring or seep-bed substrates (Williams 2016). Seep flow magnitude and timing can influence the extent of the seep habitat, disturbances, availability of food, temperature, moisture, and water quality. The invertebrate community that lives in and around the seeps is generally adapted to the range of common flow conditions. From an ecological perspective, seeps may have less diverse fauna than springs, but there may be genera found only in seeps (Williams 2016; O'Driscoll et al. 2019). Seeps are important to GDEs due to the GW inputs they provide and their influences on temperature, water chemistry, riverine biota, and in-stream processes (Boulton & Hancock 2006; O'Driscoll et al. 2019). Seeps can provide a wide range of ecosystem services, can serve as a linkage between the GW and SW system and during summer base flows, they may provide the dominant source of streamflow in some headwater catchments (O'Driscoll et al. 2019).

3. Groundwater dependent ecosystems in Latvia and Estonia

3.1. Identification of groundwater dependent ecosystems

3.1.1. Terrestrial groundwater dependent ecosystems

Agnese Priede (Nature Conservation Agency, Latvia)

Groundwater dependent ecosystems (GDEs) represent a group of ecosystems, where the groundwater conditions, biotic communities including vegetation and other conditions vary from site to site. That raises difficulties to classify and identify these ecosystems (Eamus et al. 2006; Krogulec et al. 2016). There is an enormous diversity of GDEs worldwide, and the ecosystem dependency on GW is difficult to determine and quantify, whereas in-depth studies are very complex and time-consuming, which leads to a need for simplified classification systems (Eamus et al. 2006).

A conceptual framework for GDE identification in the European Union is provided in a technical guidance document by the European Commission (EC 2011a). It does not set strict rules for the EU Member States, thus GDEs can be identified by developing national or regional methodologies.

In the Baltic countries, a methodology for identification, assessment and monitoring schemes for groundwater dependent terrestrial ecosystems (GDTEs) were first developed in Estonia by a research group in Tallinn University (Terasmaa et al. 2015). According to this methodology, in Estonia altogether 70 mires, mire systems or groups of mires were identified as GDTEs depending on groundwater bodies. Identification was fully based on expert decision using the data from the Estonian mire inventory (Paal & Leibak (eds.) 2011) as the background information.

A joint methodology for identification of GDTEs for Estonia and Latvia (Retike et al. 2020) was developed within a joint Estonia-Latvian project "Joint management of groundwater dependent ecosystems in transboundary Gauja-Koiva river basin" (GroundEco, Est-Lat62, 2018–2020) and was tested in Gauja-Koiva basin on both sides of the country border.

The joint identification methodology for GDTEs in Estonia and Latvia, though still requires expert judgment, is based on habitat types that are listed in Annex I of the Council Directive 92/43/EEC (known as Habitats Directive) – hereinafter called Annex I habitats. In Latvia and Estonia, the joint methodology for the identification of GDTEs focuses on ecosystems that critically depend on GW, i.e. deficiency or quality deterioration of groundwater may cause serious damage or loss of the ecosystem. A similar approach is used in other EU countries, e.g. Ireland, Austria, the United Kingdom.

It is important to notice that not all wetlands can be considered GW dependent. Here, "GW dependent" refers to ecosystems that *critically depend* on GW. GW may provide direct input, as in fens and spring mires, and the GW may support the maintenance of high and stable water levels within the ecosystem, as in raised bogs (Kilroy et al. 2008; Pajula et al. 2015). The "critical dependence" means that the primary, dominating source of water is GW, and, consequently, any changes in the GW body would lead to alteration in the associated ecosystems. Changes in GW supply, either in quantity or quality, may cause changes in flora and fauna and further lead to changes in the ecosystem functions.

As mentioned above, the GDTE identification in Latvia and Estonia is based on Annex I habitat maps. Annex I lists natural and semi-natural habitats of Community interest that are in danger of disappearing, or that have a small natural range, or that present outstanding examples of typical characteristics of one or more of the biogeographical regions of Europe. The Annex I habitat identification is based on common EU guidelines (EC 2013), however, in most EU Member States including Estonia and Latvia, national interpretation manuals are used. The national Annex I habitat identification manuals (Auniņš (ed.) 2013 and https://www.daba.gov.lv/lv/biotopu-kartesanas-metodikas-0; Paal 2007 in Estonia) provide more details and national adaptations that slightly differ from country to country.

Two main differences in habitat identification at the national level were found. In Latvia, the interpretation of the habitat type Alkaline fens (7230) strictly follows the EU habitat interpretation manual (Auniņš (ed.) 2013 and https://www.daba.gov.lv/lv/biotopu-kartesanas-metodikas-0) and includes only base-rich (calcareous) fens, while in Estonia it covers both rich (calcareous) and poor fens (acidic, oligotrophic). Poor fens do not have any protected habitat status in Latvia, either national or EU importance. Therefore, poor fens cannot be distinguished as GDTEs in Latvia from the Annex I habitat data, although they may be important in terms of ecosystem services and biodiversity conservation. In fact, poor fens should be considered GDTEs also in Latvia; however, currently, they cannot be identified on the map due to lack of data, unless a special inventory is done. The other major difference refers to the habitat type Hydrophilous tall herb fringe communities (6430). In Latvia, this habitat type includes only alluvial riverbank communities and dry and semi-dry forest edges and therefore does not meet the criteria of GDTEs. In Estonia, the habitat type includes not only riparian tall herb communities but also species-poor fens and paludified grasslands, which are considered GDTEs. In Latvia, poor paludified meadows depending on vegetation composition and abiotic conditions may be considered one or another grassland habitat type under Annex I and consequently not identified as GDTE.

The Annex I habitat types and their identification in the field are primarily based on phytosociology, therefore GDTEs may not be always properly identified using Annex I habitats as the base layer. For example, humid dune depressions may host various habitats, such as pioneer vegetation on wet sand, fens, transition mires, raised bogs, or deciduous swamp woods. Some GDTEs are not distinguished as separate Annex I habitat types and considered as part of larger complexes, such as GW-fed lagg fens at the edges of rain-fed bogs and thus "lost" as separate GDTE units. Still, Annex I habitat inventories provide so far the most complete ecosystem maps in both Estonia and Latvia, and therefore these are the most complete data to be used in GDE identification unless a special mapping of GDEs is done. Therefore, expert judgment is still needed.

Typical GDTE examples in the Baltic countries are springs (including spring mires and spring flushes), fens, swamp woodlands, and humid dune slacks. Although their hydrological and hydrochemical conditions, vegetation and other properties can vary from site to site, they are all primarily GW-feed. At the same time, none of these ecosystems is fed exclusively by GW, as a certain proportion of water supply comes from surface runoff and precipitation (see also Chapter 2).

Typical GDTE examples from Latvia and Estonia are given in Figure 3.1.–3.3.

Figure 3.1*. Spring flush with calciphilous fen vegetation in Muhu Island, Estonia. Photo: A. Priede.*

Figure 3.2*. Alkaline, spring-fed fen in Northwestern Latvia near Pope. Photo: A. Priede.*

Figure 3.3*. Spring-fed deciduous swamp forest in Slītere National Park, Latvia. Photo: A. Priede.*

Raised bogs, transition mires and bog woodlands are rather common wetland types across Latvia and Estonia. Transition mires, though partly dependent on GW supply, in most cases are primarily fed by precipitation, whereas raised bogs are feeding only on precipitation (Pajula et al. 2015). During the raised bog development process, shallow basins fill in with peat, and gradually the peat body rises above the local GW table (Lindsay 2016). Raised bogs may also develop by overgrowing of lakes and ponds, a process where the formation conditions are different, nevertheless, the result is similar – the climax stage is an ombrotrophic peatland fed by precipitation. In further stages, bog woodland may develop which are part of raised bog complexes (on bog margins, around mineral islands in the bog) or the bog can fully overgrow with forest.

Raised bogs covering large areas are often actually complexes of various habitats that include typical ombrotrophic (rain-fed) bog, bog pools, bog woodlands, and lagg zone in the marginal area. Laggs, though part of raised bog, have different feeding regimes and, in fact, are GDTEs (see more about laggs in Chapter 3.3.5). However, in raised bogs and transition mires the peat layer is not always fully isolated from the GW aquifer by impermeable layers. In such cases, dropping of GW table causes dropping of the mire water table (Pajula et al. 2015). Although these ecosystem types are not directly dependent on GW, lowering of GW table due to some reasons (water extraction, mining) can cause deterioration of raised bogs and transition mires, as it has already been detected in Northeastern Estonia in the vicinity of oil-shale mines (Marandi et al. 2013).

Similarly to Northeastern Estonia, such impact may also affect some raised bogs and related transition mires and bogs woodlands in Latvia where they are located close to mining areas (gravel, dolomite quarries). More on this type of impact: see Chapter 5.

Besides the above mentioned, there are several other wetland types that do not fully qualify as GDTEs, because they are primarily fed by surface runoff and/or precipitation, or primarily depend on seawater flooding or occur in river floodplains. Such ecosystems are coastal meadows (habitat code 1630*) and coastal lagoons (1150*), alluvial forests (91E0*), alluvial meadows (6450) and alluvial fens. Also, Molinia meadows on calcareous, peaty or clayey-silt-laden soils (Molinion caeruleae) (6410) are excluded from the GDTE list in Latvia and Estonia, unless they occur in a complex with alkaline fens or springs. Often such meadows, sometimes referred as fen meadows in literature, have developed after slight modification of water table in fens, and thus they may be of the same origin (van Diggelen et al. 2006) (Figure 3.4., 3.5.).

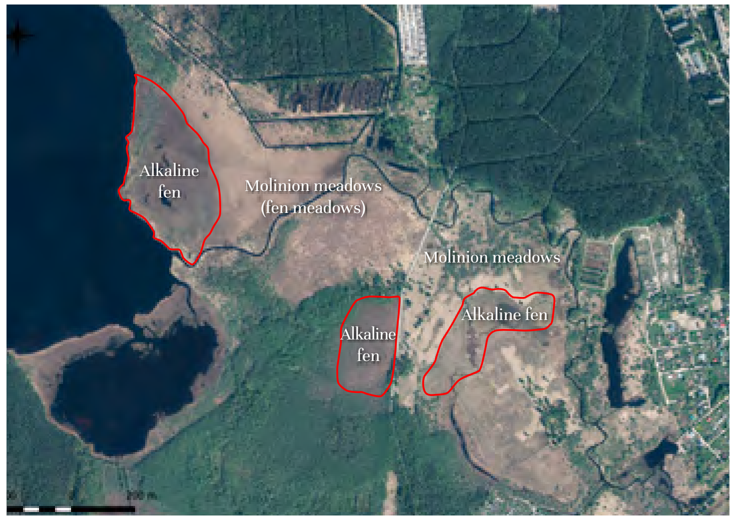


Figure 3.4. A complex of slightly untouched alkaline fens and Molinia meadows.

Figure 3.5. A complex of Molinia meadows and tufa-forming springs in Western Latvia. Photo: A. Priede.

Using available habitat data in combination with certain criteria for data selection that require both additional data layers and expert knowledge, it is still possible that some GDTEs may not be identified on the map. However, the use of existing data significantly reduces the costs of GDTE identification. GDTEs are primarily identified for the needs of the implementation of the Water Framework Directive (WFD). In terms of WFD, the status of GDTE reflects the status of groundwater body (GWB). Therefore, it was decided to focus on significant GDTEs, rather than identifying and further monitoring and managing all areas which are in fact GDTEs. Identifying and further monitoring all GDTEs would be challenging in financial terms. It means that the most significant GDTEs and their "health condition" should indicate changes in the status of the whole GWB. A similar approach is used in most EU countries. However, it does not mean that the other ecosystems with smaller size may be disturbed without any limitations. If they qualify as one or another Annex I habitat type, they are protected under nature conservation legislation, and any indications of their deterioration should be solved using both legislative and management means.

To identify the GDTE locations in Estonia and Latvia, from the list of terrestrial Annex I habitat types the ones were chosen that critically depend on groundwater input according to their definition in the EU manual for Annex I habitat identification or in the national habitat interpretation (Auniņš (ed.) 2013 and https://www.daba.gov.lv/lv/biotopu-kartesanas-metodikas-0; Paal 2007 in Estonia). The habitat types that can be automatically considered GDTEs in Estonia and Latvia are:

- Humid dune slacks (2190[5]);
- Fennoscandian mineral-rich springs and springfens (7160);
- Calcareous fens with *Cladium mariscus* and species of the *Caricion davallianae* (7210*);
- Petrifying springs with tufa formation (Cratoneurion) (7220*);
- Alkaline fens (7230);
- Fennoscandian deciduous swamp woods (9080*).

[5]Here and further the code is given according to the Annex I in the Council Directive 92/43/EEC of 21 May 1992 on the conservation of natural habitats and of wild fauna and flora.

Some habitat types are considered GDTEs only in exceptional cases either in both countries or only in Estonia. These are as follows:

- *Molinia* meadows on calcareous, peaty or clayey-silt-laden soils (Molinion caeruleae) (6410),
- Hydrophilous tall herb fringe communities of plain and of montane to alpine levels (6430),
- Active raised bogs (7110*),
- Degraded raised bogs still capable of natural regeneration (7120),
- Transition mires and quaking bogs (7140),
- Bog woodlands (91D0*) (Table 3.1.).

Table 3.1. Identification key of GDTEs in Latvia and Estonia.

Habitat types listed in Annex I of the EU Habitats Directive 92/43/EEC (21/05/1992)	Additional criteria used (Latvia/Estonia)
Considered GDTEs	
Humid dune slacks (2190)	Single polygon with at least 1 ha area or smaller if part of a habitat complex with the total area of at least 1 ha.
Fennoscandian mineral-rich springs and springfens (7160)	
Petrifying springs with tufa formation (Cratoneurion) (7220*)	
Calcareous fens with *Cladium mariscus* and species of the *Caricion davallianae* (7210*)	Single polygon with at least 10 ha/20 ha area or smaller if part of a habitat complex with the total area of at least 10 ha/20 ha. For 7210* *Cladium mariscus* stands in lakes are excluded.
Alkaline fens (7230)	
Fennoscandian deciduous swamp woods (9080*)	Single polygon with at least 20 ha area or smaller if part of a habitat complex with the total area of at least 20 ha.
Considered GDTEs in exceptional cases	
Molinia meadows on calcareous, peaty or clayey-silt-laden soils (6410)	Considered as GDTE if part of a GDTE habitat complex (e.g. 7210*, 7230) with a total area of at least 20 ha.
Hydrophilous tall herb fringe communities of plain and of the montane to alpine levels (6430)	Single polygon with 20 ha area or smaller if part of a habitat complex with the total area of at least 20 ha (only in Estonia where the habitat type, according to the national habitat interpretation, includes poor fens and poor paludified grasslands).
Active raised bogs (7110*)	*Only in Estonia!* A single polygon with 20 ha area or smaller if part of a habitat complex with a total area of at least 20 ha only in NE Estonia in oil-shale mining region); not considered GDTE in the rest of Estonia.
Degraded raised bogs still capable of natural regeneration (7120)	
Transition mires and quaking bogs (7140)	
Bog woodlands (91D0*)	*Only in Estonia!* Transition mire and bog woodlands (only in NE Estonia in oil-shale mining region) – a single polygon with at least 20 ha area or smaller if part of a habitat complex with the total area of at least 20 ha. *In both countries!* Coniferous fen woodlands (included into 91D0*) – a single polygon with at least 20 ha area or smaller if part of a habitat complex with a total area of at least 20 ha.

Figure 3.6. *GDTE-related species.*

Dactylorhiza incarnata ssp. cruenta
Photo: M. Pakalne.

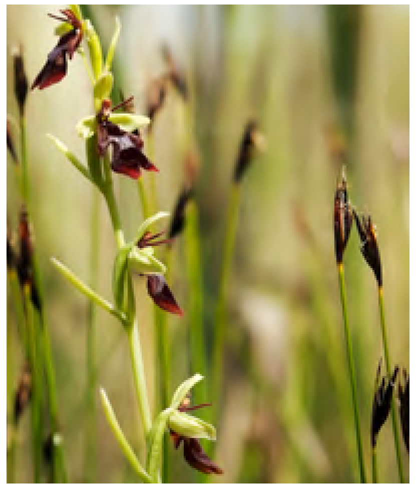
Dactylorhiza russowii
Photo: M. Pakalne.

Liparis loeselii
Photo: M. Pakalne.

Eriophorum gracile
Photo: M. Pakalne.

Juncus subnodulosus
Photo: M. Pakalne.

Saxifraga hirculus
Photo: M. Pakalne.



Ophrys insectifera
Photo: M. Pakalne.

Pinguicula alpina
Photo: M. Pakalne.

Pinguicula alpina
Photo: M. Pakalne.

Carex irrigua
Photo: A.Priede

Ligularia sibirica
Photo: A.Priede

Corallorhiza trifida
Photo: A.Priede

Saussurea alpina ssp. esthonica
Photo: A.Priede

An additional criterion is the presence of **GDTE-related species** that include species of the protection categories I and II in Estonia (according to Paal & Leibak (eds.) 2011), and species that rely on GW-fed habitats listed in Annex I of the Habitats Directive in Latvia (the list of species is given in Table 3.2. and Figure 3.6.). The presence of such species makes the site a significant GDTE, even if it does not meet the threshold of the minimum area.

Table 3.2. *GDTE related species (additional criterion in identifying GDTE sites) according to the joint Estonian-Latvian GDTE identification methodology (Retike et al. 2020)*

Important species relying on GDTEs in Estonia – protected species of the categories I and II (selection according to Paal & Leibak (eds.) (2011), protection categories according to Government Regulation No. 195)	**Important species relying on GDTEs in Latvia** – species listed in Annex II of the Habitats Directive present in Latvia
Angelica palustris, Carex heleonastes, *Carex irrigua* (syn. *Carex paupercula),* *Corallorhiza trifida, Dactylorhiza incarnata* ssp. *cruenta,* *Dactylorhiza russowii, Eriophorum gracile,* *Gentiana pneumonanthe, Gymnadenia odoratissima,* *Hammarbya paludosa, Herminium monorchis,* *Hydrocotyle vulgaris, Juncus squarrosus,* *Juncus subnodulosus, Juncus stygius,* *Ligularia sibirica, Liparis loeselii, Lycopodiella inundata,* *Malaxis monophyllos, Ophrys insectifera,* *Pedicularis sceptrum-carolinum, Pinguicula alpina,* *Rhinantus rumelicus* subsp. *osieliensis,* *Rhynchospora fusca, Rubus arcticus, Saussurea alpina subsp. esthonica,Saxifraga hirculus, Schoenus nigricans,* *Selaginella selaginoides, Swertia perennis*	*Ligularia sibirica, Liparis loeselii,* *Saussurea alpina* ssp. *esthonica, Saxifraga hirculus,* *Vertigo geyeri, Vertigo genesii, Vertigo moulinsiana,* *Vertigo angustior*[6]

Areas that have been highly damaged a long time ago and have a very low or no ecosystem recovery potential may not be considered GDTEs. The criteria to select GDTEs are focused on areas that still have the quality to reach the minimum criteria of the GDTE habitat types listed above. However, they can gain back the status of GDTE if they are successfully restored or recover themselves and again reach the minimum criteria to qualify as Annex I habitat types from the selected GDTE list (Table 3.1.). This is in accordance with the guideline provided by the European Commission (EC 2011a): in good practise, the requirements of WFD should be harmonized with the requirements of the Habitats Directive – its objective is to achieve favourable conservation status of habitats and species of Community interest, i.e. some of the important habitats are overlapping with GDEs and thus, though in a degraded condition, are in the priority list concerning restoration.

[6] The species list can be revised if the knowledge on species relation to certain habitat requirements improve.

3.1.2. Aquatic groundwater dependent ecosystems

Marko Vainu (Tallinn University), Līga Strazdiņa (Nature Conservation Agency, Latvia)

Groundwater dependent (or associated) ecosystems (GDAEs or GAAEs) can be defined as follows: “An ecosystem that is contained within one or more surface water bodies (rivers, lakes, transitional or coastal water body), the status (ecological or chemical) or environmental objectives of which could be affected by alterations of GW level or pollutant concentrations that are transmitted through GW. The level of GW dependency of the GAAEs will likely vary between years and seasons but the critical dependency of the ecosystem on GW is key to its definition and protection” (EC 2015). As for GW dependent terrestrial ecosystems, the European Commission does not provide a strict methodology for identifying GDAEs (EC 2011a). It is up to each EU member state to develop an appropriate methodology, which takes into account local conditions and available data. Recommendations and some best practises have been provided by the EC (EC 2015).

Contrastingly to terrestrial ecosystems, there is GW contribution to most surface water bodies. The extent of the contribution varies considerably, though. There are surface water bodies that are almost completely dependent on GW such as river and stream headwaters starting from springs, lakes without surface water inflows, and karst lakes. Other surface water bodies receive GW flow only during a part of the year, or GW contributes to their water budget considerably less than precipitation, snowmelt, surface runoff and/or mire water. Therefore, the importance of GW input to the status of the surface water body may vary significantly, but the general rule is that the importance increases with the increasing contribution of GW to the total water supply of the surface water body. There are some noteworthy exceptions to that rule, though. In some cases, a relatively small GW contribution may be critical for the functioning of an aquatic ecosystem, if evaluated on a seasonal rather than annual basis. Hence, even in less productive aquifers, the GW contribution to a surface water body may be significant, for example, during periods of low flow. Taking all that into account, it is evident that identifying significant aquatic ecosystems that are critically dependent on GW, is somewhat more complicated than identifying significant terrestrial ecosystems critically dependent on GW.

Table 3.3. *Categories on groundwater associated aquatic ecosystems and examples (slightly modified from EC (2015)).*

GAAE category	**Associated water body**	**Nature of groundwater dependency**	**Examples: protected sites (Natura 2000) and others**
Temporary groundwater fed lakes	Lake	<u>Critically dependent:</u> aquatic ecology in the lake is critically dependent on the flow and chemical composition of groundwater as this is the dominant water source.	Võhmetu karst lakes in Estonia, Turloughs in Ireland.
Permanently groundwater fed lakes	Lake	<u>Critically dependent</u>: groundwater is the only source of water or contains chemicals that are critical for the ecology and are not supplied by other water sources.	Most of the lakes in the Kurtna Lake District in Estonia; clear-water oligotrophic lakes like Ninieris and Ummis in Latvia.

Lakes	Lake	Associated but not critically dependent: lakes where a significant component of their water budget comes from direct groundwater inputs, but are not critically dependent on this flow or the chemistry.	Most lakes that also have river or stream inputs.
Temporary rivers or reaches of rivers primarily fed by groundwater	River	Critically dependent: groundwater is the only or dominant source of water and the river's ecology will be damaged if this source diminished significantly.	Headwaters of Piusa and Ahja rivers in Estonia; dry riverbeds of Korkuļu, Alkšņupe, Rīteru and Klintaine rivers in gypsum or dolomite karst regions in Latvia.
Alkaline rivers – rivers with a high base flow index	River	Critically dependent: groundwater is the dominant source of water that contains chemicals that are critical for the river's ecology.	Headwaters of rivers beginning from the Pandivere Upland in Estonia
Permanent rivers	River	Associated but not critically dependent: rivers where a significant component of their water budget (on an annual or seasonal basis) comes from direct groundwater inputs (for example, during low-flow events), but where the river ecology is not critically dependent on the groundwater flow or chemistry.	Most rivers that also have surface water inputs (for example, downstream from tributaries/ headwaters, where run-off is a major water source).
The aquatic ecology within springs (surface water), not the wetland ecology associated with the spring	River	Critically dependent: the ecology within the surface water is critically dependent on the groundwater outflow	One needs to be careful to distinguish the aquatic features from GDTE spring and flush, which are focussed on the terrestrial ecology and discussed in EC, (2011a).
Temporary groundwater fed freshwater seeps on tidal flats	Transitional / coastal	Critically dependent: groundwater is the dominant source of freshwater that is critical for the surface water ecology.	The ecology depends on freshwater input from groundwater (e.g. Sylt, Germany).
Estuaries, transitional and coastal waters that receive a permanent groundwater input either directly or via rivers	Transitional / coastal	Associated, but not critically dependent: without the groundwater mediated pollution the estuary would be at good status.	Horsens estuary, Denmark.
Small spaces in the sediment of rivers, lakes and estuaries	River, lake or estuary	Critically dependent: oxic groundwater discharge through the river bed maintains the oxic and temperature conditions that are critical for the surface water ecology.	Hyporheic zone of rivers as spawning habitat for salmon and refugia for salmon fry can be essential where salmon is the *Natura 2000* protected feature; Lule river, Sweden.

In Estonia, the preliminary methodology for the identification and assessment of aquatic ecosystems dependent on groundwater bodies was developed in 2015 by the Institute of Ecology, Tallinn University (Terasmaa et al. 2015). That was before the publication of Technical Report No. 9 by the European Commission (EC 2015). Therefore, the methodology does not fully follow the categories presented in Table 3.3., but in general, a similar approach was used.

At the time of developing the methodology, there were 39 groundwater bodies in Estonia, including 13 GWBs consisting of just a Quaternary aquifer and 26 GWBs consisting of just a bedrock aquifer. Most of the Quaternary GW aquifers were not associated with any GW body. The requirements of the WFD oblige the EU member states to consider GDEs in the process of assessing the status of GWB (see Chapter 1.1), not the status of all GW aquifers. Therefore, the identification methodology covered only aquatic ecosystems dependent on delineated GWBs, not all aquatic ecosystems dependent on GW. In addition to identifying the probable critical dependence on GWBs, it was necessary to select only significant aquatic ecosystems from all possible aquatic ecosystems, as was the case for terrestrial ecosystems (see Chapter 3.1). The number of all GDAEs would be so large that it would be financially impossible to include all of them in the assessment of GWBs, because it would require collecting relevant data about all of them.

The Estonian methodology covers standing and flowing water bodies, but not transitional/ coastal waters because there is almost no information about the GW dependency of Estonian coastal ecosystems.

In the identification methodology of both GW dependent standing and flowing water bodies, the first step was to define criteria for selecting the significant ones from all Estonian water bodies, and the second step was to evaluate whether they could be considered dependent on GWBs.

For standing water bodies, it was necessary to identify temporary GW-fed lakes and to separate GW-fed lakes from lakes not receiving input from any GWBs. The separation of permanently GW-fed lakes from lakes that are GW associated, but not critically dependent (as in Table 3.3.) was largely impossible using the data that is available in Estonia. For flowing water bodies, it was necessary to separate the reaches of rivers fed by GW from the reaches of rivers not receiving significant GW inflow. As for standing water bodies, the separation of critically GW dependent rivers from rivers associated with GW, but not critically dependent (as in Table 3.3.), was generally impossible because of the lack of relevant data. Though, more data on the level of GW dependency could be used for rivers than for lakes.

Springs were not considered as separate GDAEs, as in EC (2015). They are included either as part of the rivers or river reaches that they feed or as terrestrial GDEs if they have been identified as Annex I habitat types Fennoscandian mineral-rich springs and springfens (7160) or Petrifying springs with tufa formation (Cratoneurion) (7220*) (see Chapter 3.1).

Three groups of permanent lakes were considered significant:

- Lakes in the Book of Primeval Nature. It is a national database of inanimate natural features that was compiled in the 1980s and 1990s. According to its statute, only lakes associated with GW, including karst lakes, were included in it. Therefore, all the lakes in that database were considered to be significant GAAEs. Though, often there is no indication in the database, whether the lake is feeding from the Quaternary aquifer or the bedrock aquifer beneath it. Therefore, it was difficult to decide (without any fieldwork) whether to associate lakes situated in areas with a thicker overburden, but no Quaternary GWB, with the bedrock GWB or consider them not associated with any GWB at all. In questionable cases, the likeliness of being associated with bedrock GWBs was evaluated using the geological profiles and GW heads of nearby wells and boreholes in the Environmental Register. If the lakes were situated on a Quaternary GWB, then they were automatically considered being associated with it.

- Water bodies (according to WFD). These are the lakes whose status is reported to the EC and are significant for that reason. Several water-body-lakes overlapped with the ones in the Book of Primeval Nature. From the others, bog lakes (dark and soft water – water type IV according to the Estonian classification), coastal lakes (salty water – water type VIII), and lakes with considerable surface water throughflow were discarded, as it was assumed that in the water and chemical budget of these lakes the GW component is likely insignificant. For the remaining lakes, the potential association with bedrock GWBs was estimated based on expert decisions according to GW head around the lakes. If the GW head of the uppermost bedrock GWB was deeper than the bottom of the lake, then the lake was considered not associated with the GWB. Lakes situated on a Quaternary aquifer were treated similarly to the lakes in the Book of Primeval Nature.

- Lakes listed as Annex I habitats according to the Habitats Directive. The association with GWBs of lakes that were designated to belong to a habitat type according to Annex I of the Habitats Directive (3110, 3130, 3140, 3150), and were not included in the previous two groups of lakes, was evaluated only if they were situated on Quaternary GWBs or formed lake districts under protection. Otherwise, the number of significant lakes would have grown too large. Lakes belonging to the habitat type Natural dystrophic lakes and ponds (3160) were excluded. These bog lakes were considered not receiving enough GW input to be considered GW dependent. All the other lakes situated on Quaternary GWBs were considered associated with the Quaternary GWB. Only in the case of protected lake districts that were not situated on Quaternary GWBs, the potential dependence on bedrock GWBs was evaluated, similarly to the lakes in the previous sections.

Temporary GW-fed lakes are represented by karst lakes in Estonia. Karst lakes were included in the list of significant GWB-associated lakes only according to the Book of Primeval Nature. No Estonian karst lake has been listed as a water body according to WFD. According to Annex I of the Habitats Directive, there are karst lakes in Estonia that have been assigned the habitat type Turloughs (3180). The number of such objects is very small, they are geographically unevenly distributed and do not represent the actual number and distribution of karst lakes in Estonia. Therefore, it was decided that this dataset is not considered in the identification process of significant GAAEs.

The association of all lakes, considered significant and GW dependent, with GWBs was performed based on the assumption that the lakes are associated with the uppermost GWB or not associated at all. All lakes on Quaternary GWBs were considered associated with these GWBs because the interaction of lakes with the sediments surrounding them is most likely. For lakes associated with a Quaternary GWB, the potential association with the bedrock GWB beneath it was evaluated as well.

For flowing waters, it was decided that the flowing water bodies delineated according to WFD would be considered significant. The number of WFD water bodies in Estonia is strongly skewed towards flowing waters – at the time of performing the analysis, there were 644 flowing water bodies and only 90 standing water bodies in Estonia. It means that all important flowing water bodies are listed as WFD water bodies, and there was no need to add other databases. Also, the Book of Primeval Nature does not include flowing water bodies. It contains springs and spring complexes, but these were used in the methodology for evaluating the dependence of flowing water bodies on GWBs.

As one hydrologic flowing water body usually contains several WFD flowing water bodies, the association with GWBs was evaluated for each of the WFD flowing water bodies separately and no other division was performed. Firstly, the water bodies with dark water (water type VA according to the Estonian classification) were excluded as not dependent on GW. As for standing water bodies, dark, organic rich water indicates that the water originates from peatlands and not from GW aquifers. There were some exceptions in the rule in the cases where it was evident, according to the newest monitoring data and expert knowledge, that the water type had been assigned erroneously.

For the remaining water bodies with clear water, the association with GW was evaluated based on the presence of springs. Springs are the simplest indicator for identifying GW contribution to a flowing water body if there is no other data available. Though, the magnitude of the contribution remains unknown. Springs feeding flowing water bodies are visually easier to identify than springs feeding standing water bodies; consequently, there are more springs close to flowing water bodies in the Estonian Environmental Register than close to standing water bodies. It must be considered, though, that, as in lakes, GW may seep into rivers and streams also through the bottom in a diffuse way and that is not detectable using only the data on spring locations.

The association of flowing water bodies with GW was determined using spatial analysis: dependence on GW was assumed if there were springs present in a 1 km radius of the water bodies. Some water bodies were excluded afterwards, where, according to the expert opinion, GW contribution from the spring(s) was clearly insignificant. The resultant water bodies were associated with the topmost GWB beneath the water body.

There is historical information on the share of GW in annual discharge at selected locations for the largest rivers in Estonia, but the data is more than 50 years old. Therefore, that could not be taken as the criteria for the selection. According to EC (2015), critical dependence on GW means that GW should be the dominant source of water (i.e. >50%) in a lake, stream or river. Thus, the Estonian selection is most probably overestimated, as the presence of springs does not guarantee that the origin of most of the water in the water body is GW. Similarly, the selection of permanent standing water bodies is likely overestimated, as there were no means to evaluate the actual GW contribution to them and to pick out only those that have more than 50% of the water coming from GW.

In Latvia, the identification of the GAAEs is almost identical according to the methodology used in Estonia. However, mainly due to differences in the information available on the water bodies, there were some exceptions and the methodology could not be directly adapted. In general, the following criteria were used to identify standing and flowing GAAEs in Latvia (Retiķe et al. 2021):

- Recently selected lake water bodies (LEGMC 2021) were used for standing GAAEs, except the Type L11 for shallow, dark and soft lakes with pH <5.5, as they are mainly associated with raised bogs. Type L4 for very shallow, dark and soft water lakes and Type L8 for shallow, dark and soft lakes with pH >5.5 were included only if sufficient knowledge of their inflows was available. The classification of all types of WB is described in the Regulation of the Cabinet of Ministers (19/10/2004) No. 858 "Regulations regarding the characterisation of the types, classification, quality criteria of surface water bodies and the procedures for determination of anthropogenic loads".

- Only those lakes that have not yet been described as lake water bodies (because their area is <50 ha) or are significant in Latvia according to expert judgement (for example, closed basin lakes) were taken from the list of Annex I habitats of the Habitats Directive. Habitat types 3130, 3140, 3150 and 3190 were used in this order, while habitat types such as 3160 and 3180 were automatically excluded.

- Lake water bodies were additionally classified according to their hydrological regime (lakes with flow-through, runoff and no inflow-out-flow regime) and water exchange rate. On this basis, flow-through lakes were also described as GAAEs if the water exchange rate was longer than two years.

- For flowing water bodies, the recently updated database was used (LEGMC 2021).

- In the classification of Latvian rivers, unlike in Estonia, dark water rivers cannot be excluded, because the flowing water bodies in the country are relatively huge and not homogeneous.

- Additional criteria for flowing water bodies were the average summer water temperature – if it was below +18 °C, it was concluded that the flowing water body corresponds to a cold water river with a significant inflow of GW.

3.1.3. GDTE and GDAE identification example

Līga Strazdiņa (Nature Conservation Agency, Latvia)

An example of GDTE and GAAE identification in the Gauja River Basin is shown in Figure 3.7. In the first phase, all known EU habitat types in the area were analysed. Habitats that coincide with GDTEs, such as 7160 (see also Chapter 3.1), and GAAE, such as 3130 (see also Chapter 3.2), were selected from the habitat database. In the next step, only those polygons that met the minimum criteria, such as total area, water parameters, or the presence of GDTE-related species, were filtered out using expert judgement. Many polygons did not meet these requirements and thus did not move on to the next step. For example, Lake Ungurs, although classified as habitat type 3130 with *Lobelia dortmanna* and *Isoëtes lacustris*, is a soft dark water lake with transparency of only 1.3 m, and most of the water comes from a nearby raised bog, so it was removed from further analysis. On the contrary, at the bottom of the map, many habitat 7160 polygons near the Gauja are <1 ha in size and fragmented, and it seems that they should be removed. However, these springs are connected to each other through the same GWB and are therefore considered as a GDTE complex. In the third stage, the quality of each considered GDTE and GAAE was assessed. In this example, the quality is based on the composition of the species or the stage of habitat degradation that is not always directly related to GW.

If the habitat quality was described as moderate or poor, the site had to be carefully examined to determine if it was associated with anthropogenic impacts on the GW, and therefore the entire GDE is considered to be in poor condition. In the case of GDTE, an algorithm developed by the GroundEco project can be used to (1) assess the significant damage to GDTE caused by quantitative pressure in GWB and (2) assess the significant damage to GDTE caused by qualitative (chemical) pressure (Retike et al. 2020). If a problem is identified, the site needs to take a number of improvement measures (e.g. additional monitoring, studies, site restoration, etc.). In this example (Figure 3.7.), all GDEs were in

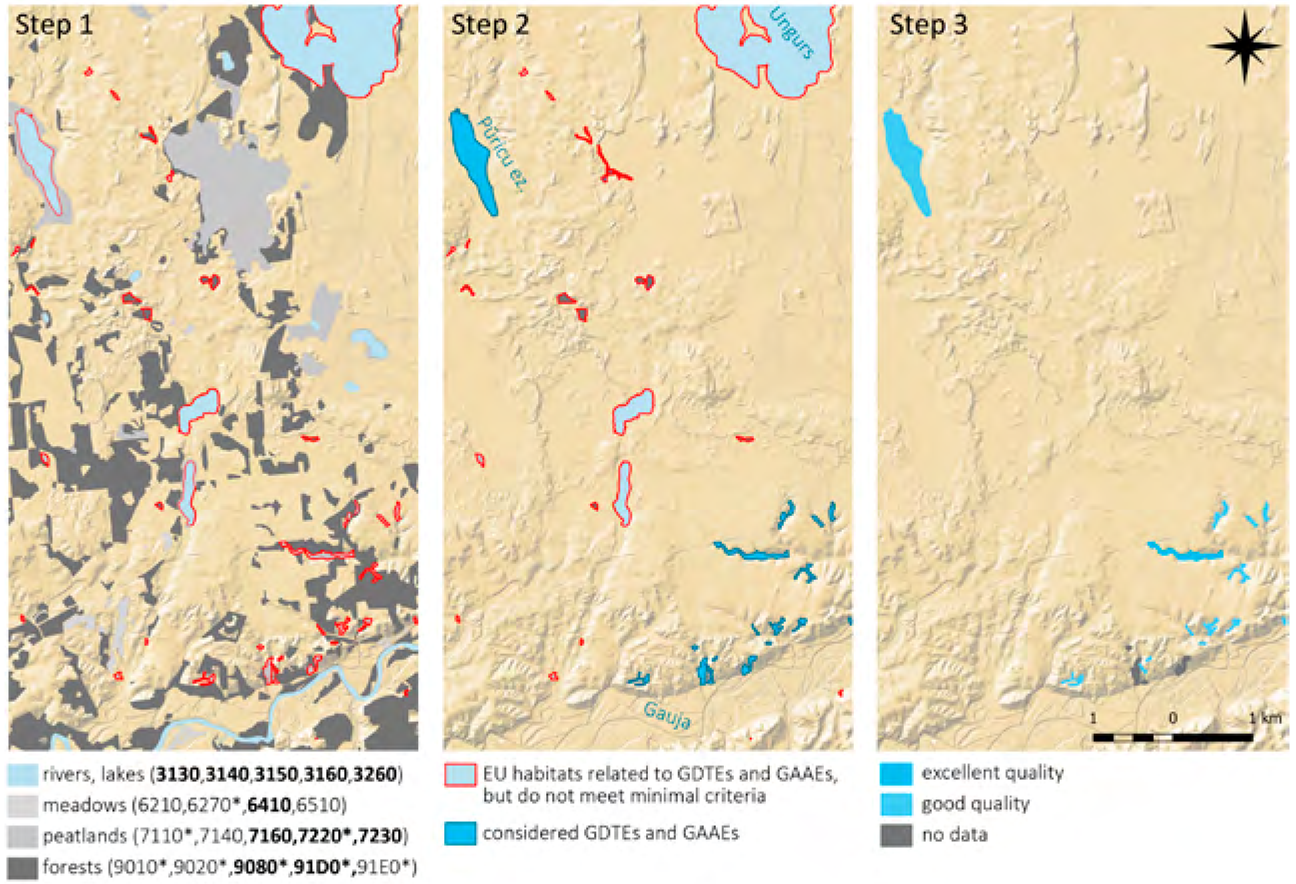


Figure 3.7. *Step-by-step process of identification of GDTEs and GAAEs for the national monitoring programme. An example from Gauja River Basin near Cēsis. Habitat types related to GW are highlighted in bold. Data source: Nature Conservation Agency, 2021. LiDAR local relief model © LĢIA.*

good or excellent condition and do not require further investigation. In total, 128 GDEs have been identified on the Latvian side of the border in the Gauja/Koiva River Basin and the Salaca/Salatsi River Basin, of which 12 are GAAEs and 116 GDTEs (WaterAct final report, in prep., Nature Conservation Agency 2021; Retike et al. 2020; Retiķe et al. 2021). In the rest of Latvia, 169 GAAEs and 288 GDTEs have been identified (Retiķe et al. 2021). On the Estonian side, 37 GDEs have been identified in the Gauja/Koiva River Basin and the Salaca/Salatsi River Basin, of which 13 are GAAEs (Terasmaa et al. 2015) and 24 GDTEs (WaterAct final report, in prep.; Retike et al. 2020). In the rest of Estonia, 324 GAAEs and 70 GDTEs have been identified (Terasmaa et al. 2015). The GDTEs in the rest of Estonia were identified using expert judgement, not the methodology described in Retike et al. (2020) (Figure 3.8.).

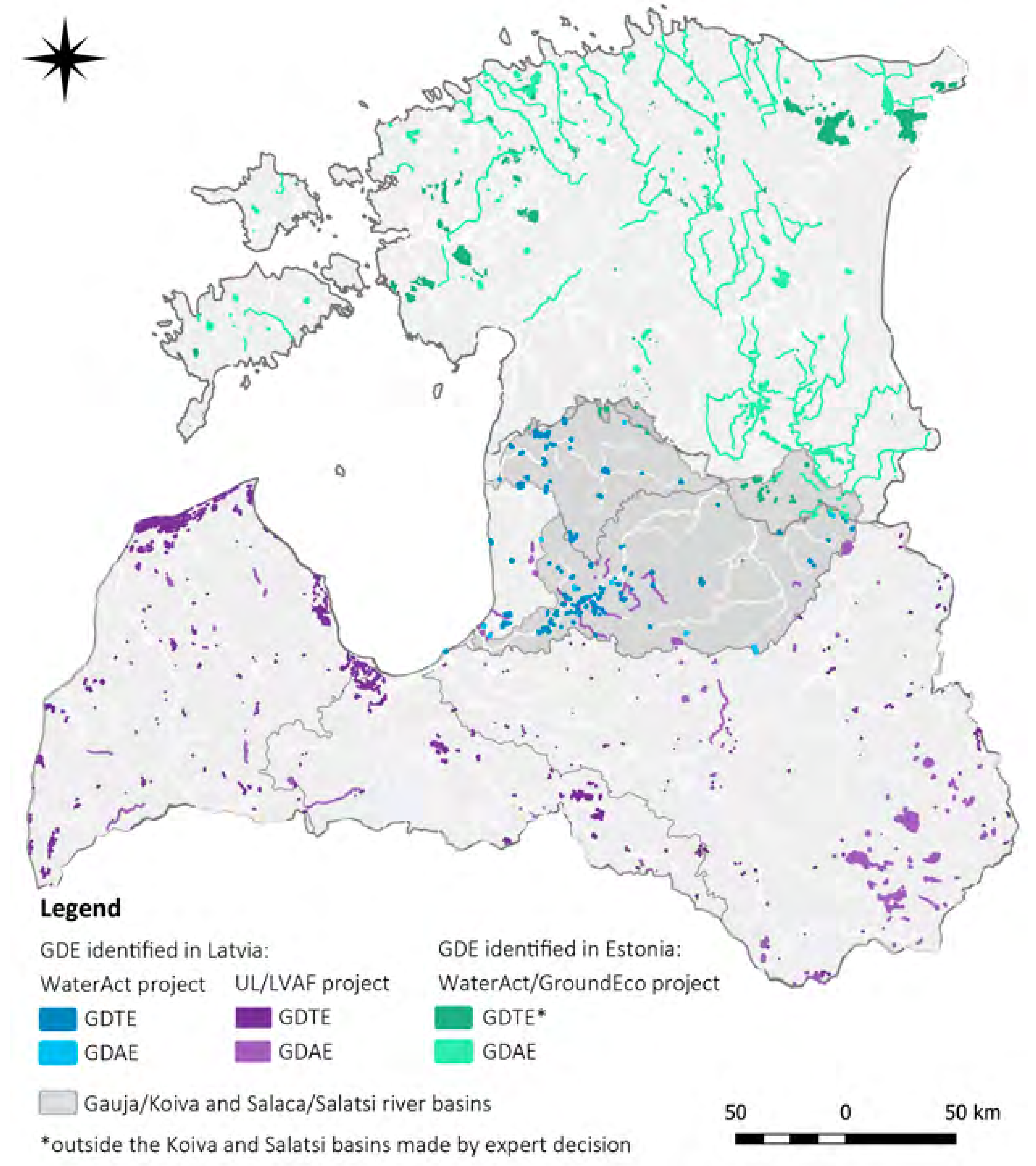


Figure 3.8. *GDEs identified in the Gauja/Koiva river basin and Salaca/Salaca river basin and in the rest of the country in Latvia and Estonia. Data source: GroundEco project (Retike et al. 2020), WaterAct project (final report, in prep.), Nature Conservation Agency 2021, University of Latvia (Retiķe et al. 2021), Tallinn University (Terasmaa et al. 2015).*

Also, the habitat quality of all GDEs in Latvia was analysed. In the WaterAct project area (Gauja/Koiva basin and Salaca/Salatsi basin), the total number of poor quality habitats considered GDEs is 26. In the rest of Latvia, 147 habitats considered GDEs have been assessed in poor condition (Retiķe et al. 2021). These results show that 30% of all identified habitats considered GDEs in the territory of Latvia are in poor condition. Many habitats considered GDEs still lack information on their quality, thus the total number of poor ecosystems could increase.

3.2. Diversity of groundwater dependent ecosystems in Latvia and Estonia

Agnese Priede, Līga Strazdiņa (Nature Conservation Agency, Latvia), Marko Vainu (Tallinn University)

3.2.1. Spring seeps, spring channels, pools, spring mires

Springs, spring seeps and channels, and spring mires are places with GW outflows connected to the GWB where the water emerges above the ground. These ecosystems can be either fed by unconfined, or confined aquifers (see Chapter 2.1.2). The spring outflows can be point-like that usually form spring channels (streams) or spring pools, with one or numerous outflows. The spring outflows can be diffuse forming wet patches (seeps or flushes), sometimes mixed with point-like outflows. Springs, spring flushes and spring mires can occur in lowlands, uplands or at the slope foots in valleys.

Spring seeps and channels (see also Chapter 2.3.1), and spring mires are highly diverse not only in terms of their location in relief, type of outflows, GW chemical and physical properties but also in biotic communities. They represent a wide range of plant communities, from patches and mats dominated by bryophytes to tall herb or small sedge communities (UK BAP 2012). They can occur in forested areas and in open landscape, either as small elements of landscape mosaic, for example, in wet grasslands or other types of mires (see Chapters 3.2.3, 3.2.3), or form spring mires that are distinct patches in the surrounding landscape.

The spring seeps are wet places where the GW reaches the surface from an underground aquifer. In Latvia and Estonia, in terms of vegetation, the spring seep communities are those belonging to Montio-Cardaminetea class that is divided into several lower phytosociological units (alliances): Cardamino-Montion, Caricion remotae (non-calcareous mineral-rich spring flushes occurring mostly in forests), Cratoneurion commutati (tufa-forming calcareous springs) (Hadač 1971; Zechmeister & Mucina 1994).

Non-calcareous spring seeps and spring channels (Figure 3.9.) occurring in forests are usually rich in bryophytes, such as *Trichocolea tomentella* (Figure 3.10.), *Plagiomnium undulatum, P. elatum, P. ellipticum, Cratoneuron filicinum, Brachythecium rivulare, Conocephalum conicum, Rhizomnium punctatum, Philonotis* spp., *Pellia* spp., etc. The most common vascular plants are *Cardamine amara, Carex remota, Myosotis palustris, Crepis paludosa, Veronica beccabunga, Epilobium palustre, Caltha palustris*, *Impatiens noli-tangere* (Paal 2007; Hájková & Hájek 2011; Ikauniece 2013a; Smieja 2014). Higher species richness is usually related with places with stable GW supply from deeper aquifer, while sites connected to shallow GW may be more prone to drought.

Water emerging from carbonate-rich aquifers can create conditions for tufa-forming spring flushes. Tufa precipitates out on contact with air forming deposits (Figure 3.11.). The formation of tufa is a complex process involving physical, chemical and biological interactions, but the essential processes start when GW receives carbon dioxide (CO_2) from the bedrock and becomes a weak carbonic acid. That dissolves calcium and carries it away as soluble calcium bicarbonate ($Ca(HCO_3)_2$). When the water emerges at the surface, it loses CO_2 in various ways, and the soluble $Ca(HCO_3)_2$ reverts to insoluble calcium carbonate which is deposited as tufa. On this inorganic substance, which is more or less constantly wet, specialised flora and fauna develop (Heery et al. 2014).

Figure 3.9. *Spring seep at the footslope in East Latvia. Photo: A. Priede.*

Figure 3.10. *Spring seep with a carpet of Trichocolea tomentella, a spring specialist bryophyte species, in East Latvia. Photo: A. Priede.*

Calcareous spring seeps belonging to Cratoneurion commutati plant community are species-rich and support many rare species. Species richness of vascular plants is usually low, it decreases as the amount of calcium carbonate precipitation increases. In such conditions, bryophytes form well-developed carpets that can significantly increase calcium carbonate precipitation due to high carbon dioxide uptake from water for photosynthesis (Hájková & Hájek 2011). The most common species are bryophytes *Palustriella commutata* (usually dominates), *Brachythecium rivulare, Bryum pseudotriquetrum, Eucladium verticillatum, Pellia endiviifolia, Philonotis calcarea, Fissidens taxifolius, Amblystegium riparium, Aulacomnium palustre, Brachythecium rivulare, Bryum bimum, Campylium chrysophyllum, P. commutata, Cratoneuron filicinum, Dichodontium pellucidum, Eucladium verticillatum, Fissidens arnoldii, Plagiomnium medium*. The herbaceous vegetation is composed of *Cirsium oleraceum, Myosotis palustris, Poa palustris, Cardamine amara, Geranium robertianum, Mycelis muralis, Brachypodium sylvaticum, Myosotis palustris, Epilobium parviflorum* (Pakalne et al. 2002; Hájková & Hájek 2011; Heery et al. 2014; Retike et al. 2020). Such spring flushes can emerge in alkaline fens, either in lowland or slope fens that usually creates a very rich, diverse vegetation pattern.

Figure 3.11. *Calcareous spring in an agricultural land in Western Latvia with active tufa formation, Cratoneuron filicinum and other calciphilous species. Photo: L. Strazdiņa.*

In comparison to spring seeps, spring mires are usually larger spring-fed areas; they are highly variable in abiotic and biotic conditions, as they are fed by GW with different physical and chemical properties and lie in different locations. The trophic conditions may vary from poor to rich. They can be at different successional stages which, in addition to the abovementioned factors, has a considerable impact on their vegetation composition and biotic communities. They can be forested (Figure 3.12.) and open, treeless (Figures 3.13., 3.14.). Pristine open spring mires are characteristic with *Menyanthes trifoliata, Thelypteris palustris, Galium uliginosum, Epipactis palustris, Stellaria crasssifolia, Festuca rubra, Poa palustris, Caltha palustris, Myosotis palustris, Rumex acetosa, Lychnis flos-cuculi, Cirsium oleraceum, C. palustre, Pedicularis palustris*. The bryophyte vegetation is usually species-rich, the bryophytes form carpets, in drier conditions *Sphagnum (Sphagnum capillifolium, S. compactus, S. squarrosum*, etc.) hummocks may be present. Typical bryophytes are *Plagiomnium ellipticum, P. undulatum, Calliergonella cuspidata, Aulacomnium palustre, Paludella squarrosa, Hylocomium umbratum, Calliergon cordifolium*. Parts of spring mires may be dominated by *Carex paniculata* tussocks or *Carex acuta*.

Figure 3.12. *Forested spring mire in Northeastern Latvia, with Menyanthes trifoliata, Equisetum fluviatile, Lychnis flos-cuculi, very rich in bryophytes. Photo: A. Priede.*

Figure 3.13. *Matsi open spring mire in Southeastern Estonia. In the front – tall sedge dominated vegetation with Carex paniculata and C. acuta, in the back – bryophyte dominated vegetation. Photo: A. Priede.*

Figure 3.14. *Vegetation pattern in Rakši open spring mire in Gauja National Park, Latvia, rich in herbaceous and bryophyte species. Photo: A. Priede.*

Figure 3.15. *Protected nature monument, geological site, Baltavots (White Spring) near Kuldīga in Western Latvia – a spring with upward leakage flow has created a spring pool up to 2 m in diameter and a spring channel. Many believe that Baltavots is a sacred place, and tie ribbons in trees around the spring. Photo: L. Strazdiņa.*

In spring pools (Figure 3.15.), the vegetation may be absent, though they may host specific algae flora. In spring brooks, the vegetation is usually species-poor: the banks may host either calcareous or non-calcareous spring flush vegetation of Montio-Cardaminetea; in the streams, there may be bryophytes, such as *Fontinalis antipyretica, Rhynchostegium* spp. and few vascular plant species, e.g. *Veronica beccabunga, Mentha aquatica*.

According to the Annex I habitat classification scheme (Paal 2007; Ikauniece 2013), depending on non-presence or presence of carbonates, the spring flushes are mapped as Fennoscandian mineral-rich springs and spring fens (7160) or Petrifying springs with tufa formation (7220*). Spring seeps may emerge in fens classified as Alkaline fens (7230), thus may be considered as part of the structural diversity in mire. Forested spring mires are often confused with bog woodlands (91D0*), though they differ both in feeding regime and vegetation. Spring seeps and mires are identified as significant GDTEs if they meet the area criteria (Table 3.1. in Chapter 3.1).

3.2.2. Fens

In a broad sense, the term "fen", originally used for mires other than bogs (van Diggelen et al. 2006), refers to a variety of ecosystems, which include minerotrophic mires, floodplain tall sedge fens and reed swamps, and forested fens – carrs or swamp forests. They all are peatlands, though they may be at different stages of development, represent different hydrogeomorphology, and, moreover, different vegetation composition. Active peat accumulation is one of the key indicators to determine ecosystem health. In fens, peat is composed mostly of sedges, reed, rushes, brown mosses.

The complexity and diversity of fens have introduced a need for classification, which again uses different criteria to classify these ecosystems. Fens are classified according to vegetation composition, base richness, productivity, pH level (Rydin et al. 1999; Wheeler & Proctor 2000; Økland et al., 2001; van Diggelen et al. 2006). The classification schemes do not always draw a clear border between GW-fed (minerotrophic), floodplain (alluvial, primarily fed by surface runoff), and ombrotrophic (rain-fed) mires. Without entangling the various approaches and setting a clear focus on the term "fen", the use of Annex I (Habitats Directive) habitat typology would introduce even more confusion. Here, in this document, we focus on minerotrophic peatlands which primarily depend on GW supply, regardless of their productivity, geochemical settings, base richness and pH. Floodplain fens and fen meadows are not considered GDTEs, as they are predominantly fed by surface runoff – these include tall sedge fens and reed swamps (Wheeler & Proctor 2000; van Diggelen et al. 2006).

Fens develop in lowlands and depressions (Figure 3.16., 3.17., 3.18.), but may occur also on slopes or lower parts of slopes (spring fens, Figure 3.19.). The fens may develop on sand, dolomites, limestone, clay. The GW flows in fens may be regional or local (Lammers et al. 2015). In the early stages of development, the fens are characteristic of a higher concentration of pH-buffering minerals and nutrient supply from the mineral-rich GW that allows the establishment of relatively rich vegetation. During the course of peatland development, the long-term succession leads towards conditions with lower pH and nutrient-poorer status (Lammers et al. 2015), consequently also with the shift of plant communities from rich or moderately rich fens to poor fens. In further stages, the fen development may lead to the development of transition mire and ombrotrophic bog or the fen may overgrow with trees leading to carr woodland (swamp forest); in rare cases, other vegetation types may develop.

Figure 3.16. *Tall sedge and reed dominated mesotrophic to eutrophic fen in a depression in Western Latvia. Photo: A. Priede.*

Figure 3.17. *Poor cottongrass and Sphagnum dominated fen in a depression between sand dunes in the Ādaži military area. Photo: A. Priede.*

During the succession, the mire complex may develop with a different speed that is affected by a wide range of factors leading to heterogeneous wetland complexes.

Figure 3.18. *Avaste fen in Estonia with alkaline fen vegetation. Photo: A. Priede.*

Figure 3.19. *Sloping fen with alkaline fen and reed vegetation, Drubazas, Western Latvia. Photo: A. Priede.*

The vegetation types and internal vegetation diversity of fens are much higher than that of bogs. That mostly depends on the richness of the mineral nutrition of the site (Rydin et al. 1999), succession stage, regional flora, climate variables, disturbance level, etc. The internal micro-relief and floral diversity include hummocks and depressions composed of different species (may include elements of bogs, such as hummocks with *Sphagnum* dominance and presence of dwarf-shrubs and some typical bog species), open mud bottoms, presence of spring seeps and spring brooks. There may be also internal diversity of fen vegetation, as it may occur on different deposits, e.g. tufa mounds with calcareous substrate and Caricion davallianae vegetation may be surrounded by nutrient-poorer deposits and tall sedge Magrocaricion vegetation.

According to the EU manual for Annex I habitat interpretation (EC 2013), only calcareous fens are considered habitats of Community interest, of which two types – Calcareous fens with *Cladium*

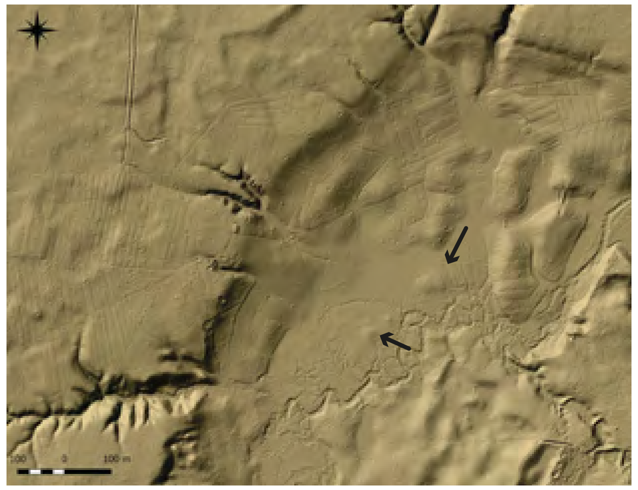


Figure 3.20. *Tufa mounds in a river valley – slight elevations. Digital relief model with 40 cm resolution. LiDAR local relief model © Latvian Geospatial Information Agency, 2019*

Figure 3.21. *Tufa mound with Caricion davallianae vegetation in a river valley, surrounded by spring-fed eutrophic wetland. Photo: A. Priede.*

mariscus and species of the *Caricion davallianae* (7210*) and Alkaline fens (7230) – are present in Latvia and Estonia. According to the manual (EC 2013), these habitat types are distinguished primarily by vegetation composition, not by hydrogeomorphology and feeding regime. Therefore, the third habitat type affiliated to calcareous fens in the EU manual, Petrifying springs with tufa formation (Cratoneurion) (7220*) represents different plant communities, although the geomorphological settings and feeding regime may be very similar to sites that are considered the habitat Alkaline fens (7230) – alkaline fens on slopes with Caricion davallianae vegetation that may develop on tufa deposits.

There are also slight differences in the national interpretations of Annex I habitat types in Estonia and Latvia. In Estonia, poor fens are considered the habitat Alkaline fens (7230). In certain conditions, poor fens and poor paludified grasslands that are GDEs by feeding regime are considered the Annex I habitat type Hydrophilous tall herb fringe communities of plain and of montane to alpine levels (6430). In Latvia, poor fens do not have any protection status, whereas poor paludified grasslands may be classified as one or another Annex I grassland type, e.g. *Molinia* meadows (6410) or the wet subtype of Fennoscandian lowland species-rich dry to mesic grasslands (6270*)[7] , see also Chapter 3.1.

3.2.3. Mixed mires

In a broader sense, fens and bogs with uncommon patterns in their hydrological or topographical characteristics containing features from more than one mire type are referred to as mixed mires. However, it should not be mistaken for a transition mire which is a separate mire type and succession stage from a grass-dominated fen to raised bog.

In the boreal zone, especially in Sweden, the mixed mires are divided into two groups. String mixed mires have well developed, 10–40 cm high damming ridges or strings with hummock vegetation e.g. *Sphagnum fuscum, Andromeda polifolia, Betula nana, Empetrum nigrum* and *Rubus chamaemorus* and narrow depressions or flarks with lawn or carpet vegetation e.g. *Carex* spp., *Molinia caerulea,*

[7] In Latvia and Estonia, deeper analysis of comparing the national Annex I habitat interpretations concerning semi-natural grassland types is not done.

Sphagnum balticum, S. angustifolium and low dwarf shrubs such as *Betula nana* (Rydin et al. 1999; Breeuwer et al. 2008; Jeglum et al. 2011). In boreal mixed mires, there are also sharp differences in nutritional status. While *Sphagnum* hummock islands behave as true miniature bogs, the adjacent lawns are influenced by minerogenous influxes from the catchment (Gerdol 1990).

Figure 3.22. *Raganu Mire (on the left) and Zaļais Mire (on the right) in Ķemeri National Park, Latvia, represent features of the mixed mire. The complexes host raised bog, transition mire, poor fen, rich fen vegetation, and alkaline sulphurous springs with Schoenus ferrugineus dominated plant community. Photos: L. Strazdiņa, A. Priede.*

Another mixed mire type has a more irregular mixture of small bog and fen elements and can be referred to as mosaic mixed mire, and they have no marked structures. Bog elements are irregularly distributed on a fen lawn, or mixed with fen areas in a mosaic (Rydin et al. 1999; Gunnarsson & Löfroth 2009). Examples from Latvia of such mixed mires are Raganu Mire and Zaļais Mire in Ķemeri National Park (Figure 3.22.). The landscape of these mires is very diverse. The dominant plant communities in both raised bogs, Raganu Mire and Zaļais Mire, are Sphagnetum magellanici, Scheuchzerio-Sphagnetum cuspidati, Caricetum limosae, Caricetum rostratae (Pakalne 1998), while Cladietum marisci and Schoenetum ferruginei communities form island-like structures in the GW-fed patches with spring outflows. In a small area, several spring pools filled with alkaline sulphurous water occur, with calciphilous fen vegetation (Priede 2017). Other examples of spring discharges in raised bogs, fens or transition mires in Latvia are Dūņezera Mire (Figure 3.23.), Ķemeri Mire, Veseta Floodplain Mire, and some others.

Figure 3.23. *Dūņezera Mire, a small raised bog with a spring outflow and transition mire elements. Photos: A. Priede.*

3.2.4. Swamp woods

Despite the significant role of SW in regulating the water regime of the ecosystem, swamp woods of black alder, ash, downy birch and spruce that are flooded continuously or annually are also considered GDTEs if they meet the area criteria (one polygon of 20 ha or a complex of more than one smaller patches with the total area of at least 20 ha; see also Chapter 3.1). Swamp woods provide a multitude of ecological niches along a wide hydrotopographical gradient, ranging from dry hummocks and tree bases to permanently water-filled hollows (Hörnberg et al. 1998). In Latvia, most of them represent the habitat Fennoscandian deciduous swamp woods (9080*) listed in Annex I of the Habitats Directive, however, it can be applied also to boreal spruce swamp forests (Figure 3.24.). Usually, the swamp wood habitat patches are fragmented and scattered or strained around geological formations (e.g. in wetter inter-dune depressions) or along river floodplains (Auniņš (ed.) 2013). Swamp woods are often located on the margins of raised bogs at the GW discharge zone (see also Chapter 3.2.5).

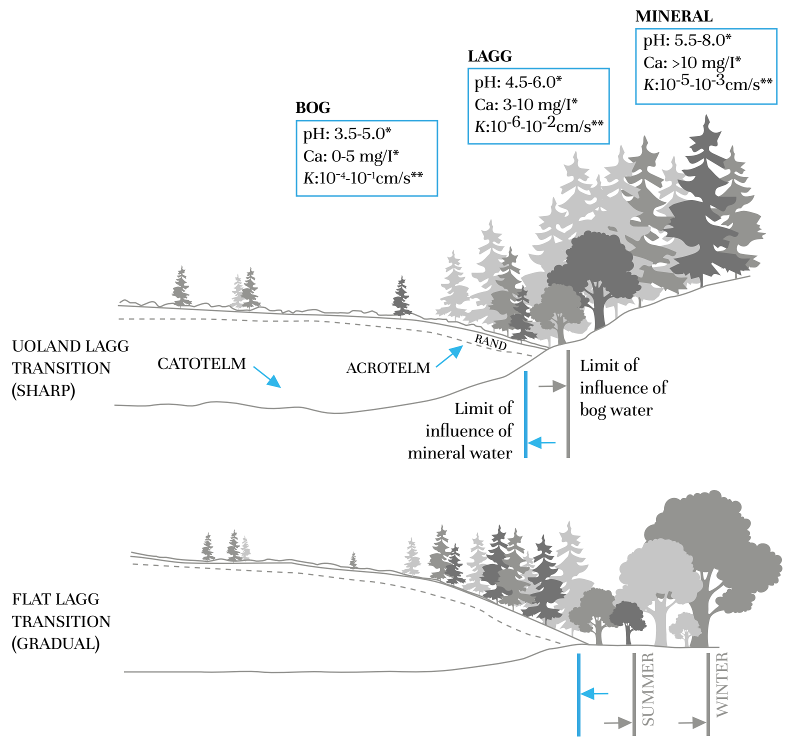
Figure 3.24. *Swamp woods with black alder in Western Latvia and with Norway spruce in Gauja National Park, Latvia. Photo: L. Strazdiņa (left), M. Pakalne (right).*

Reasons for fluctuations of water level in swamp woods can be moving water near the surface (typically in slopes often with several springs and small brooks), seasonal flooding in depressions and in streams or along lakes, as well as precipitation peaks during spring or autumn. The flooding regime, mainly duration, depth and frequency determine the vegetation composition in the habitat. These woods are exceptionally important for many species: the vascular plants (Ingerpuu et al. 2001), wood-inhabiting fungi (Kunttu et al. 2016) and the cryptogams like many desiccation intolerant liverworts (Darell & Cronberg 2011) who depend on forest continuity, high humidity and structural diversity of microhabitats. In Sweden, an inventory of 10 small remnants of old-growth spruce swamp forests that together had an area of 20 ha yielded a total of 517 species of vascular plants, bryophytes, lichens, and wood-inhabiting fungi (Ohlson et al. 1997). Bryophyte cover in swamp woods is usually higher than in open ecosystems due to low physiological stress throughout the summer season as a result of shaded conditions (Maanavilja et al. 2015).

Long flooding periods and submersion in stagnant and eutrophic waters in swamp woods can result as too slow diffusion of CO_2 and low assimilation, deposition of litter and sediment, though the litter layer in black alder forest is thin as compared to forests of other deciduous tree species. The vascular plants have to deal with oxygen deficiency, leading to mosaic patterns of vegetation communities; the dominating species are *Urtica dioica*, *Filipendula ulmaria* and *Athyrium filix-femina*. Plants and epigeic bryophytes typically occupy open patches only if long periods of dry weather with low water level occur. The bryophytes are often zonated as follows: usually no species in the stagnant water; a few at the edge of open water patches (for example, *Calliergon cordifolium*, *Sphagnum squarrosum*, *Plagiomnium ellipticum*), more species at the banks of the streams and sometimes rather many at the flooded part of the stools of black alder (some hepatics such as *Pellia epiphylla*, *Scapania nemorea* and *Chiloscyphus polyanthos*). Higher up on the black alder stools and on the bark there are again few other epiphytic species (Darell & Cronberg 2011). Snowmelt pools make up a well-defined and important habitat for many aquatic insects in the swamp forest (Hörnberg et al. 1998).

3.2.5. Laggs at the edges of raised bogs

According to the joint Estonian-Latvian GDTE identification methodology (Retike et al. 2020), in most cases ombrotrophic (rain-fed) bogs in Latvia and Estonia are not considered GDEs, because they are fed by precipitation. However, the water in the peat layer may not be fully isolated from the GW aquifer. A drop in GW level may cause a drop in the bog water level if the peat layer is not separated from the aquifer by a continuous impermeable aquitard (Retike et al. 2020; see also Chapter 3.2). For this reason, bogs may be considered GW dependent, though not directly, in specific conditions (see Chapter 3.1).



* Balfour and Banack 2000; Glaser 1992; Bourbonniere 2009
** Baird et al. 2008, Lapen et.al. 2005, Rydin and Jeglum 2006

Figure 3.25*. Cross-section of two lagg forms – upland and flat (modified from Howie & van Meerveld (2011)).*

Raised bogs may have lagg zones at the edges, which are connected to GW and play an important role in the "viability" and development of the bog. They are part of bogs, though not always present or well-pronounced, and play a crucial role in the "health" of the bog ecosystem.

Originally, "lagg" is a Swedish word that refers to wet margin around the raised bog, also called lagg fen. This zone at the edge of the raised bogs collects water from the ombrotrophic bog and adjacent mineral soils (Rydin et al. 1999; Howie & van Meerveld 2011; Lindsay 2016). Water from precipitation received on the bog dome is lost through evapotranspiration, is stored in the peat body of the bog (catotelm), or moves through the upper living layer of the bog (acrotelm) to the marginal area and then is being discharged in the lagg zone where it merges with local GW table (Lindsay 2016). Laggs are fed by mixed water from the GW and from the bog runoff, therefore, in contrast to the bog itself, are considered GW dependent (minerotrophic). Laggs are characteristic with higher pH and electric conductivity than in the central part of the bog. In contrast to the central part of the bog and bog dome slopes, supply of GW ensures higher nutrient availability and consequently higher productivity than in the bog itself.

Figure 3.26. *Moat-like lagg in the eastern part of Ķemeri Mire in Latvia, a very wet lagg fen with some open water pools is constrained by inland dunes. The bog expansion is limited by relatively steep slopes. Orthophoto: © Latvian Geospatial Information Agency, 2019. Photo: J. Matuko.*

The key function of lags is sustaining the water in the peat body that is supported by high water level in the lagg, and, at the same time, laggs ensure natural drainage of excess water from the bog dome (Howie & van Meerveld 2011). Artificial drainage and peat extraction in the lagg zones and adjacent lands on mineral soils on the bog margin affects the basal level of the entire raised bog (Lindsay 2016). In this sense, bogs are indirectly groundwater dependent.

Artificial drainage of the laggs and/or the adjacent zone on mineral soils cause tree encroachment, consequently causing higher evapotranspiration, increased desiccation of the bog surface, expansion of trees toward the central part of the bog. Draining the laggs and nearby areas may severely disturb the natural self-sustaining regulation function and induce overall degradation of the ecosystem with all further consequences (e.g. decline for *Sphagnum* cover, disturbed peat development, altered carbon sequestration and water regulation functions). Sometimes laggs and consequently the functions of the bog are damaged also by peat extraction in the past by establishing peat quarries (pits) in the marginal zone that usually have similar effect as drainage ditches.

Laggs play an essential role in the development of bogs. The character of bog expansion at horizontal scale, i.e. bog expansion toward margins is primarily influenced by the surrounding relief near the lagg. If the bog margin (lagg) is surrounded by well-pronounced slopes, the bog expansion is very limited. In flat areas where bogs have formed in shallow depressions, the bog expansion zone may be wide and the neighbouring mineral soils are being gradually overwhelmed by the paludification process (Rydin et al. 1999; Lindsay 2016).

Figure 3.27. *Moat-like lagg at the Eastern edge of Ķemeri Mire, bordering with steep dunes. Photo: A. Priede.*

Figure 3.28. *Diffuse lagg (greener at the bottom of the image) in a raised bog located in a gently sloping area on the margin of Teiči Bog. Photo: J. Matuko.*

Laggs constrained by steep slopes (Figure 3.25.) are usually well-pronounced, very wet, with open water pools; the vegetation is dominated by fen plants and floating *Sphagnum* mats, often with reeds and sedges, usually that is vegetation typical for poor fens (Figure 3.27.). Diffuse laggs in flat or gently sloping areas are drier and the vegetation resembles transition mires and bogs (Figure 3.26., 3.28.). The water level is fluctuating, and the fluctuations, though naturally occurring in the entire bog, are sharper at the bog edge in the fen, because the water is collecting there both from precipitation and runoff from the bog and from the adjacent lands on mineral soils (Howie & van Meerveld 2011).

In habitat inventories using the Annex I habitat classification scheme (Paal 2007; Auniņš (ed.) 2013), laggs are considered part of raised bogs (Active raised bogs (7110*)) or mapped as Transition mires and quaking bogs (7140), though they have different hydrological regime and water supply. Nevertheless, they are in fact GDTEs, although in practise are not mapped as separate habitats and thus would not be selected as GDTEs according to the GDTE identification methodology (see Chapter 3.1). Since the GDTE identification methodology for Estonia and Latvia (Retike et al. 2020) is based on Annex I habitat maps (the first step in GDTE selection), identifying laggs are separate GDTE units is not possible, unless special mapping is done or the GDTE selection is supplemented with additional remote sensing data or other relevant data.

3.2.6. Wet dune slacks and coastal ecosystem complexes

The coastal zone of Latvia is characterized by specific habitats formed due to seawater regression and high GW levels approximately 4700 years ago in the inter-dune depressions in former Littorina Sea lagoons (Kalnina et al. 2014) (Figure 3.29.). The GW in these inter-dune mires possibly follows cascading systems similar to other inter-dune wetlands in Northwestern Europe (Elshehawi et al. 2020). The sandstone aquifer occurs at a depth of about 20 m below the surface (LEGMC 2018).

Figure 3.29. Wet dune slacks: the pioneer stage (Slītere National Park, Northwestern Latvia) and an inter-dune transition mire at the Gulf of Rīga near Carnikava, Latvia. Photos: A. Priede.

Research of both the temperature and the electrical conductivity of GW in valley mires in Slītere National Park in Northwestern Latvia shows that each side of the mire differs considerably (Figure 3.30.). While one side receives mineral-poor and warm GW in the upper layers through infiltration (EC <15 mS/m), the opposite side has mineral-rich cold GW after discharge (EC >25 mS/m) (Wolejko et al. 2019). Despite the fact that these mires have a mixed water supply, they still correspond to the described GDTE ecosystem with its typical vegetation and habitat complex.

The regional base-rich GW discharges at the inter-dune mires, with a contribution from the local systems, creates an intermediate mixed system with high vegetation biodiversity (Figure 3.31., 3.32.). The inter-dune mires of Slītere National Park represent the wide range of mire types. Such a large variation of near-natural ecological mire types concentrated in a rather small area is nowadays very rare in Europe. Dominant vegetation communities are Caricion davallianae, Saxifrago-Tomentypnion and

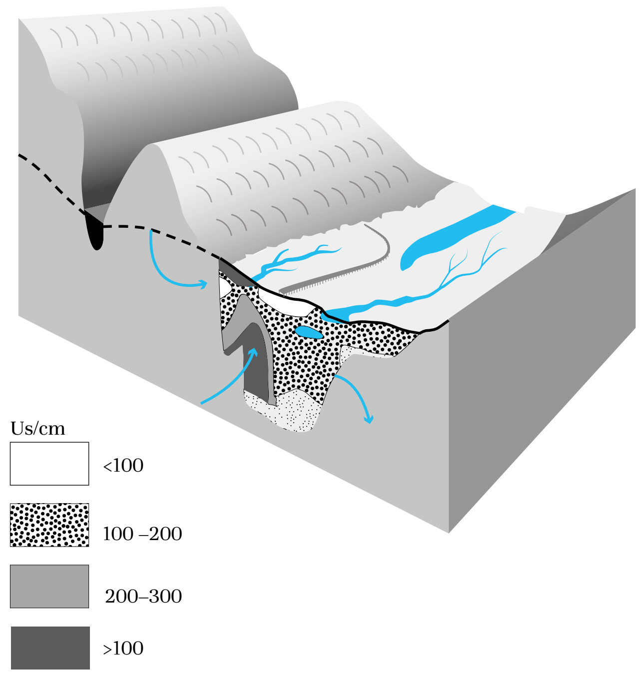


Figure 3.30. The electrical conductivity (EC) reflects the amount of dissolved minerals in the inter-dune mire in Slītere National Park. High values point to the presence of calcareous groundwater, very low values – to rainwater. Conceptual model modified from Grootjans & Wolejko (2016).

Stygio-Caricion limosae, which consist of so-called dystrophic hollow communities of Scheuchzerion palustris (Wolejko et al. 2019). Primary succession in dune slacks can be roughly divided into four phases: (1) a phase in which microbial mats and algae are dominating and where the accumulation of organic material is low; (2) colonization by phanerogams which are adapted to low nutrient availability; (3) development of a moss layer of pleurocarpic bryophytes and invasion by tall grasses and shrubs; (4) rapid accumulation of organic matter, partly due to acidification of the top layer which leads to replacement of poorly competitive plant species by shrubs and trees (Grootjans et al. 1997). The composition of vegetation in moist to wet dune slacks depends on the constancy of the annual water table fluctuation, the relative shortage of nutrients in the soil and in the GW, the gradients of wet and dry in the slacks, succession stage, and the scant development of shrubberies and forest under the influence of seawind and grazing.

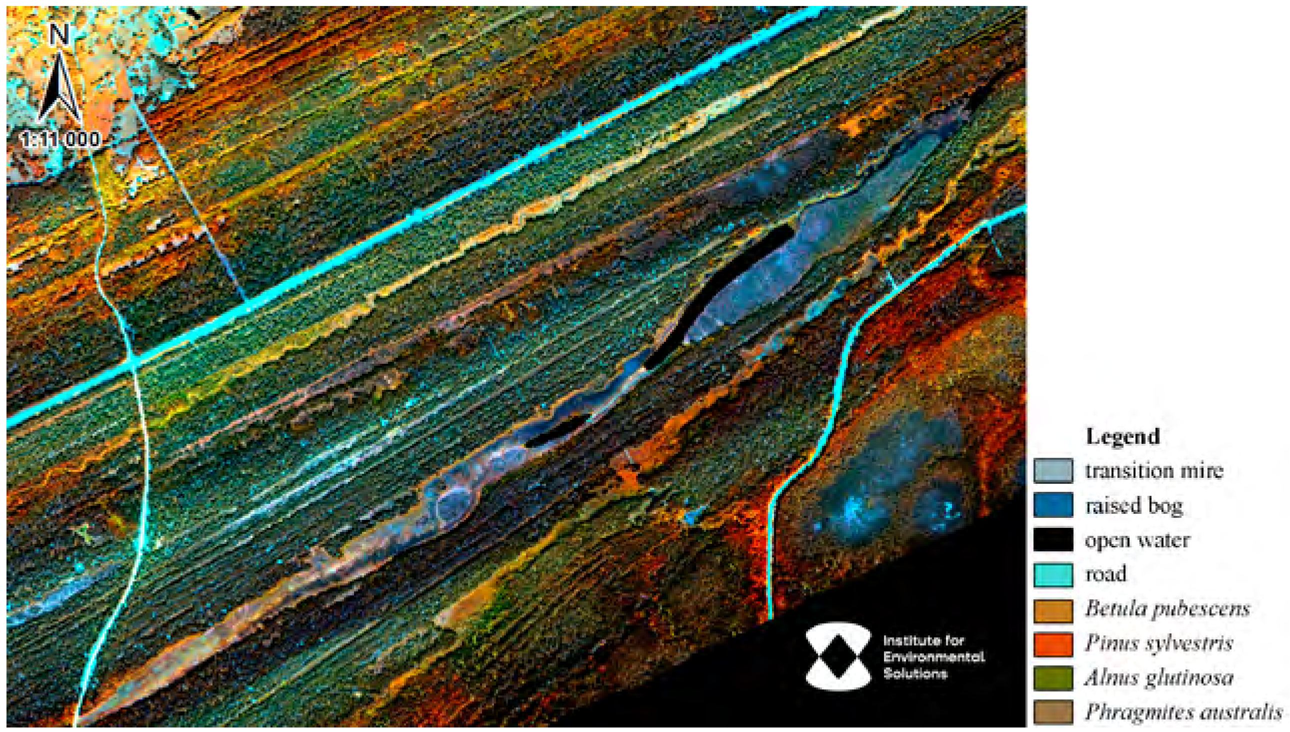


Figure 3.31. Hyperspectral data of Pēterezers inter-dune mire complex and its surroundings in Slītere National Park, Latvia. Each colour represents different vegetation communities whose composition depends on hydrological conditions, peat/soil layer properties, relief and management history. Map © Institute for Environmental Solutions, 2018.

Figure 3.32*. In Slītere National Park, a mixture of different forest habitats, often with springs, has developed on drier and older dunes, while the wet depressions are still taken by mires, fens and wet dune slacks. Photos: K. Libauers.*

The dune systems are very vulnerable systems and are strongly affected by very minor influences, such as a slight increase in water extraction, the digging of run-off ditches in source areas, or increasing evaporation from woods (van Dijk & Grootjans 1993).

3.2.7. Groundwater dependent aquatic habitats

Standing, flowing and coastal waters may all be GW dependent (see Chapter 3.2). Critical dependence is considered in situations where GW provides most of the water in the water body or GW input is critical for its chemical composition. In Estonia and Latvia, there is better knowledge on GW dependent lakes and rivers than coastal waters, therefore the following chapter describes different types of critically GW dependent habitats of the first two.

Karst lakes and other temporary groundwater-fed lakes

Karst lakes are temporary lakes that form in depressions that have karstic origin, inflow and/or outflow. According to Annex I of the Habitats Directive, they are considered under habitat Turloughs (3180*). Karst lakes may be fed both by surface water or GW. Surface water-fed karst lakes form during wet periods in sinkholes if surface water inflow exceeds the acceptance rate of the ponors in the bottom of the sinkhole (Figure 3.33.), or there are no ponors in the bottom of the sinkholes at all. That type of karst lakes should be considered GW dependent as well, as their existence often depends on a high GW level that reduces the infiltration or inflow of surface water to the karst system. "True" karst lakes are fed by GW, though. These lakes form in closed depressions in karst areas during periods of high GW level, usually in springs after snowmelt, and disappear after the GW level drops, usually in the middle of the summer. GW inflow and outflow occurs through sinkholes, which act as springs during the filling phase and as ponors during the emptying phase.

The lengths and widths of true karst lakes may range from a couple of hundreds of meters to a couple of kilometers. Karst lakes are shallow, their depth seldom exceeds two meters, except in locations of sinkholes. Water is very transparent, clear and alkaline (HCO_3 - >200 mg/l) (Paal 2007). During the dry phase, the lake areas are usually covered by moist grasslands or *Salix* dominated shrub communities (Figure 3.34.).

Figure 3.33*. Surface-water fed karst lake during a dry and wet stadium. During high flow, the discharge of River Salajõgi in Western Estonia exceeds the acceptance rate of the Salajõe karst system and a temporary karst lake is formed at the location of sinkholes. Photos: Estonian Land Board.*

Figure 3.34*. Einjärve groundwater-fed karst lake during a dry and wet stage, located in the Pandivere Upland in Central Estonia. Photos: M. Vainu.*

Due to their dual nature, karst lakes act both as aquatic and terrestrial habitats, and on average years the dry phase lasts markedly longer than the wet phase. Therefore, there is considerably better knowledge on the vegetation of karst lakes, than their aquatic fauna. Due to their temporary nature, there are no fish in karst lakes and also very little phytoplankton, but the zooplankton communities in some Estonian karst lakes contained rare species in the 1960s and 1970s (Mäemets 1977). In the 21st century, there has been only one hydrobiological study in a single karst lake with published results in Estonia, and then these rare zooplankton species were not found (Ott 2010). The vegetation in karst lakes consists of grassland and mire species (Figure 3.35.), with occasional aquatic species that survive the dry phase in deeper and wetter hollows. What is typical to karst lakes is that vegetation starts to develop considerably later there, than in non-inundated areas, therefore in July or August the plants may be behind in development for more than a month, compared to surrounding areas. In terms of plant communities, various grassland types may grow in karst lakes, depending on the length of the inundated period and the composition of the overburden. A specific type of karst meadow

***Figure 3.35**. Meadow community in the bottom of a karst lake during a dry phase. A sinkhole is visible in the middle of the picture. Photo: M. Vainu.*

has been identified in Estonia with *Viola-Potentilla* or *Phalaris* communities, in addition to that dry, paludified, floodplain, and alvar meadow communities may exist in karst lakes. Typically to karst lakes, the bryophyte layer is often absent or is species-poor compared to analogous communities in non-inundated areas (Vainu et al. 2019).

***Figure 3.36**. Temporary groundwater-fed Lake Koltsi on the island of Saaremaa. It receives water from the surrounding Quaternary aquifer. Therefore, it cannot be considered a karst lake, but is still critically groundwater dependent. Photos: Estonian Land Board.*

In Estonia, karst lakes are located in the northern and western parts of the country, where the bedrock is formed by Ordovician and Silurian limestones. Few and less studied karst lakes are also present in the south-easternmost part of the country, in the area of Devonian limestones. Karst lakes in Latvia are not GW-fed temporary lakes, therefore they represent spring-fed lakes in the gypsum-karst regions described in the following section.

Temporary GW-fed lakes may receive their water also from Quaternary aquifers and though their hydrological regime resembles that of karst lakes, these should not be called that way, because they are not connected to karstified bedrock layers. These lakes are very rare in Estonia. In Estonia, they are represented for example by Lake Koltsi on the island of Saaremaa (Figure 3.36.).

Spring-fed permanent lakes

Spring-fed lakes have formed to the locations of limnocrene springs (see Chapter 2.3), where landscape topography is such that water cannot flow away from the spring locations freely, but has inundated the surrounding depression. The border of what to consider as a solitary limnocrene spring (see Chapter 2.3.1) and what as a spring lake is not fixed and therefore may be considered

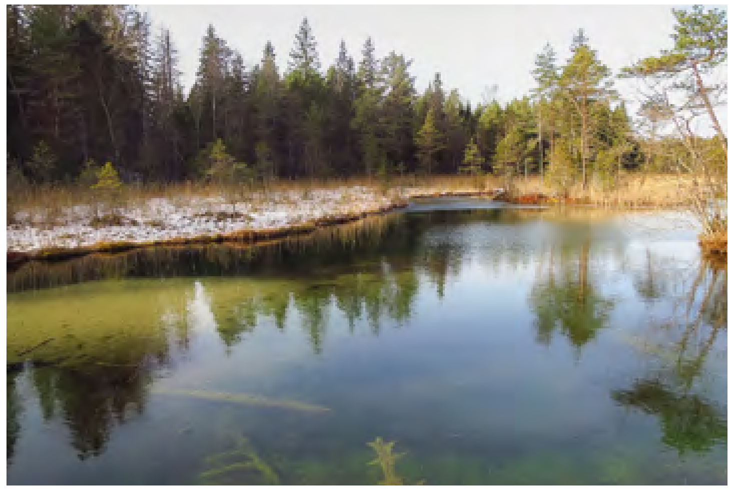


***Figure 3.37**. Small natural Aegviidu Siniallikate spring-fed lake in Estonia with an area less than 1 ha. Photo: M. Vainu.*

vague. Generally, the minimum limit of a pool of water to be considered a lake is 1 ha, but considerably smaller lakes (including spring lakes) have been included in the Estonian Environmental Register in the list of lakes as well. Thus, it may be said that a spring lake is a pool of water fed by springs that feels too large to be called just a spring (Figure 3.37.).

Spring-fed lakes have no or small surface water inflow, but they have surface water outflow that on average is considerably larger than the inflow. The outflow is largest during the periods of high GW level and decreases during periods of low GW level. If there is no surface water inflow to the lake, then the outflow may also be temporary, during drought periods with low GW level (Figure 3.38.). To be considered a critically GW dependent lake, GW has to be the dominant source of water in the lake. A good proxy would be comparing average annual surface water inflow and outflow and if the outflow

Figure 3.38. *Permanent spring-fed lakes may reduce in size considerably during drought periods, but such low-water periods happen rarely and do not last long enough to allow meadow vegetation to grow on the bottom of them, as in temporary karst lakes. Jõepere spring lake in Central Estonia. Photos: M. Vainu.*

is at least two times higher than the inflow, then the lake may be considered a critically GW dependent lake. Though, all lakes with the presence of outflow, but no or small inflow, can not be considered spring-fed lakes automatically. A similar pattern is typical to dystrophic brown- and soft-watered bog lakes as well, to where water from the bog massif accumulates and from where bog water flows out via a stream. Therefore, an additional feature to distinguish a spring-fed lake is clear water (color <100° on the Pt-Co scale). There may be more complex cases, where a semidystrophic lake is fed both by mire water and springs, and, as a result, the color of the water is yellowish. In such cases, the share of GW has to be determined using chemical tracers in order to classify a lake critically GW dependent.

Fully spring-fed natural lakes in Estonia are most often connected with karst areas (alkalitrophic lakes) and are located in regions with a permanent GW outflow from the karst system. Several (probably even more than there are natural ones) such lakes have been constructed artificially by damming the flow of spring-fed streams (Figure 3.39.). In other regions, lakes that are only spring-fed are very rare and mixed feeding patterns are more common. Spring-fed lakes in karst areas are small (a couple of hectares) and have clear, transparent and alkaline (HCO_3^- >240 mg/l) water (Figure 3.40.). Their water has a high content of calcium compounds, which react with phosphorus compounds and make them difficult to absorb. So, even if the nitrogen content in these lakes is relatively high, there is not enough available phosphorus for phytoplankton, which contributes to the high transparency. Due to the relatively constant temperature of GW, they rarely freeze in winter and stay cold during

Figure 3.39. *Dammed spring-fed lakes may look morphologically similar to natural ones but tend to grow full of filamentous green algae if located in populated places close to fields. Photo: M. Vainu.*

Figure 3.40. *Natural spring-fed lake Äntu Sinijärv has the highest transparency of Estonian lakes (>6 m). Photo: M. Vainu.*

summer. Their vegetation contains a low number of species, but high biomass: *Chara* dominated communities dominate, which can dissolve the inactivated phosphorus compounds. Also, the fish communities host a low number of species (mostly pike, perch and roach). According to the Habitats Directive, they belong to the habitat type Hard oligo-mesotrophic waters with benthic vegetation of *Chara* spp. (3140) (Paal 2007).

Spring-fed lakes where the GW content is smaller, but still above 50%, may belong to various lake types according to all existing lake classifications (national ones, based on the Habitat Directive or based on WFD), depending on the chemical composition of the GW aquifer that feeds them and the share of GW in their water budget (Figure 3.41.). Therefore, it is not possible to describe their

Figure 3.41. *Large (111 ha) Lake Raigastvere in Eastern Estonia (habitat type Natural eutrophic lakes with Magnopotamion of Hydrocharition-type vegetation (3150); Water Framework Directive's type III – stratified lakes with moderate alkalinity) has been shown to receive less than 50% of its annual surface water outflow from surface water inflow (Palo & Ott 2020), therefore it has to be considered as a critically groundwater dependent spring-fed lake. Photo: M. Vainu.*

typical characteristics. Though, spring-fed lakes are not to be found among lakes classified as Natural dystrophic lakes and pools (3160) according to the Habitats Directive, type IV (lakes with soft and dark water) in Estonia or type IV, VIII and XI (very shallow and shallow lakes with soft brown water) in Latvia according to WFD, and dystrophic according to national classifications.

Closed-basin clear-water lakes

Closed-basin lakes have no surface-water outflow and usually also no channelized surface water inflow (Figure 3.42.). This type of lakes may be located in various types of geological settings, but in Estonia and Latvia are most common in areas with thick Quaternary sediments, both glacial and postglacial. Closed-basin lakes in postglacial peat deposits should not be considered as critically GW dependent under normal circumstances, as their water originates from and their water level is controlled by the surrounding bog massif. Mire water is not considered GW. They may be classified as GW dependent in exceptional cases if they are located close to or above mines, which lower the GW level below the bog considerably. As these activities may result in a decrease in the bog water level, the water level of such lakes may be affected as well. But generally, dystrophic brown-watered closed-basin lakes classified as Natural dystrophic lakes and pools (3160) according to the Habitats Directive and type IV (lakes with soft and dark water) in Estonia or type IV, VIII and XI (very shallow and shallow lakes with soft brown water) in Latvia according to WFD are not GW dependent.

The water level in closed-basin lakes situated in mineral sediments (most often sands, but also gravel, till) is controlled by the GW level in the surrounding aquifer. They are basically reservoirs of GW that happen to be above the ground because the lake depression extends below the GW table (Figure 3.43.). Therefore, such lakes have GW inflow from some part of their bed, and their water

Figure 3.42. *Lake Martiska in Northeastern Estonia is a typical closed-basin clear water lake located in sandy terrain. Photo: M. Kohv.*

seeps into GW from another part – GW flows through them. As they have no surface water outflow, the amount of GW seeping into them can not be higher than the amount of GW seeping out of them. The latter has to also compensate for the amount of water the lake receives via surface runoff and the amount of direct precipitation, which has not evaporated from the surface. For that reason such lakes are usually not fed by confined aquifers, which would provide them with too much GW inflow, therefore there are no large limnocrene springs with visible boiling points in the bottom of them as

Figure 3.43. *Closed-basin lakes have formed to locations where the bottom of depressions shaped during the last glacial retreat are below the groundwater table and therefore have filled with near-surface groundwater. Excerpt from the Estonian Base Map by the Estonian Land Board.*

Figure 3.44. *Groundwater seepage into closed-basin lakes becomes visible during winter, as the areas with the highest seepage are not frozen. Photo: M. Vainu.*

well. Generally, there are no visible springs in or on the shores of such lakes at all, which is the reason why they are treated separately from spring-fed lakes. GW seeps into such lakes in a diffuse way. The inflow may be detected in winter, when the lakes are frozen over, but some parts of the shore remain open (Figure 3.44.), or during summertime, when walking in the shallow near-shore water and parts of the lake bottom feel considerably colder than the surroundings

Closed-basin clear-water lakes may belong to various lake types, but often they are oligo- or semidystrophic and may be found under Habitats Directive habitat Oligotrophic waters containing very few minerals of sandy plains *Littorelletalia uniflorae* (3110) and under WFD lake type V (lakes with soft and clear water) in Estonia or types III, VII, X (very shallow, shallow and deep lakes with soft and clear water) in Latvia. Such lakes are surrounded by nutrient-poor Quaternary sediments (mostly sands) and therefore feed on near-surface nutrient-poor GW which develops in close surroundings. For that reason, the lakes are or at least have been until the last three quarters of a century low on mineral and biogenic substances, their transparency is high (≥ 4 m) and they host rare species of plants and zooplankton. Such lakes are the last resort e.g. for *Lobelia dortmanna* (Figure 3.45.), *Isoëtes lacustris*, *Sparganium angustifolium*, *Myriophyllum alterniflorum*. The bottom of such lakes may be covered by mosses, especially from the genus *Fontinalis*. The fish fauna is usually rather poor, consisting mostly of pike, perch and roach.

Figure 3.45. *Lobelia dortmanna is a rare species that grows in clear-water oligotrophic or semidystrophic lakes, which as a rule are critically groundwater-dependent, whether spring-fed or closed-basin. Photo: U. Suško (left), A. Soms (right).*

In Latvia, the permanent closed-basin groundwater-fed lakes found in the gypsum-karst regions are classified under EU habitat Lakes of gypsum karst (3190*). Most of these lakes are located near the border with Lithuania in Skaistkalne (Figure 3.46. and 3.47.). The largest sinkholes are 10–12 m deep, while many reach only 0.5–1 m in depth. Some of these karst lakes have hydraulic connection to the Salaspils aquifer and are fed from GW discharge. However, other lakes are formed in the sinkholes, therefore are isolated by the glacigene till layer and are fed by atmospheric water (Delina et al. 2012). The karst formation process in the region is still very active, as proved by the high content of sulphates in GW.

As the previously described hydrological types of GW dependent lakes, also closed-basin clear-water lakes, are sensitive to changes in the GW level surrounding them and alterations of GW chemistry. If some anthropogenic activity, e.g. mining or GW extraction causes a GW level decrease in the aquifer that they depend on, the level on such lakes decreases as much as the

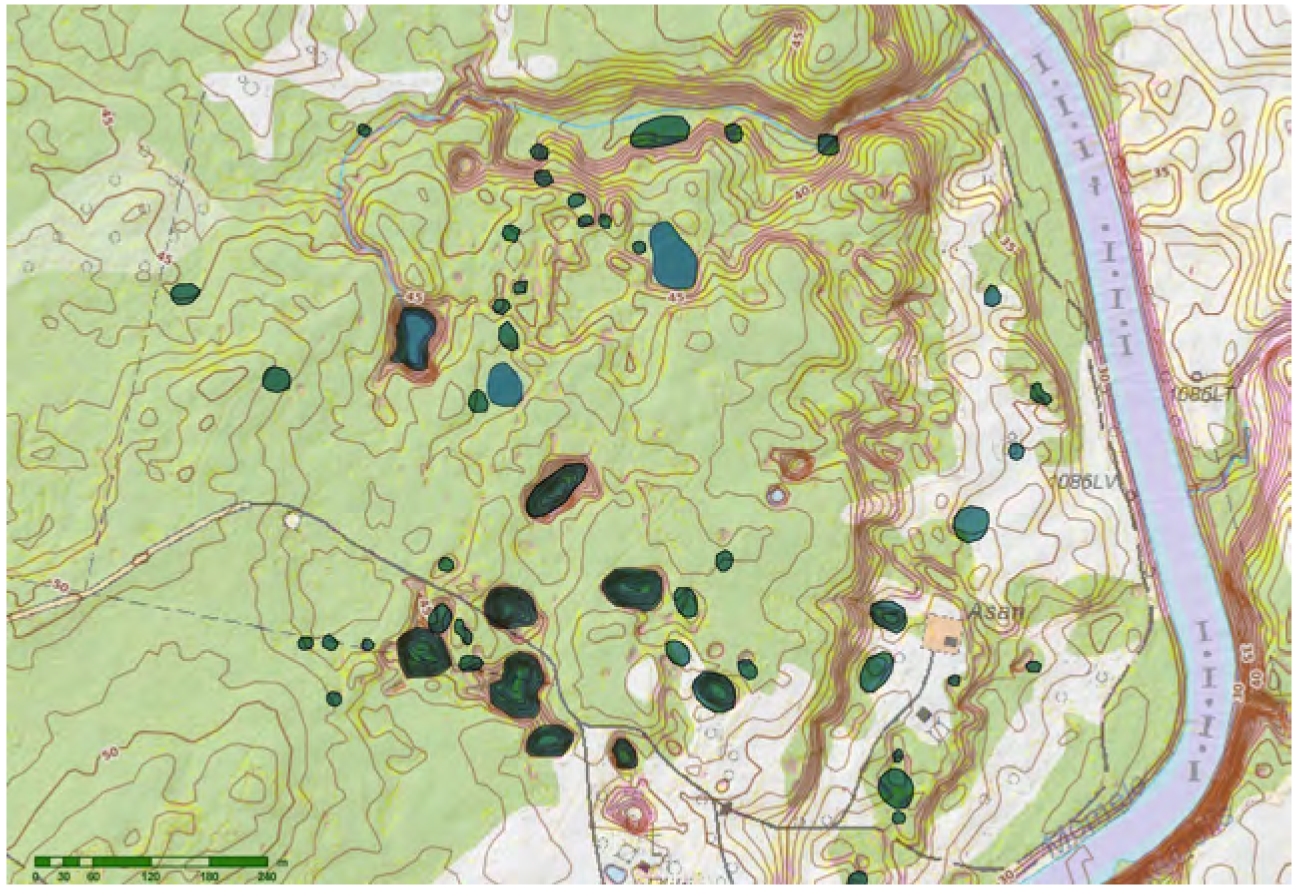


***Figure 3.46**. In total, 72 karst sinkholes (some with permanent water) were found in Skaistkalnes karsta kritenes Nature Monument, a Natura 2000 site, and are classified as the Annex I habitat Lakes of gypsum karst (3190*). LiDAR local relief model © LĢIA. Habitat distribution map © NCA.*

***Figure 3.47.** Gypsum karst lake in Skaistkalne. Photo: J. Matuko.*

GW level. Additionally, oligotrophic and semidystrophic closed-basin lakes are exceptionally vulnerable to excessive nutrient levels in the aquifer feeding them, caused by various anthropogenic practices, as their natural nutrient level is very low.

Groundwater dependent rivers

A river or a reach of a river should be considered critically GW dependent if GW is the only or dominant (>50% of discharge) source of water in it. Therefore, such rivers start from springs, spring lakes or spring fens and may also receive additional diffuse GW inflow in their course. If lakes are either critically GW dependent or not, a river may be critically GW dependent in the headwaters section, but become less dependent downstream, as the contribution of surface water to it increases (Figure 3.48.). It means that it is highly likely that a longer river has to be separated into at least two segments if evaluating the potential effect of GW to it: the section where the average annual GW contribution is more than 50%, and the section where it is lower. In reality, it is impossible to determine such a fixed border, and for practical considerations it is more reasonable to use the same

***Figure 3.48**. River Kunda in Northeastern Estonia starts from springs and is therefore critically dependent on groundwater in its headwater section. The left image is taken 12 km from the river source and the clearness of water is a definite sign of a very high share of groundwater. The right image is taken 31 km from the river source and the water has become significantly darker. It cannot be said, though, barely looking at the color, that the river is not critically dependent on groundwater anymore. It has been determined (though decades ago) that the share of groundwater in the annual discharge of the river has been 54% even another 10 km downstream (Protasjeva & Eipre (eds.) 1972). The color of river water may change already with a relatively small portion of added mire-water. In such situations, the actual share of groundwater should be determined with chemical tracers or baseflow separation methods. Photos: M. Vainu.*

segments of water bodies that are used for reporting the river status under WFD. So, the average share of GW in the river discharge is determined somewhere in the middle part of the water body and based on that the segment of the river is listed as critically GW dependent or not.

The river reaches with high GW contribution are characterized by a stable temperature throughout the year (Figure 3.49.), most importantly for biota, cold or cool water during warm seasons. Though the oxygen content of GW, and therefore also spring water, is low (usually <5 mg/l), water flowing as a thin layer is quickly enriched with oxygen from the air. Thus, the water of such rivers contains a considerable amount of dissolved oxygen (except for the first couple of hundred meters below the source) and because of the low temperature, maintains that oxygen content also in summer

(Järvekülg 2001). Water in GW-fed rivers is low in organic matter and therefore clear regarding both water color and turbidity. It must be noted, though, that all these traits are characteristic to GW dependent rivers close to their source. Further downstream temperature, oxygen content and color may start to resemble surface water-fed rivers, but as long as surface water does not become dominant in the river, it is still highly vulnerable to changes in GW level and chemistry in the aquifer feeding its headwaters.

***Figure 3.49**. Võllinge Stream in winter in Central Estonia. Groundwater-fed rivers generally do not freeze over in winter, at least close to the springs. That is not an absolute rule, though, further downstream where groundwater content may still be over 50%, but the river gradient is low and low air temperature has had time to cool the water down, such a river may freeze like any other. Photo: M. Vainu.*

A feature that does not change during the course of such rivers is lower discharge variability compared to water bodies where precipitation or meltwater make up the largest share of the flow. The lower variability is achieved through low flow periods that are not as extreme as in other rivers, thanks to the relatively constant GW flow. On the other hand, the headwater sections of rivers or streams feeding from Quaternary GW aquifers or bedrock aquifers in topographically higher regions, where GW level fluctuations are significant, may dry during drought periods, as non-GW-dependent rivers or streams (Figure 3.50.).

***Figure 3.50**. Dry river bed at the headwaters of critically groundwater dependent River Loobu in Central Estonia after a dry summer. During periods of high groundwater level, the river starts from the Jõepere spring lake on the northern slope of the Pandivere Upland. After long drought periods, the groundwater level in the limestone aquifer drops so much that outflow from the spring lake ceases (Figure 3.38.) and therefore also the beginning of the river moves ca. 500 m downstream. Photo: M. Vainu.*

Flowing waters that stay cold and therefore oxygen-rich during the summer are suitable habitats for several endangered fish species, e.g. trout, salmon and grayling. Species-richness (e.g. of phytoplankton or macrophytes) in general is relatively low in such rivers, though cold temperatures and usually fast flow do not make them a preferred habitat for typical riverine species.

In Estonia, the highest concentration of critically GW dependent rivers occurs in the region of Pandivere Upland – generally, all rivers starting from the upland are such, at least in their headwater

reaches. The second region of high occurrence of such rivers in Estonia is in the south-eastern part (Protasjeva & Eipre (eds.) 1972).

An exceptional type of critically GW dependent rivers are those that flow through a karst system. Generally, these rivers start from mires and therefore contain no or only a fraction of GW and in that sense should not be considered GW dependent. But as with time they have dissolved for themselves a new path through limestone below ground and have become dependent on changes mostly in the GW level, but also to some extent in GW quality. In the sections where these rivers flow underground, they are in contact with the topmost GW aquifer, and therefore the water composition of these rivers, after they have re-emerged to the surface, differs from the composition before entering the ground (Figure 3.51.). A significant drop in the GW level would likely decrease the amount of water re-emerging to the ground, as more of the river water would enter the GW aquifer, and therefore the river ecosystem downstream of the karst system would suffer. Therefore, in contrary to typical GW dependent rivers, they should not be considered critically GW dependent in their headwater section, but from that point onwards where they flow below the ground.

***Figure 3.51**. River Tuhala in Central Estonia is a typical karst river. It is a dark-watered river that starts from a bog and initially flows above the surface. In the Kata village, it enters a karst system and flows below the ground for ca. 1 km. It re-emerges at the Veetõusme Springs (in the picture) and continues its flow above the ground. Photo: M. Vainu.*

Neither the habitat classification of the Habitats Directive nor the river classification according to the WFD does distinguish between rivers based on the origin of the water. In Estonia, all ecologically valuable river reaches that are in a natural state have been classified as habitat type Water courses of plain to montane levels with the Ranunculion fluitantis and Callitricho-Batrachion vegetation (3260). In Latvia, also the habitat type Rivers with muddy banks with Chenopodion rubri p.p. and Bidention p.p. vegetation (3270) is distinguished, but that does not help to determine the feeding regime of the river. Whereas, the classification of rivers according to the WFD is based on the surface water catchment size of the rivers, both in Estonia and in Latvia. In Estonia, the rivers are further classified based on organic matter content in the water and water color; in Latvia based on river gradient and bottom substrate – mineral vs. organic sediments. Therefore, in Latvia GW dependent rivers may be found under any river type. In Estonia, they are generally under river types with low organic matter content: COD_{Mn} <25 mg O/l (types V1B, V2B and V3B). Exceptions of that rule are dark-watered rivers that pass through a karst system in their course.

4. Major physical, chemical, biotic indicators and criteria in assessing groundwater dependent ecosystems

4.1. Assessing quantitative and qualitative effects on groundwater dependent ecosystems

Agnese Priede (Nature Conservation Agency, Latvia)

Both quantitative and qualitative assessment schemes have been developed for GDTEs during the collaboration between Estonia and Latvia within Interreg Estonia-Latvia project "Joint management of groundwater dependent ecosystems in transboundary Gauja-Koiva river basin" (GroundEco). The schemes were based on the preliminary Estonian assessment scheme (Terasmaa et al. 2015) and the procedure suggested by the European Commission (EC 2009) for the assessment of significant damage to GDTEs caused by quantitative pressures in GWB. It must be noted that the schemes were developed only for terrestrial GDEs, not aquatic ecosystems. However, the same principles may be applied also to GDAEs, as their GW dependency is the same, though different indicators and thresholds may be used.

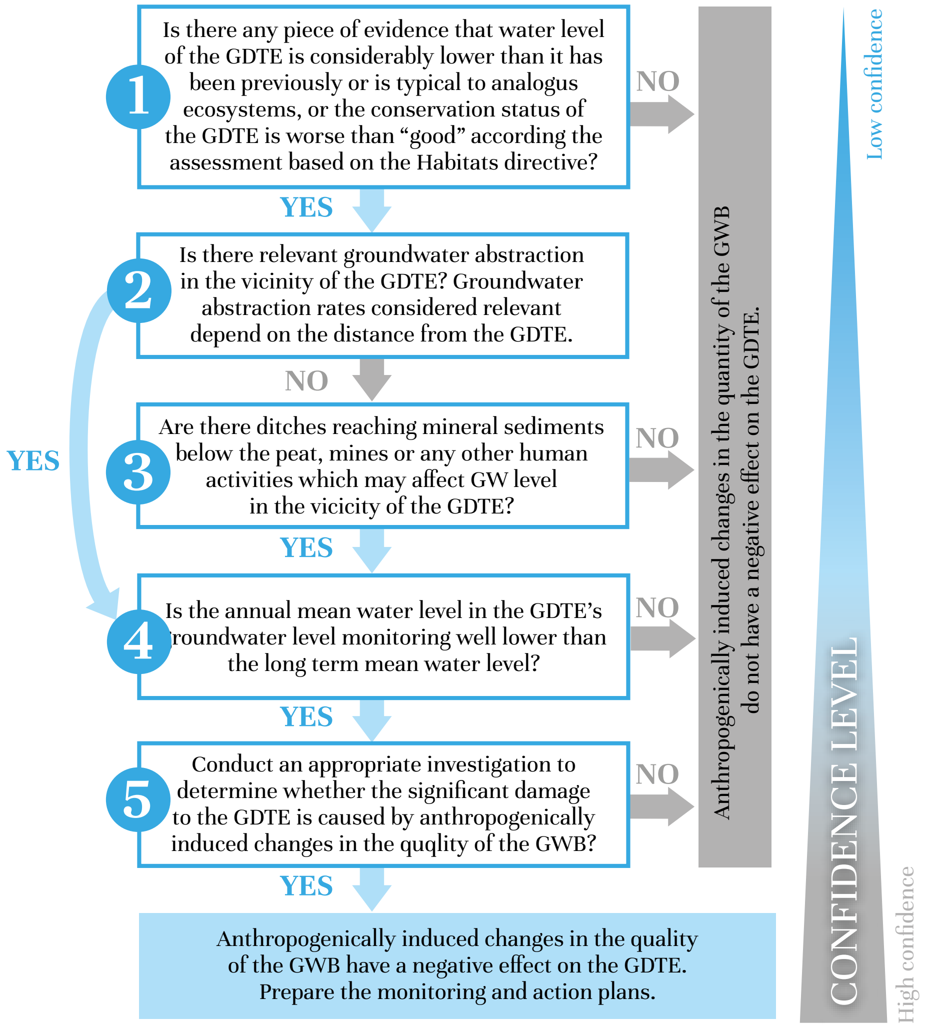


Figure 4.1. *Assessment scheme for the quantitative effect of groundwater on groundwater dependent terrestrial ecosystems (from Retike et al. (2020)).*

Once a GDE is identified using the habitat approach (see Chapter 3.1), the GW dependence should be preferably verified using GW level and water chemistry data (Retike et al. 2020). Then, the potential quantitative and qualitative effects caused by GWB should be assessed (for the purposes of WFD implementation). However, the schemes may be applied for GDE assessment not only for assessment of significant damage to GDTE caused by quantitative and qualitative pressures in GWB but also for the planning of ecological restoration and biodiversity management or assessing environmental impacts in terms of habitat quality outside the context of WFD. The schemes do not include impacts that do not depend on GW quantity and quality, such as trampling and invasive species, as they are not GWB related. Such impacts not related to GWB may significantly alter the GDEs as habitats (in narrower sense), however, the impacts have no relation to GW and do not have a "fundamental role" in the functionality of these ecosystems.

The quantitative assessment scheme consists of five steps (Figure 4.1., from Retike et al. 2020). Each step requires a clear answer (yes or no), supported by evidence that is based on data.

The qualitative assessment scheme for detecting significant damage to GDTE caused by qualitative (chemical) pressures on GWB includes five steps and is given in Figure 4.2.

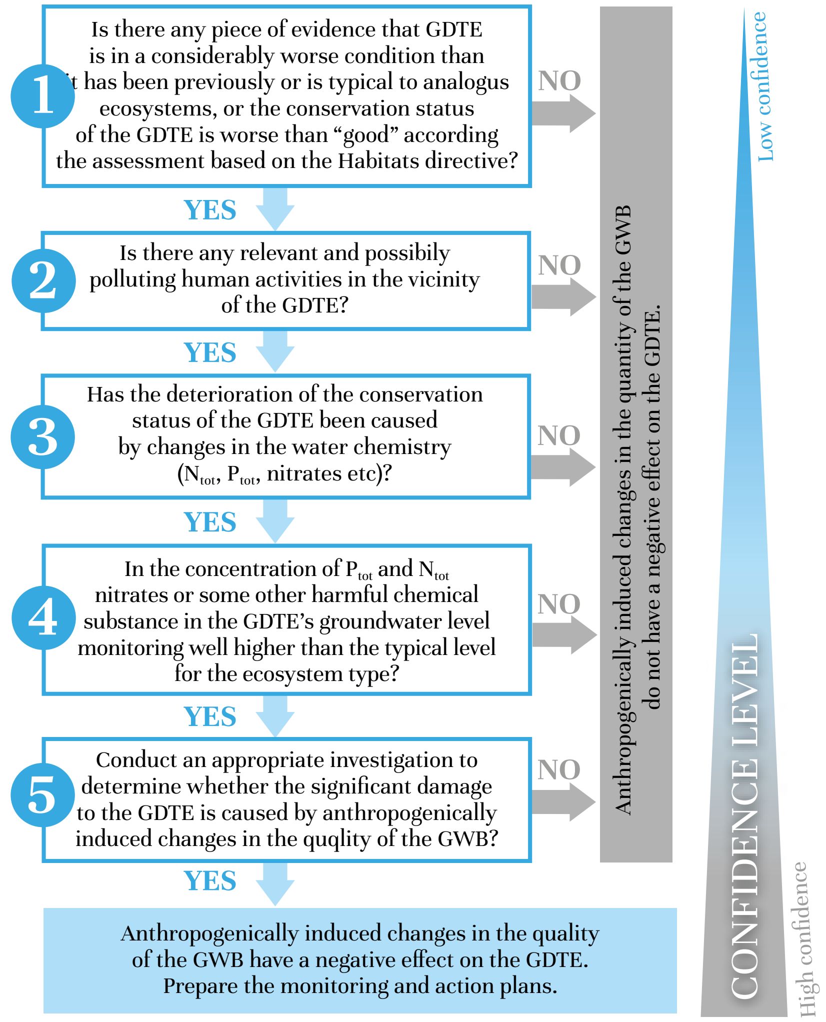


Figure 4.2. *Assessment scheme for the qualitative effect of groundwater on groundwater dependent terrestrial ecosystems to detect significant chemical pressures (from Retike et al. (2020)).*

Each answer leads to the next step or judgement that derives from the answer. Full application of the scheme requires a comprehensive data set, both habitat data and data on GW quantity and quality based on monitoring. Moreover, additional knowledge on the water abstraction amounts in the catchment is needed. Therefore, practical application of the schemes at the site level may be impossible due to a lack of data.

A detailed description of the application of both schemes is provided by Retike et al. (2020) in the final report of GroundEco project (pp. 51–57), thus not repeated here.

4.2. Major physical and chemical indicators

Jānis Bikše (University of Latvia), Jaanus Terasmaa, Oliver Koit (Tallinn University)

There are a large number of GW parameters or indicators that can give detailed information about GW chemistry and physical properties, while only a few important parameters can be enough for a general understanding of GW origin, evolution and its possible impact on ecosystems. This chapter lists the most important indicators of GW that can be relatively easily measured and can provide useful data for the assessment of GDEs. More elaborated text on full-scale assessment of GW and GDEs is given in Chapter 8.

Most indicators listed in this chapter can be measured by a vast variety of equipment and procedures. It is not always necessary to use the most advanced equipment if general knowledge must be acquired. High-level equipment is expensive, while the high accuracy it provides is often not needed to find a clue. Even simple and cost-effective equipment is better than no equipment, therefore it is advisable to keep a few measurement tools at your disposal to rapidly check the GW status in a general way if needed.

4.2.1. Water temperature

Water temperature is an important characteristic of GW. GW usually features a year-round water temperature (6–9 °C; median 8.8 °C) similar to the annual air temperature. In winter, the GW temperature is usually higher compared to the air temperature, and in summer, when the surface water temperature follows the daily average air temperature, the GW temperature is noticeably lower. This makes water temperature a versatile indicator for detecting GW discharge. In spring and autumn, there is a transitional period when the surface water and GW temperatures are similar to the mean daily air temperature.

Water temperature affects:

- the solubility of oxygen and other essential dissolved gases crucial for life in water: the solubility of dissolved oxygen decreases with increasing water temperature;
- chemical reactions: the rate of chemical reactions increases with increasing water temperature. A relevant example of this is the temperature dependence of the electrical conductivity of water;
- biological processes, such as metabolism, growth, and reproduction: aquatic species usually have a limited habitable temperature interval.

Water temperature reading can be obtained by using an alcohol-filled thermometer or an electronic temperature meter. The measurement should be carried out in flowing water – in the case of a well or piezometer a pump and a flow cell should be used to ensure freshwater to the thermometer, while in the case of springs a measurement must be made as close to the spring resurgence as possible. A string tied to the thermometer can be used if the water is out of reach. The reading of the thermometer must be taken after the reading stabilizes. It may take several minutes, depending on the type of measuring device. At first, it should be at least three minutes. The preferred practice is to obtain the temperature reading while the thermometer or probe is still in the water to limit errors due to the impact of air temperature. If this is not possible, use a beaker or a bucket filled with the sample water to obtain a reading away from the sampling point (as quickly as possible). If the measuring device is not of high precision or there are difficulties to ensure proper measurement procedure, take approximately three measurements with one minute intervals apart and write down the average value.

For GW screening needs, thermal imaging by an infrared camera might be used to find a GW discharge in places where the detection of GW seepage is limited. Mapping a thermal signal within a GDTE may result in a better understanding of GW role and connection to the GDTE. Thermal imaging yields the best results if there is a large difference between GW temperature and the surroundings – typically in winter or summer. However, vegetation might obscure the ground where a seepage might occur, therefore best results typically are achieved in winter at lower air temperatures, when GW stands out in the thermal images as spots or areas of elevated temperature anomalies.

4.2.2. pH

pH is a measure of the logarithmic concentration of hydrogen (H+) and hydroxide (OH-) ions in a water environment. pH indicates a relative amount of free hydrogen and hydroxyl ions in the water – water that has more free hydrogen ions is acidic, whereas water that has more free hydroxyl ions is basic or alkaline. pH values of less than 7 indicate an acidic environment, whereas a pH value greater than 7 indicates alkalinity. Because the pH scale is logarithmic, a change in one pH unit indicates the actual acidity/basicness change by 10-folds.

The acid and base balance of water will influence the speciation and mobility of minerals/ions, thus affecting most geochemical and biochemical processes that take place in water. Aquatic species commonly have a narrow tolerance range in pH. Low (<6.5) and high (>8.0) pH environments might be stressful to the majority of aquatic life. However, some ecosystems have adapted to extreme conditions, for example, the ones that inhabit acidic bogs and alkaline fens. The pH in GW may range from pH 6 to 9, while it commonly averages at pH of 7.7.

pH is commonly measured using a hand-held pH-meter in logarithmic pH units (scale pH 0–14). The measurement should be carried out in flowing water – in a flow cell during well/piezometer pumping or as close to the spring resurgence as possible. Like in the case of measuring water temperature and dissolved oxygen, the pH sensor needs to be immersed in flowing water long enough to ensure the stabilization of the reading. Obtain a reading while the pH probe is still immersed in the water. If this is not possible, use a beaker or a bucket filled with the sample water to obtain a reading away from the spring (as quickly as possible).

pH sensors generally have a limited lifetime and are susceptible to calibration drift, thus they must be calibrated on a regular basis. If a pH-meter is not available, a colorimetric test or a strip test kit would be a viable alternative.

4.2.3. Electrical conductivity and specific conductance

Electrical conductivity (EC) is a measurement of the ability of water to carry an electrical current, which is primarily dependent on the amount of dissolved ions in the water. EC is highly dependent on the temperature of the water, thus a temperature compensated (default is 25°C) parameter, also known as **specific conductance** (SEC), is typically used for the comparability of the measurements. SEC may provide insight into the origin of the water. Rainwater, the water in bogs or young GW (freshly infiltrated rainwater) usually feature a low SEC (10–250 µS/cm) while the SEC of GW in shallow aquifers usually falls between 400 to 800 µS/cm, with the median and average of 550 and 650 µS/cm, respectively. The SEC of GW in the deeper aquifers and seawater may range from 1000 to more than 10000 µS/cm. Unexpectedly high SEC reading in surface water or GW may be a sign of pollution.

EC and SEC are measured using a hand-held conductivity meter and the results are typically expressed in µS/cm (1000 µS/cm = 1 mS/cm). Most modern conductivity meters automatically provide a SEC value based on the EC and temperature measurement. If your device does not automatically provide the SEC value, it can be calculated manually by determining the EC and temperature of water and using a default temperature coefficient of 1.91%. SEC can be calculated as follows:

$$SEC\ (25°C) = EC/(1 + 0.0191 \times (T - 25)),$$

where 1 + 0.0191 is the temperature coefficient and T is the measured water temperature.

The measurement should be carried out in flowing water – in a flow cell (GW pumping) or as close to the spring resurgence as possible. Obtain a reading while the probe is still immersed in the water. If this is not possible, use a beaker or a bucket filled with the sample water to obtain a reading away from the sampling point (as quickly as possible).

The meter should be kept clean, have fresh batteries, and calibrated on a regular basis following the manufacturer's recommendation. Although EC and SEC are generally more robust parameters than pH, for example, the same routine should be followed when measuring EC and SEC.

4.2.4. Total dissolved solids

The ability of water to carry an electrical current is primarily dependent on the amount of dissolved ions, therefore SEC can be used to estimate the **total dissolved solids** (TDS) content in the water (expressed in mg/l). TDS is mainly made up of inorganic salts formed from the major cations

found in water (Ca^{2+}, Mg^{2+}, Na^{+}, and K^{+}), the complementary anions (HCO_3^{-}, SO_4^{2-}, and Cl^{-}), and any neutral or uncharged compounds like dissolved silica (H4SiO4), including colloids and even organic matter. Shallow GW usually features a TDS in the range of 300 to 600 mg/l, while mineral water featuring TDS above 2000 mg/l, can be found in deeper aquifers. Elevated TDS in shallow GW or spring water might indicate pollution, while, for example, hydrogen sulfide-rich springs also typically contain higher levels of TDS.

TDS is measured by using a hand-held conductivity meter in mg/l. Most modern conductivity meters automatically provide a TDS reading based on the EC and temperature measurement. Although there is generally a strong linear relationship between EC and TDS, solutions with the same EC reading but with different prevailing dissolved ionic species will feature a slightly different TDS concentration due to the difference in molecular weights. Thus, a conversion factor in the range from 0.55 to 0.8 is used depending on the composition. However, we suggest using a conversion factor of **0.65** as it is the most commonly used factor for freshwater. TDS is calculated from the SEC as follows:

$$TDS = SEC \times 0.65$$

The measurement should be carried out in flowing water – in a flow cell during well/piezometer pumping or as close to the spring resurgence as possible. Obtain a reading while the probe is still immersed in the water. If this is not possible, use a beaker or a bucket filled with the sample water to obtain a reading away from the sampling point (as quickly as possible).

The meter should be kept clean, have fresh batteries, and calibrated on a regular basis following the manufacturer's recommendation.

4.2.5. Dissolved oxygen

Dissolved oxygen (DO) is a measure of how much free oxygen is dissolved in the water. DO in water primarily originates from the atmosphere and photosynthesis. DO is vital for living aquatic organisms. DO concentration in water regulates the valence state and mobility of trace metals, as well as constrains the bacterial metabolism of dissolved organic species. DO is often considered to be absent in the saturated zone of the aquifer due to bacterial reduction, giving way to the reduction of nitrate, mobilization of manganese and iron (e.g. redox ladder). Thus, DO can be an indicator for identifying GW originating from the saturated zone. However, some DO is not rare in shallow GW, especially in shallow karst aquifers, where the flow is rapid and turbulent. Commonly the DO concentration and the percent of air saturation in shallow GW ranges from 0.3 to 3.5 mg/l and 3 to 17 %, respectively.

DO is commonly measured by using an electronic oximeter with either an electrochemical or optical sensor. Most modern oximeters report both the DO concentration in mg/l or ppm and the percent of air saturation. The measurement should be carried out in flowing water – in a flow cell during pumping or as close to the spring resurgence as possible. Obtain a reading while the probe is still immersed in the water. If this is not possible, use a beaker or a bucket filled with the sample water to obtain a reading away from the sampling point (as quickly as possible). Like in the case of measuring pH, the DO sensor needs to be submerged in flowing water long enough to ensure the stabilization of the reading. DO sensors generally have a limited lifetime and are susceptible to calibration drift, thus they must be calibrated on a regular basis.

If an oximeter is not available, a colorimetric test is a viable alternative. The indigo carmine and the rhodazine D method are the two approved colorimetric methods. Both use colorimetric reagents that react and change color when they react with oxygen in the water. The DO concentration is determined with a spectrophotometer, colorimeter or a simple comparator.

4.2.6. Redox potential

Redox potential (ORP) is a measure of the oxidizing or reducing potential of a water environment. Just like pH describes the hydrogen activity, ORP characterizes the electron activity, which controls many crucial redox processes in water (e.g. nitrification, iron and sulfate reduction etc.). ORP is directly related to the amount of DO in the water, as well as other oxidants that function similarly to oxygen. For example, ORP of less than −100 mV indicates the water environment is anaerobic, while values greater than 100 mV indicate aerobic conditions. As an example, arsenic and manganese are more likely to be mobile and present in toxic concentrations in anoxic GW, while concentrations of uranium, selenium, and nitrate are likely to exceed threshold levels in oxic GW.

ORP is measured by using a multiparameter water quality meter also featuring a pH sensor or a combined pH/ORP sensor (expressed in mV). The measurement should be carried out in flowing water – in a flow cell during pumping or as close to the spring resurgence as possible. A reading must be made while the probe is still immersed in the water. If this is not possible, use a beaker or a bucket filled with the sample water to obtain a reading away from the sampling point (as quickly as possible). Like in the case of measuring pH, the ORP or ph/ORP sensor needs to be submerged in flowing water long enough to ensure the stabilization of the reading. Just like pH sensors, ORP or pH/ORP sensors generally have a limited lifetime and are susceptible to calibration drift, thus they must be calibrated on a regular basis.

4.2.7. Alkalinity

Alkalinity refers to the capacity of water to neutralize acid. In uncontaminated GW, alkalinity is primarily a measure of dissolved HCO_3^- ion (if pH >4.5), and CO_3^{2-} ion (if pH >8.3) concentrations. Alkalinity is often referred to as the buffering capacity of a water body, a measure of the ability to neutralize acids and bases, and thereby maintain a stable pH which is vital for the dependent ecosystem. This is because CO_3^{2-} and HCO_3^- ions can neutralize two and one hydrogen ions present in water.

In GW, the alkalinity comes mostly from the dissolution of carbonate minerals. HCO_3^- concentrations in groundwater commonly range from 150 to 350 mg/l.

Alkalinity is measured by collecting a water sample and measuring the amount of acid needed to bring the sample down to a pH of 4.2 (pH indicator methyl orange endpoint). At this pH, all the alkaline compounds in the sample are "used up". The result is reported as mg/l of $CaCO_3$. By multiplying the result by 1.22, you get the HCO_3^- concentration of the water. There are many many easy-to-handle commercial alkalinity titration kits available, for example, MACHEREY-NAGEL VISOCOLOR HE Alkalinity AL 7, Hach AL-AP, Hach AL-DT, Hanna HI-775 or similar. The sample should be obtained from flowing water – from a pumping well tube or as close to the spring resurgence as possible. Use a pre-washed/rinsed (using the sample water) bottle, syringe, beaker or a bucket to collect the sample water to carry out the titration. The titration procedure should be performed as soon as possible.

4.2.8. Nitrates

Nitrogen is an important nutrient to all living things. Although naturally occurring in GW in low concentrations, intensifying human activities, in particular the excessive fertilizer use in agriculture, has led to the widespread pollution of GW with nitrogen compounds. Nitrate is the most commonly occurring form of nitrogen, mobile in aerobic GW. In anoxic conditions, nitrate is reduced to nitrogen gas. Excess nitrate concentrations in GW cause eutrophication in the dependent ecosystems, which in turn may lead to oxygen depletion and the consequent death of fish and small aquatic animals. Long-term consumption of drinking water with high nitrate concentrations may also be harmful to humans, especially for infants. Nitrate concentrations in GW normally range up to about 10 mg/l, while in agricultural areas it may well exceed 50 mg/l.

In the field conditions, nitrate concentration is commonly measured using a colorimetric test kit implementing the cadmium reduction method. There are many easy-to-handle commercial test kits available, for example, Hach NI-11, NI-12 or similar. The sample should be obtained from flowing water – from a pumped well/piezometer tube or as close to the spring resurgence as possible. Use a pre-washed/rinsed (using the sample water) bottle, syringe, beaker or bucket to collect the sample water to carry out the analysis. In case you want to analyze the sample later in the laboratory, collect the sample in a bottle. Until the analysis, keep the bottle in a cold box in the field or in a refrigerator indoors at 4°C. Conduct the analysis within 48 hours from sampling.

4.3. Major biotic indicators

Loreta Urtāne (society „WaterScape")

A GDE assessment scheme has been developed for the purpose of this document and has not yet been tested in practice. Three categories of indicators have been identified to assess the impact on GDE:

A: **Positive Indicator Species** (PIS) – typical habitat species that indicate good habitat status (umbrella species within the meaning of the Habitats Directive);

B: **High Quality Indicator Species** (HIS) – rare species and species for which GDE is only (or almost the only) suitable habitat;

C: **Negative Indicator Species** (NIS) – species that indicate a certain impact (changes in groundwater levels, nutrient enrichment, etc.).

The selection of species included in the specific indicator category is based on a literature review. This should be clarified on the basis of field research and supplemented by quantitative characteristics.

4.3.1. Spring quality assessment

Fennoscandian mineral-rich springs (7160) and petrifying springs with tufa formation (7220)

Mineral-rich springs are characterised by cold and oxygen-rich water, which is also rich in minerals: most often iron, less frequently sulphurous compounds or other mineral substances. Spring-specific vegetation may develop at the spring discharge sites (Auniņš (ed.) 2013; Priede (ed.) 2017). The invertebrate fauna is specific for springs, and flora is rich in northern species (EC 2013; Ikauniece 2013a).

Due to natural succession processes and the site-specific chemical composition of the water, the structure and species composition of the habitat, as they are highly variable, cannot be sufficiently determined as an indicator of the good conservation status of the habitat (Priede (ed.) 2017). Nor can biotic indicators be sufficiently used to assess GDE status. However, **Positive Indicator Species** representing habitat typical species, **High Quality Indicator Species** representing rare species of spring fauna and flora may be used in combination with more precise physical and chemical indicators.

Table 4.1. *GDE impact assessment scheme for Fennoscandian mineral-rich springs (7160)*

Indicator species	Assessment
A: Presence of Positive Indicator Species (PIS):	
Bryophytes: *Trichocolea tomentella, Marchantia polymorpha, Philonotis* spp.*, Cratoneuron filicinum, Scorpidium cossonii, S. revolvens, Calliergonella cuspidata, Calliergon giganteum, Campylium stellatum, Paludella squarrosa, Sphagnum teres, S. warnstorfii, Hamatocaulis vernicosus, Plagiomnium undulatum, P. elatum, P. ellipticum, Brachythecium rivulare, Hylocomiastrum umbratum.* Vascular plants: *Carex remota, C. appropinquata, Sagina nodosa, Cardamine amara, Chrysosplenium alternifolium, Cirsium oleraceum, Crepis paludosa, Myosotis palustris, Veronica beccabunga.*	The minimum requirements for habitat identification are (1) the presence of at least one typical vascular plant species, (2) the presence of at least two bryophyte species, and (3) a certain absence of lime sedimentation processes in spring (Ikauniece & Auniņa 2016). Due to the uncertainty of the habitat structure and species composition parameters, the presence of PIS should be used in combination with a more accurate physical and/or chemical indicator to assess the status of the GDE.
B: Presence of High Quality Indicator Species (HIS):	
Vascular plants: *Montia fontana, Equisetum telmateia, Stellaria crassifolia, Saxifraga hirculus, Ligularia sibirica, Dactylorhiza fuchsii, D. maculata* (Ikauniece 2013a); Invertebrate fauna: *Carychium minimum, C. tridentatum, Cochlicopa lubrica, Vallonia costata, V. pulchella, Euconulus fulvus, E. alderi, Punctum pygmaeum, Vertigo substriata, V. angustior, V. antivertigo, V. pusilla, V. pygmaea, Nesovitrea hammonis, Pupilla muscorum* (Ikauniece & Auniņa 2016).	The absence of an HIS is only an additional indicator and should be used only in combination with the PIS and a more accurate physical and/or chemical indicator. In the presence of PIS, this does not indicate a loss of connectivity to groundwater or other impact.
C: Presence of Negative Indicator Species (NIS):	
Vascular plants: *Carex acuta, C. paniculata*	The assessment of the GDE status should be based on comparative data describing the dynamics of the habitat structure until numerical values are developed on the bases of monitoring data

Some habitat-specific species of the genus *Carex*, although calciphilous, are found in both mineral-rich springs and hard water springs with active formation of tufa. The predominance of such species in mineral-rich springs indicates a change in the GDE condition. Therefore, these species can be classified as **Negative Indicator Species** (Table 4.1.).

The groundwater connected petrifying springs with tufa formation (7220*) are springs with carbonate-rich waters flowing through the underground layers consisting of carbonate sediments (limestone, dolomite). The most important precondition for the existence of this habitat type is hard water spring activity and the formation of tufa (EC 2013; Grootjans et al 2021). Habitat structure and species composition of these GDE vary depending on the nature of the site (water mineral content, sediment type, surrounding habitats, topography, etc.), natural processes and the suitability of thesite conditions to the species (Onete 2014; Priede (ed.) 2017). However, a number of species can be identified for which petrifying springs with tufa formation are the only or almost the only suitable habitat. For the purpose of the GDE impact assessment, these species have been classified as High Quality Indicator Species.

Flora species that form a spring habitat cannot exist without an active flow of spring water (Priede (ed.) 2017). This allows the classification of typical tufa formation-specific vascular plant species that indicate good habitat status (including good connectivity to groundwater) as Positive Indicator Species.

If the connectivity to the groundwater is affected or lost, tufa formation remains as a dry calcareous substrate. As a result, species characteristic of both dry and wet calcareous habitats become dominant (Table 4.2.). One such species is *Carex ornithopoda*, which is defined as an umbrella species in both petrifying springs and dry grasslands on calcareous substrates (6210). Other species of dry calcareous substrate that can be classified as Negative Indicator Species must be identified based on site visits.

Table 4.2. *GDE impact assessment scheme for Petrifying springs with tufa formation (Cratoneurion) (7220*).*

Indicator species	Assessment
A: Presence of Positive Indicator Species (PIS):	
Bryophytes: *Palustriella commutata, P.decipiens, Philonotis calcarea, Seligeria pusilla* (EC 2013; Priede (ed.) 2017). Vascular plants: *Pinguicula vulgaris, Primula farinosa, Carex ornithopoda, Dactylorhiza* spp. (EC 2013; Priede (ed.) 2017).	The absence of PIS is not always due to lost connectivity to groundwater and may be due to other factors. The presence of invasive species must also be assessed in order to evaluate the loss of connectivity to groundwater. In the presence of invasive species, the selected more precise physical and/or chemical indicator should be used to indicate lost connectivity to groundwater or other impact. Invasive species: *Heracleum sosnowskyi, Impatiens glandulifera, I.parviflora, Solidago canadensis, Petasites hybridus.*

B: Presence of High Quality Indicator Species (HIS):	
Bryophytes: *Eucladium verticillatum, Gymnostomum aeruginosum, Seligeria pusilla.* Invertebrate fauna: *Vertigo genesii, V. geyeri, V.angustior* (Rēriha 2013; Rēriha & Auniņa 2016).	The absence of an HIS is only an additional indicator and can only be used in combination with the PIS. In the presence of PIS, this does not indicate lost connectivity to groundwater.
C: Presence of Negative Indicator Species (NIS):	
Vascular plants: *Phragmites australis* and species of dry calcareous substrate (Lyons & Kelly 2016).	*Phragmites australis* can be used as NIS if it is abundant or dominant in tufa-forming areas. The predominance of *Carex flacca, C. ornithopoda* may indicate a lost connectivity to groundwater. In the case of the predominance of these species, the selected more precise physical and/or chemical indicator must be used to indicate lost connectivity to groundwater.

4.3.2. Fen quality assessment

Groundwater connected fens are represented by Calcareous fens with *Cladium mariscus* (7210*) in Estonia and Alkaline fens (7230), which include both rich (calcareous) and poor (acidic, oligotrophic) fens in Estonia and only base-rich (calcareous) fens in Latvia.

Alkaline fens (7230)

Both groundwater regime and water quality are crucial in determining whether these fens have formed (Damman 1995; Mitsch & Gosselink 2000). The alkaline fen is maintained by a constant high water table, perennial irrigation with groundwater and appropriate water chemistry (nutrient poor, but base–rich or calcareous). The alkaline fen peat is rich in calcium, alkaline (pH >6), and poor in nutrients (Wolejko et al 2005; CCW 2008; McBride et al. (eds.) 2011).

Vegetation is rich in species, dominated by small-sedges and brown mosses, often with a large proportion of rare species (Auniņa 2016). In this document, habitat specific rare species are classified as **High Quality Indicators**. For example, *Liparis loeselii* as a small-sized orchid requires (1) low-growing vegetation, (2) high lime content in the soil (pH >6), and (3) easy access to mobile groundwater (Cederberg & Löfroth (eds.) 2000; Roze 2015). Therefore, this species depends on a constant hydrological regime. In case of temporary changes in the water regime, plants do not bloom and survive only in the vegetative stage (Šefferová et al 2008).

Bryophytes are ecologically important and predominate in alkaline fens, and they respond faster to water level lowering than vascular plants (Mälson & Rydin 2007; Šefferová et al 2008). Therefore, habitat-specific bryophyte species are classified as **Positive Indicator Species**.

The absence of PIS and the presence of *Phragmites australis* (according to NIS values) in the alkaline fen could indicate that the alkaline fen is enriched with nutrients and that conditions change, making edaphic and hydrological conditions less favourable for species indicating an alkaline fen.

Table 4.3. *GDE impact assessment scheme for Alkaline fens (7230).*

Indicator species	Assessment
A: Presence of Positive Indicator Species (PIS):	
Bryophytes: *Scorpidium revolvens, S. cossonii, Campylium stellatum, Scorpidium scorpioides* (Mälson & Rydin 2007; Priede (ed.) 2017). Vascular plants: *Schoenus ferrugineus, Carex davalliana, C. panicea, C. elata, C. lasiocarpa, C. buxbaumii, Primula farinosa, Pinguicula vulgaris* (Priede (ed.) 2017).	GDE vegetation with an appropriate connection to groundwater and water chemistry (nutrient poor and base–rich) is characterized as follows: • Dominated by small-sedge species – *Schoenus ferrugineus, Carex davalliana, C. panicea*, and/or some tall sedge species – *C. elata, C. lasiocarpa, C. buxbaumii;* • Small, light-demanding plants are found – *Primula farinosa, Pinguicula vulgaris*, and mosses – most often *Scorpidium revolvens, S. cossonii, Campylium stellatum, Scorpidium scorpioides* (Priede (ed.) 2017); • Surface water is visible or expressable on footfall between *Schoenus sp. tussocks* in 30% of the area (CCW 2008).
B: Presence of High Quality Indicator Species (HIS):	
Bryophytes: *Leiocolea rutheana, Preissia quadrata, Moerckia hibernica* (Auniņa 2013; Roze 2015). Vascular plants: *Liparis loeselii, Saussurea alpina* ssp. *esthonica, and several species of Dactylorhiza spp.* (Roze 2015; Auniņa 2013). Snails: *Vertigo genesii, V. geyeri, V. angustior* (Auniņa 2013). Butterflies: *Lycaena dispar, Maculinea teleius* (in Latvia), *Euphydryas aurinia.* Dragonflies: *Leucorrhinia pectoralis, Coenagrion ornatum* (in Latvia). Spiders: *Dolomedes plantarius* (Šefferová et al. 2008).	The absence of an HIS is only an additional indicator and can only be used in combination with the PIS. In the presence of PIS, this does not indicate lost connectivity to groundwater.
C: Presence of Negative Indicator Species (NIS):	
Vascular plants: *Molinia caerulea, Phragmites australis* – if exceed NIS values (CCW 2008).	The vegetation of the impacted GDE is characterized as follows: • The predominant role of *Molinia caerulea* in vegetation indicates an increase in nutrients; • *Phragmites australis* > 10 live plants within a radius of 1 m from each sampling point in combination with the absence of positive indicator species indicate that the alkaline fen is enriched and conditions change, making edaphic and hydrological conditions less favourable for alkaline fen species (CCW 2008).

Due to groundwater level modifications and drainage, the peat decomposition rate of peat increases, causing an increase in the concentration of nutrients (mainly phosphorus). This promotes the nutrient-demanding species, such as *Molinia caerulea*, classified as **Negative Indicator Species** (Table 4.3.).

Calcareous fens with *Cladium mariscus* (7210*)

The *Cladium mariscus* stands are maintained by a relatively high water table and nutrient poor, but base-rich water quality (EC 2013). Perennial groundwater supply is critical. *Cladium* fen at good favourable condition should have standing water in winter and at least 15 cm below the ground in summer (CCW 2008). Such conditions also indicate a good connectivity with groundwater.

The great fen-sedge *Cladium mariscus* is a calciphilous and umbrella species. The habitat-specific species are classified as Positive Indicator Species. In the case of calcareous fens, the charophytes, in particular *Chara asper*, *Chara globularis*, *Chara tomentosa* are classified as High Quality Indicator Species.

Table 4.4. *GDE impact assessment scheme for Calcareous fen with Cladium mariscus (7210*).*

Indicator species	Assessment
A: Presence of Positive Indicator Species (PIS):	
Bryophytes: *Campylium stellatum, Scorpidium cossonii, Scorpidium scorpioides.* Vascular plants: *Cladium mariscus, Carex elata, C.lasiocarpa, Schoenus ferrugineus.*	GDE vegetation with appropriate connection to groundwater and water chemistry (nutrient-poor, calcium-rich) is described as follows: • umbrella species *Cladium mariscus* with >50% coverage (CCW 2008); • calciphilous species – *Campylium stellatum, Scorpidium cossonii, Schoenus ferrugineus* – are present; • another PIS species is present.
B: Presence of High Quality Indicator Species (HIS):	
Charophytes: *Chara asper, Chara globularis, Chara tomentosa.*	The absence of an HIS is only an additional indicator and can only be used in combination with the PIS. In the presence of PIS, this does not indicate a lost connectivity to groundwater.
C: Presence of Negative Indicator Species (NIS):	
Vascular plants: *Phragmites australis* – if exceed NIS values.	The vegetation of the impacted GDE is characterized as follows: • There are > 30 *Phragmites australis* stems in any open fen within a radius of 1m; • In any 1 m radius, the open fen has > 20% coverage of the dead vegetation litter (CCW 2008).

The absence of calciphile PIS, the presence of *Phragmites australis* (according to NIS values) and the increase in dead vegetation litter in the calcareous fen could indicate that conditions are changing, making edaphic and hydrological conditions less favourable for species indicative for this type of fen. This allows the *Phragmites australis* to be classified as a Negative Indicator Species (Table 4.4.).

4.3.3. Swamp wood quality assessment

Fennoscandian deciduous swamp woods (9080*)

Typical swamp woods include wet deciduous forests with wet growing conditions, persistently high groundwater table and frequent flooding depending on the season and precipitation. The most typical tree species of swamp woods are *Alnus glutinosa*, in combination with *Betula pubescens*, *Salix* spp., less commonly – *Picea abies* and *Fraxinus excelsior*. Trees are usually biologically old, with low dimensions (both in diameter and height). Most trees grow on hummocks that support vegetation that is different from the rest of the ground-floor (Ikauniece 2013 b).

The natural connection with groundwater leads to the formation of habitat-specific microtopography and structure – a combination of wet depressions and hummocks – characterised by mosaic vegetation structure. There are no dominant species in the herb layer, except in the habitat formation phase (Ikauniece (ed.) 2017; Priedītis 1997).

When a habitat loses its connectivity with groundwater, both the structure of the stand and the composition of species change. Decreased moisture improves soil aeration, increases the rate of

Table 4.5. *GDE impact assessment scheme for Fennoscandian deciduous swamp woods (9080*).*

Indicator species	Assessment
A: Presence of Positive Indicator Species (PIS):	
Trees: *Alnus glutinosa*, in combination with *Betula pubescens, Salix* spp., *Picea abies, Fraxinus excelsior.* **Vegetation of wet depressions –** Bryophytes: *Climacium dendroides, Calliergoniella cuspidata, Plagiomnium elatum, Sphagnum squarrosum.* Vascular plants: *Solanum dulcamara, Lycopus europaeus, Galium palustre, Carex* spp., *Iris pseudacorus.* **Vegetation of hummocks –** Bryophytes: *Rhytidiadelphus triquetrus.* Vascular plants: *Oxalis acetosella, Vaccinium myrtillus, Dryopteris* spp., *Lysimachia vulgaris.* Scrubs and trees: *Frangula alnus, Ribes nigrum, Salix* spp. (Ikauniece (ed.) 2017).	GDE vegetation with and appropriate connection to groundwater and water chemistry (calcium-rich) is characterized as follows: • The nature of ground-floor vegetation is mosaic, represented by the PIS of vascular plants and bryophytes (in some cases, the vegetation of herbs and mosses may be lacking in wet depressions); • There are no dominating species in the herb layer, except in the habitat formation phase; • PIS tree species are present; • Trees of different ages and dead wood accumulate in the habitat; • Biologically old trees are small in diameter.

B: Presence of High Quality Indicator Species (HIS):	
Bryophytes and lichens: *Cetrelia* spp., *Leptogium* spp., *Arthonia spadicea, Jungermannia leiantha, Geocalyx graveolens, Arthonia spadicea, A. vinosa, Menegazzia terebrata, Trichocolea tomentella* (Ikauniece 2013b). Snails: *Vertigo moulinsiana* (Pilate 2009). Birds: *Picoides tridactylus, Dendrocopos leucotos, Glaucidium passerinum* (Petriņš 2014).	The absence of an HIS is only an additional indicator and can only be used in combination with the PIS. In the presence of PIS, this does not indicate a lost connectivity to groundwater.
C: Presence of Negative Indicator Species (NIS):	
Bryophytes: *Pleurozium schreberi, Hylocomium splendens.* Vascular plants: *Phragmites australis, Trientalis europaea, Deschampsia cespitosa, Oxalis acetosella, Urtica dioica* (Prieditis 1997).	The vegetation of the impacted GDE is characterized as follows: • The predominant role of *Phragmites australis* in vegetation, the presence of other NIS and the absence of PIS suggest that conditions are changing, making edaphic and hydrological conditions less favourable for species indicating swamp wood; • The presence of *Urtica dioica* in the vegetation indicates peat mineralisation and increased nutrients.

microbial decomposition and mineralisation, which reduces the thickness of peat layer and increases the availability of nutrients to plants (Indriksons 2007). The expansion of *Phragmites australis* and the presence of generalist species, such as *Trientalis europaea, Deschampsia cespitosa, Pleurozium schreberi, Hylocomium splendens*, and *Oxalis acetosella* indicate changes in the natural hydrological regime (Prieditis 1997). Similarly, the presence of *Urtica dioica* indicates an increase in nutrients (Table 4.5.).

4.3.4. Groundwater dependent aquatic habitats

Karst lakes, spring-fed permanent lakes and other temporary groundwater-fed lakes

To characterized biotic indicators, all karst lakes formed in depressions of karstic origin are combined and include both (1) temporary groundwater-fed lakes with inflow and/or outflow and (2) spring-fed permanent lakes.

Karst lakes that form in depressions of karstic origin vary greatly in shape, size, age of origin and type. Older karst lakes during the succession have have transformed into different types of mires or waterbodies with type-specific vegetation. The vegetation of karst lakes consists of grassland and mire species – mainly plant species that are well adapted to fluctuating water levels. There are also free floating and submerged aquatic plant communities. Terrestrial plant communities are also found in older karst lakes (Eņģele 2013).

In Latvia, no characteristic plant or animal species are found in karst lakes, and no species related to karst processes have been identified (Urtāns (ed.) 2017). All plant species represented in the karst lakes survive the dry phase in deeper and wetter hollows. Also, the composition of fauna species depends more on the type of soluble rocks (types of carbonate rocks – dolomite, limestone, sulphate – gypsum) than other abiotic factors. Due to the temporary nature of karst lakes, hydrobiological studies of aquatic fauna (fish, phytoplankton and zooplankton) of different trophic groups have been rarely conducted. Rare species have been reported in some karst lakes in Estonia in the 1960s and 1970s (Mäemets 1974). Due to limited studies, the existence of such species is not subsequently confirmed (Ott 2010). Therefore, biotic indicators cannot be sufficiently used to assess the status of karst lakes.

Closed-basin clear-water lakes

According to the assessments performed (see Chapter 3.1.2), only clear-water lakes in the group of closed-basin lakes are critically dependent on GW. GW in closed-basin clear-water lakes is concluded to be the only source of water or to contain chemicals that are critical to the ecology of plant and animal species adapted to such specific conditions. However, it should be understood that none of the aquatic habitats can be completely isolated from the surface impacts. Thus, even for a group of closed-basin lakes, GW is not only, but the main source of water, as water is also collected through direct surface runoff. Air mass transfer is also an important factor influencing nutrient enrichment and acidification processes.

Another process that is important in developing a GDE impact assessment scheme is the succession of lakes, which is a natural process driven mainly by the introduction of organic matter (especially nutrients) and sediment into the lake system. In terms of nutrient accumulation, lakes are oligotrophic (young, with low nutrient levels and productivity), mesotrophic (mature, with medium nutrient levels and productivity) or eutrophic (old age, with high nutrient levels and productivity) (Urtāne 2014). Nutrient accumulation processes in GDE are significantly slower than in other lakes. However, the GDE impact assessment scheme cannot be based on the GW connectivity factor alone, as the biological indicators are initially related to the state of the lake (habitat within the meaning of the Habitats Directive), but dependence on GW ensures a high quality of a particular habitat. The closed-basin clear-water lakes belong to different lake types and different habitats, respectively. The connection with the GW is more important in the early stages of the lake's development, while the processes related to nutrient accumulation are becoming more and more important as the lake reaches a eutrophic state.

Closed-basin clear-water lakes in Latvia belong to 3 different habitats:

- Oligotrophic to mesotrophic standing waters with vegetation of *Littorelletea uniflorae* and/or *Isoeto-Nanojuncetea* (3130);
- Hard oligo-mesotrophic waters with benthic vegetation of *Chara* spp. (3140);
- Natural eutrophic lakes with magnopotamion or Hydrocharition – type vegetation (3150).

It must be noted, that the interpretation of habitat types differs among countries. In Estonia, all the soft water oligotrophic, formerly oligotrophic or semidystrophic lakes with *Lobelia-Isoëtes* complex and *Sparganium angustifolium* communities are classified as habitat Oligotrophic waters containing very few minerals of sandy plains (*Littorelletalia uniflorae*) (3110). This type of habitat is not recognized in Latvia at all. Still, the given GDE impact assessment of clear-water lakes of habitat type 3130 in Latvia can be apllied to habitat type 3110 in Estonia. However, the interpretation of habitat 3130 in Estonia (used term is Oligo- and mesotrophic lakes with medium hardness), which are lakes with medium hardness and typical plant species like *Potamogeton filiformis*, *Eleocharis acicularis*, *Ranunculus reptans, Elatine hydropiper, Chara aspera, C. hispida, C. tomentosa, C. contraria* are more similar to habitat 3140 in Latvia.

Given the state of development of the lakes and the specificities of the particular lake habitat, the GDE impact assessment scheme has been developed for soft-water and hard-water oligotrophic lakes, while a more general scheme has been developed for natural eutrophic lakes due to the complex interactions of variables.

Brown-water closed-basin lakes

Brown-water closed-basin lakes are interconnected but not critically dependent on GW. These lakes belong to the habitat Natural dystrophic lakes and pools (3160), which by their origin are primary or secondary succession lakes. Primary succession lakes, which are remains of relic lakes and have maintained contact with mineral ground, may have GW inflows. In contrast, secondary succession lakes, which are formed when a peat layer sinks under the impact of gravity, are not connected to GW. In limnological practice, the origin of a dystrophic lake is indicated by the composition of aquatic species (Urtāne & Kļaviņš 1995; Druvietis et al. 1997, 1998). At the current level of research, it is concluded that connectivity to GW is not leading in both groups of dystrophic lakes, therefore all dystrophic lakes are classified as non-critically dependent on GW. However, this assessment may be revised in the future on the basis of detailed natural studies and / or modelling.

Oligotrophic to mesotrophic standing waters with vegetation of *Littorelletea uniflorae* and/or *Isoeto-Nanojuncetea* (3130)

The habitat is represented by both clear-water (oligohumic) and brown-water (polyhumic) lakes, and species of the *Lobelia-Isoëtes* complex are important for this habitat (Urtāns (ed.) 2017). Depending on their origin, soft-water oligotrophic to mesotrophic lakes are poor to moderately rich in nutrients (clear-water lakes) or rich in nutrients (brown-water lakes) consisting of complex, water-insoluble humic substances (Urtāne 2014). Therefore, the habitat is represented by both clear-water and brown-water lakes, while the GDE is limited to clear-water lakes. According to current research, closed-basin brown-water lakes are unlikely to be critically dependent on GW.

The water exchange in this GDE is long-lasting and water level fluctuations are negligible, leading to relatively stable environmental conditions, including reduced nutrient leaching and the development of sparse vegetation or the existence of vegetation-free littoral areas with mineral bottoms (Figure 4.3.).

Increased water level fluctuations are the first signs of a possible loss of connection to the GW, but may also be due to other factors, such as intense beaver activity. Lost connectivity to GW is in most cases expressed as a decrease in groundwater level, while altered connectivity is expressed as an increase in groundwater level. The altered connectivity to GW in the case of Lake Buļļezers was expressed as an increase in water levels, which led to increased nutrient run-off and habitat degradation (Figure 4.4.).

Lake Laukezers is a GDE representing the mesotrophic lake habitat (3130). The connectivity to GW is initially indicated by the presence of other GDEs (mineral-rich sources) in the vicinity of the lake and stable water levels. The water exchange period in the lake is very long - 9% per year, which is 4058 days or about 11 years (Urtāne 2014). Connectivity to GW in combination with lake morphometry (mean depth 6.7 m, max - 19.8 m) ensures high habitat quality, although increasing recreational load is recorded.

Presence of Positive Indicator Species:

- Vegetation of the lake littoral part is presented by 8 PIS;
- The emergent vegetation is sparse and occurs in combination with vegetation-free zones;
- The layer of dead aquatic plants and dead leaves at the bottom is thin and sparse.

Presence of High Quality Indicator Species:

- The vegetation of lake littoral part is presented by 3 HIS.

Figure 4.3. *Application of the GDE impact assessment scheme – a case study of Lake Laukezers in Latvia. Photos: L. Urtāne.*

1969. Source: Šablis 1971.

1988. Photo: U.Suško.

2009. Photo: A.V.Urtans

2009. Photo: A.V.Urtans

Between 1969 and 1988, habitat degradation occurred with increasing recreational pressure. PIS was registered duringthis period, and an indication for eutrophication was also registered in 1988 (see next section). After the *drainage of the* surrounding areas changed the connectivity with GW, the habitat was completely destroyed - the *Lobelia-Isoëtes complex* (PIS) disappeared in the littoral and was replaced by *Phragmites australis, Carex* spp. and *Polygonum amphibium* in the central part of the lake. Massive occurrence of Cyanophyta spp. (algal blooming) was recorded.

Figure 4.4. *Degradation of GDE as a result of multi-factor impact – a case study of Lake Buļļezers in Latvia.*

Species of the *Lobelia-Isoëtes* complex have been classified as **Positive Indicator Species** to define the GDE impact assessment scheme. Given that compatibility with GW ensures the high quality of this habitat (Urtāns (ed.) 2017), rare and relict species such as *Najas flexilis* and *N. tenuissima* are classified as **High Quality Indicator Species** (Table 4.6.).

Hard oligo-mesotrophic waters with benthic vegetation of *Chara* spp. (3140)

The habitat is mostly recorded in water bodies rich in calcium and magnesium compounds (Urtāns (ed.) 2017). The bottoms of these lakes are covered with carpets of charophytes (*Chara* spp. and *Nitella* spp.) (EC 2013). Charophytes provide the lower layer with oxygen, enhancing nitrification and denitrification processes and preventing the return of phosphorus from the lake's sediments to the aquatic environment (Blindow 1991).

The presence of Positive Indicator Species and High Quality Indicator Species would indicate that (a) the groundwater body feeding the lake is not contaminated with excess nutrients and does not have a negative chemical effect on the lake, or (b) groundwater level and therefore the lake level is stable and not affected by anthropogenic interventions. The presence of Negative Indicator Species, on the other hand, indicates that (a) the lake could be receiving excess nutrients from the groundwater body, or (b) the groundwater level is fluctuating or has dropped, but not ceased to exist.

Charophyte lakes are home to a number of rare and protected species. The most common are *Najas marina*, other rare and protected species are vascular plants *Ceratophyllum submersum*, *Potamogeton pusillus*, *P. rutilus*, *Zannichellia palustris*, macroscopic algae *Chara polyacantha*, *C. strigosa* and a bryophyte *Ricciocarpos natans* (Gavrilova et al. 2005; Zviedre & Grīnberga 2012). In the GDE impact assessment scheme, these species are classified as **High Quality Indicator Species**.

These GDEs are characterized by stable water levels and nutrient-poor water inflow. As a result, the lakes remain stable for a long time due to the immobilization of phosphorus, the high oxygen concentration in the bottom layer and the dense cover of charophytes, which limits the presence

Table 4.6. *GDE impact assessment scheme for Oligotrophic to mesotrophic standing waters with vegetation of Littorelletea uniflorae and/or Isoeto-Nanojuncetea (3130).*

Indicator species		Assessment
A: Presence of Positive Indicator Species (PIS):		
Bryophytes: *Fontinalis dalecarlica, Warnstorfia trichophylla, W. exannulata* (Urtāns (ed.) 2017). Vascular plants: *Isoëtes echinospora, I. lacustris, Juncus bulbosus, Littorella uniflora, Lobelia dortmanna, Myriophyllum alterniflorum, Nuphar pumila, Ranunculus reptans, Sparganium angustifolium, S. gramineum, Subularia aquatica.*		GDE vegetation with an appropriate connection to GW and water chemistry (soft-water) is characterized as follows: • The vegetation of littoral part of the lake is represented by at least 2–3 PIS species; • The emergent vegetation is sparse and occurs in combination with vegetation-free zones; • Layer of dead aquatic plant parts and dead leaves on the bottom is thin or this layer is absent.
B: Presence of High Quality Indicator Species (HIS):		
Charophytes: *Chara filiformis, C. strigosa, Nitella flexilis, N. tenuissima* (Suško & Bambe 2002; Suško & Āboliņa 2010). Bryophytes: *Calliergon megalophyllum, Drepanocladus sordidus, Platyhypnidium riparioides.* Vascular plants: *Callitriche hermaphroditica, Hydrilla verticillata, Najas flexilis, N. tenuissima, Potamogeton pusillus, P. rutilus, P. sturrockii, P. filiformis, Scolochloa festucacea.*		The absence of an HIS is only an additional indicator and can only be used in combination with the PIS. In the presence of PIS, this does not indicate deterioration caused by changes in GW quality or level.

C: Presence of Negative Indicator Species (NIS):		
Vascular plants: *Phragmites australis.*		The vegetation of the impacted GDE is characterized as follows: • The predominant role of *Phragmites australis* in the vegetation of littoral and the lack of PIS suggest that conditions are changing, making edaphic and hydrological conditions less favourable for habitat-indicating species; • Dense stands of *Phragmites australis*, a layer of dead aquatic plants are recorded in combination with a retreating shoreline. In the case of these indications, the more precise physical and / or chemical indicator chosen must be used to indicate that the changes in GW quality or level have caused the impact.

of other plant species. The lowering of the water level due to the loss of the connectivity with GW creates better conditions for the development of emergent vegetation, which destroys the vegetation of the charophytes. The multiplier effect of the destroyed charophytes is its replacement with other submerged and floating plant species (Table 4.7.), which causes a decrease in oxygen concentration in the bottom layer and inhibition of phosphorus immobilization processes.

Table 4.7. *GDE impact assessment scheme for Hard oligo-mesotrophic waters with benthic vegetation of Chara* spp. *(3140).*

Indicator species		Assessment
A: Presence of Positive Indicator Species (PIS):		
Charophytes: *Chara aspera, C. contraria, C. globularis, C. intermedia, C. hispida, C. tomentosa, C. virgata, C. rudis, C. vulgaris, Nitellopsis obtusa*		GDE vegetation with an appropriate connectivity to GW and water chemistry (hard-water) is characterized as follows: • There are large continuous charophyte stands throughout the lake; • Only separate clusters of floating-leaved or emergent plants are formed between these stands; • Charophytes have good vitality, their lateral branches overlap and form a dense branch structure; • Charophytes are not covered by dead parts of vascular plants.
B: Presence of High Quality Indicator Species (HIS):		
Charophytes: *Chara polyacantha, C. strigosa.* Bryophytes: *Ricciocarpos natans.* Vascular plants: *Ceratophyllum submersum, Cladium mariscus, Najas marina, Potamogeton pusillus, P. rutilus, Scirpus tabernaemontani, Zannichellia palustris*		The absence of an HIS is only an additional indicator and can only be used in combination with the PIS. In the presence of PIS, this does not indicate a loss of connectivity to groundwater.
C: Presence of Negative Indicator Species (NIS):		
Vascular plants: *Potamogeton pectinatus, Ceratophyllum demersum, Phragmites australis.*		The vegetation of the impacted GDE is characterized as follows: • Charophyte coverage is not dense; • The co-dominant role of *Potamogeton pectinatus, Ceratophyllum demersum* in the lake vegetation and the lack of PIS indicate that conditions are changing, making edaphic and hydrological conditions less favourable for habitat-indicating species; • Dense stands of *Phragmites australis*, a layer of dead aquatic plant debris recorded in combination with a decrease in shoreline length (could be assessed using historical reports and monitoring data). In the case of these indications, the more precise physical and/or chemical indicator chosen must be used to indicate the loss of connectivity to GW.

Natural eutrophic lakes with Magnopotamion or Hydrocharition - type Vegetation (3150)

Depending on their depth, natural eutrophic lakes are usually characterized by a narrower or wider species-rich belt of submerged plants, a sparse or continuous zone of floating-leaved plants, and also a fragmented sparse or continuous zone of emergent plants. This creates a high diversity of plant species, which is characteristic of natural eutrophic lakes in the early stages of eutrophication. The accumulation of nutrients leads to a simplification of vegetation, in particular a decrease in the total number of aquatic plant species, a predominance of species indicating a nutrient-rich state, and so on. See the box "Plants as environmental indicators in aquatic ecosystems" for a more detailed description.

As mentioned above, the link with the GW lakes is more important in the early stages of the lake's development, while nutrient accumulation processes are becoming increasingly important as the lake reaches a eutrophic state. The reaction of aquatic organisms even in the closed-basin natural eutrophic lakes refers to the trophic state of the lake (nutrient accumulation level), while the dependence of GW is the result of the quality of the aquatic environment and the formation of relatively stable environmental conditions. Therefore, biotic indicators cannot be sufficiently used to assess the status of GDE. However, signs of deterioration and eutrophication in the aquatic environment (can be assessed using historical studies and / or monitoring data) are the first indications of possible loss of connectivity to GW, but may be due to other factors such as intensive beaver or human pollution activities.

Plants as environmental indicators in aquatic ecosystems

Aquatic plants have a number of basic characteristics that make them valuable indicators for assessing the state of the environment. Several methods for determining the quality of the aquatic environment using water plants as biotic indicators have been developed and are used in practice. For example, the saprobic index developed to assess the level of organic pollution in rivers using water organisms as indicators (Cimdins et al. 1995), including water plants (Urtans 1995) or the “Methodology for ecological quality assessment of river and lake waterbodies”, where a complex of indicator species or plant community structure is used to determine the status and level of pollution and which is used to prepare River Basin District Management Plans. Experts also use the presence of specific species to assess the state of the aquatic environment (Grīnberga 2011; Urtāne 2014; Vizule-Kahovska & Uzule 2016; Urtāns (ed.) 2017). The use of macrophytes to assess the quality of the aquatic environment is summarised in Figures 4.5. and 4.6.

Figure 4.5. *“Healthy”, naturally developed lakes are characterized by all zones of aquatic plants, diverse species composition, and developed zone of submerged aquatic plants. The massive growth of aquatic plants and algae indicates deterioration of the lake and accelerated anthropogenic eutrophication.*
Aquatic plants in lakes: emergent and floating-leaved aquatic plants on the left (photo: L. Vizule-Kahovska), submerged aquatic plants in the middle (photo: V. Līcīte), floating-leaved aquatic plants on the right (photo: L. Vizule-Kahovska).

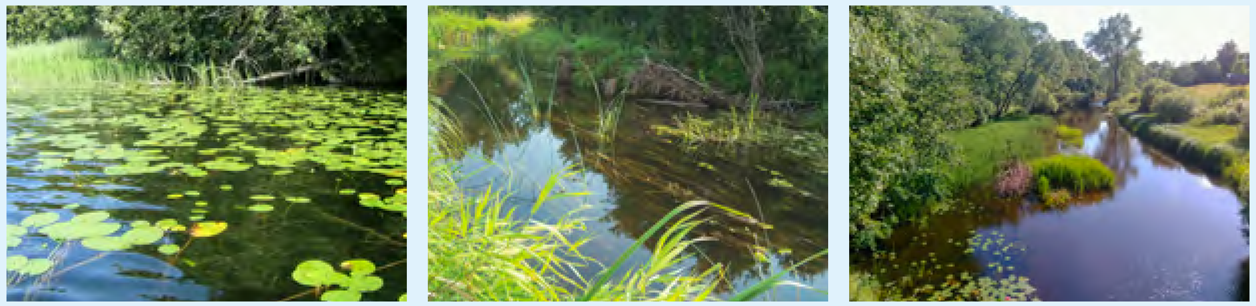


Figure 4.6. *Aquatic plant zones are also found in rivers, but they are usually narrower than in lakes, depending on the width and depth of the river. The number of aquatic plant species in rivers is relatively smaller than in lakes – up to 10 species in rivers on average, but up to 20 in lakes. Reeds, water-lilies and various species of pondweeds are typical and widespread species of aquatic plants in both lakes and rivers.*
Aquatic plants in rivers: emergent and floating-leaved aquatic plants on the left (photo: I. Upena-Rasuma), submerged aquatic plants in the middle (photo: L. Vizule-Kahovska), emergent and floating-leaved aquatic plants on the right (photo: L. Vizule-Kahovska).

4.3.5. General scheme for lake and river quality assessment

Loreta Urtāne (society „WaterScape”), Lauma Vizule-Kahovska (Nature Conservation Agency, Latvia)

Cosed-basin natural eutrophic lake

Although each lake must be assessed individually, taking into account both its hydromorphological and physico-chemical parameters and the impacts, knowing the main interconnections, it is possible to assess the processes taking place in the lake, as well as its quality (Figure 4.7.).

– Negative changes in the lakes are indicated by	+ Good conditions in the lakes are indicated by
Low species diversity, dominance of one or few species, including nutrient tolerant species	High diversity of species, especially in the submerged plant zone (many species, but a small number of individuals of each species)
Homogeneous vegetation. Dense and lush bands of all plants submerged, floating and emergent vegetation	Mosaic of plant communities
Dense free-floating plant stands. Disappearance of exposed sandy lakeshores	Sparse free-floating plant stands and open sandy lakeshores
Presence of nutrient-demanding plants, expansion of free-floating plants, obvious overgrowth of mesotrophic water indicator species and green algae	Presence of oligotrophic-mesotrophic water indicator species
Algal bloom	Absence of nutrient enrichment indicator species
Low water transparency	High water transparency
Overgrowth with floating-leaved and emerged aquatic plants exceeds 50% of the lake surface area	Overgrowth with floating-leaved and emerged aquatic plants does not exceed 30% of the lake surface area
Measured debris and sludge accumulation	A small layer of organic debris

Figure 4.7. *Indications to changes in the river environment. All photos: L. Vizule-Kahovska.*

Groundwater-dependent rivers and groundwater-dependent river reaches

Rivers are open systems that collect water from precipitation through a drainage basin from surface runoff and groundwater recharge. As a lotic system, rivers are associated with vast surrounding areas (Urtāns (ed.) 2017). Therefore, only springs and large and medium-sized river reaches can be

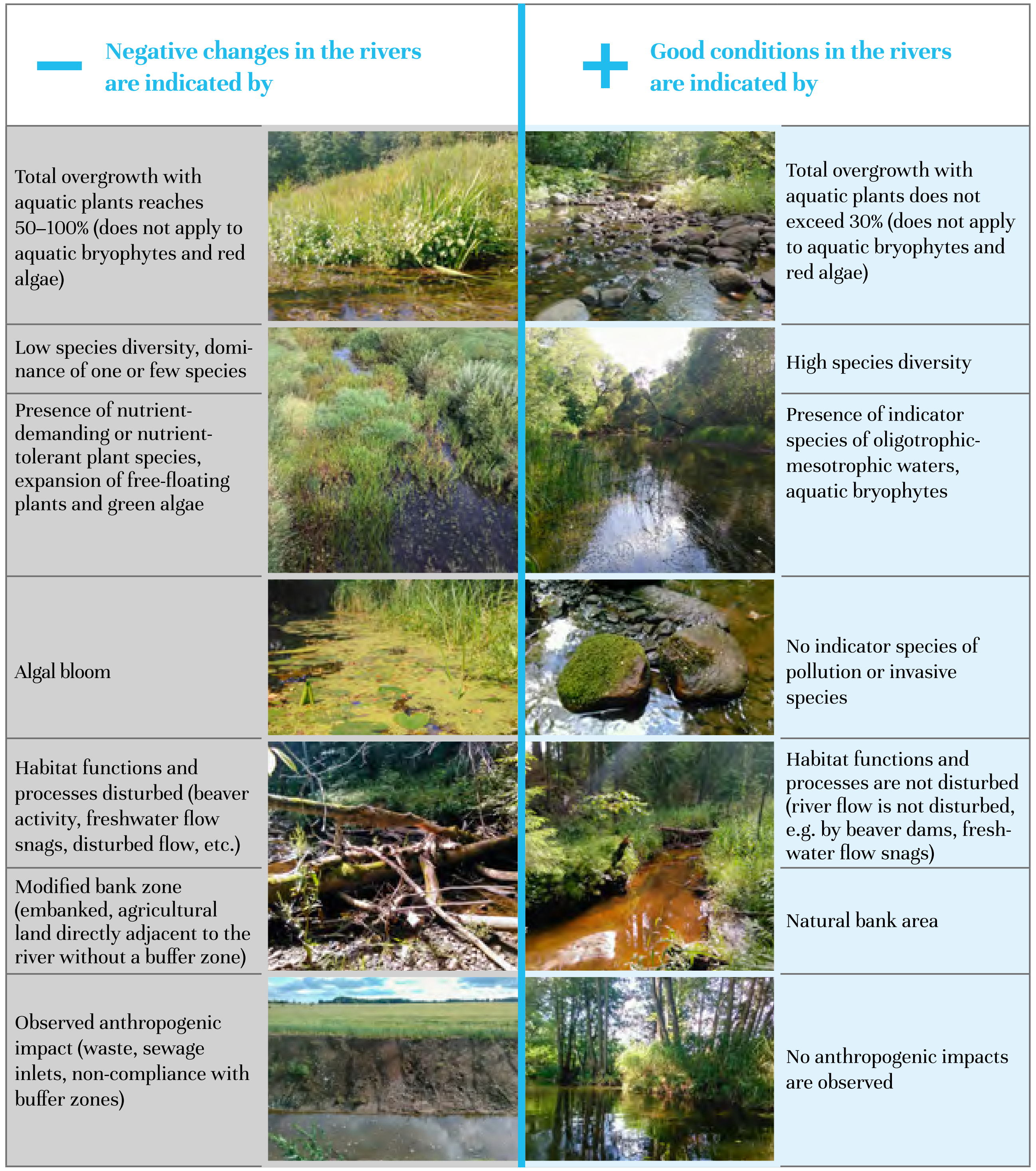


Figure 4.8. *Indications for deterioration of GDE quality in rivers (on the right - good quality; on the left - the result of deterioration). All photos: L. Vizule-Kahovska.*

classified as GDE. Both springs and GDE river reaches are characterized by a continuous flow of cold water, a uniform temperature and are rich in oxygen and minerals (Averis 2003, Blaus et al. 2020). The water temperature fluctuates around 18 °C during the summer and only temporarily exceeds 20-22 °C (Birzaks 2013).

These GDEs belong to habitat type Water courses of plain to montane levels with Ranunculion fluitantis and Callitricho–Batrachion vegetation (3260),which includes a subtype – a riverbed covered with boulders or pebbles, with a stream velocity> 0.2 m/s. The most important precondition for the existence of this habitat type is a stream velocity exceeding 0.2 m/s, a river gradient >1 m/km and a low water temperature. The temperature of these habitats is determined by GW connectivity and/or optimal riverbank sunlight, where the sunlit and shady shores form a mosaic with a ratio of 30:70 (Urtāns 1989; Anon. 2002). Therefore, an increase in water temperature may indicate a loss of connectvity to GW, but may also be caused by climate change or inadequate riverside sunlight conditions (Figure 4.8.). Due to the complex interactions of the variables, a GDE impact assessment scheme cannot be developed for GD rivers (river reaches), but it is recommended to use a general biotic indicator scheme and more precise physical and/or chemical indicators to assess the GDE status.

4.3.6. Response of aquatic plants to nutrient enrichment in water or sediments

Loreta Urtāne (society „WaterScape"), Lauma Vizule-Kahovska (Nature Conservation Agency, Latvia)

Very clean and nutrient-poor lakes with sandy littoral are characterized by a *Lobelia-Isoëtes* complex that forms rosette-shaped bottom vegetation. Species of this complex, such as *Lobelia dortmanna*, are slow-growing and adapt to oligotrophic conditions (Roelofs 1983). These plants have low competition capacity because their leaves are located closer to the lake bed, but the leaves of submerged aquatic plants located closer to the lake surface are able to reduce the availability of light (Sand-Jensen & Vestergaard 1999). As the lake becomes eutrophic, the species characteristic of oligotrophic waters disappear and are replaced by more nutrient-demanding and light-tolerant species of submerged aquatic plants (Pokorný & Björk 2010). For example, in the first half of the 19th century, *L. dortmanna* was found in at least 64 lakes in Latvia (Urtāns (ed.) 2017), of which the species has survived in no more than 14 lakes (Suško 1999). Most of them are GDE, such as Lake Ummis, Lake Ārdavas, Lake Sīvers, Lake Mazuikas, Lake Dridzis, Lake Laukezers, which are closed-basin clear-water lakes. Lake Ummis is considered to be one of the most outstanding lakes in the *Lobelia-Isoëtes* complex in Latvia (Latvijas dabas fonds 2020).

In eutrophic lakes, the plants have excellent feeding conditions. As the concentration of nutrients rises, the total diversity of aquatic plant species decreases, but the number of individuals of each species increases (Dudley et al. 2008). Species that require nutrient-rich waters are beginning to spread, overgrowth in the coastal zone is becoming denser, creating shading and competing with floating-leaved and submerged vegetation (Pokorný & Björk 2010). Eutrophic lakes generally have unsuitable growth conditions for submerged aquatic plants, as in shallow waters there is competition for light and space with emergent macrophytes and floating-leaved aquatic plants, but in deeper waters, the amount of light is limited by high phytoplankton concentrations (Sand-Jensen & Vestergaard 2000).

The highest diversity of total and submerged aquatic plants can be observed in mesotrophic and weakly eutrophic lakes. At extremely high nutrient concentrations, species diversity is negligible (Dudley et al. 2008). Such nitrogen-rich lakes are usually dominated by free-floating aquatic plants, phytoplankton and periphyton, while the diversity of aquatic plants rooted in the soil is low (Tracy et al. 2003) (Figure 4.9.).

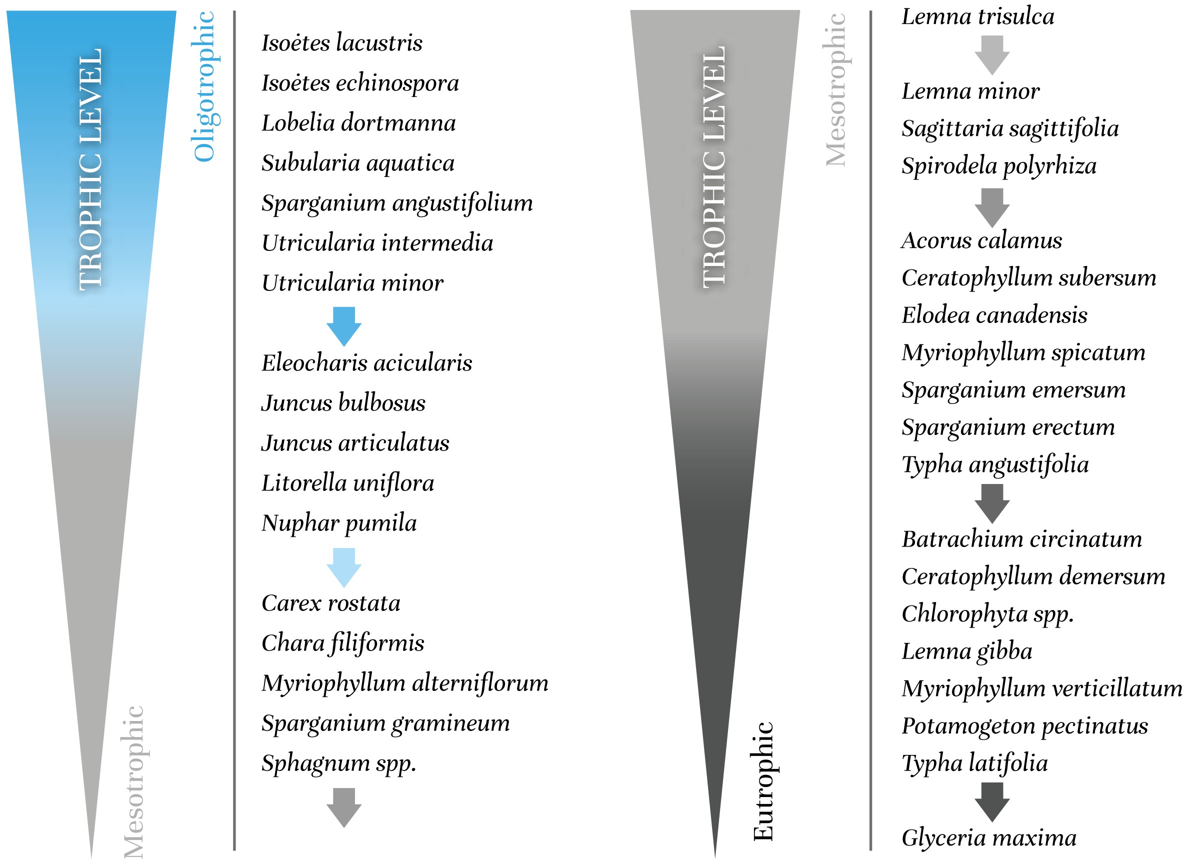


Figure 4.9. *Macrophytes as indicators of trophic level.*

The lake trophic level does affect not only the composition of aquatic plant species, but also the maximum depth of their occurrence in the lake and overgrowth (Markarger & Middelboe 1997). As the concentration of nutrients increases, the depth of occurrence of aquatic plants decreases. Floating-leaved aquatic plants and algae, along with suspended solids, also affect water transparency. Oligotrophic and mesotrophic lakes rarely reach more than 20% of the total overgrowth with aquatic plants, while eutrophic and hypereutrophic lakes can potentially reach up to 100% of the total overgrowth (Gasith 1998).

The effects of eutrophication on the composition and abundance of aquatic plant species are also applicable to rivers. It is important to note that eutrophication is in itself a natural and irreversible process, but anthropogenically induced eutrophication is more dynamic and significantly accelerates it. It is also important to distinguish the factors influencing natural aquatic plants from anthropogenic ones. For example, the natural factors influencing aquatic plants in lakes are water transparency, water color, water hardness and water pH, while in rivers – river width and depth, current speed, shading and soil composition.

The correlation between environmental factors and the occurrence and development of aquatic plants described above are applicable only to closed-basin and karst lakes, as well as GW dependent rivers. Such processes are not observed in other GDEs. However, there are indicator species – free-floating aquatic plants and algae, which in any aquatic environment indicate an increase in nutrients.

Response of aquatic bryophytes

Līga Strazdiņa (Nature Conservation Agency, Latvia)

The term *aquatic bryophyte* generally refers to mosses and liverworts that are able to conduct all or part of their life cycle submerged in water. Relatively few bryophytes are *obligate aquatics*, that is, submerged for most of their life cycle and having little or no tolerance to dry conditions. *Facultative aquatics* are able to tolerate considerable fluctuation in water level, sometimes totally submerged, and can withstand short or extended periods of desiccation and xerophytic conditions. *Semi-aquatic emergents* grow in locations where they are partly in the water and partly out of it, but usually moist (Porley & Hodgetts 2005; Glime 2021).

From an ecological perspective, aquatic bryophytes can be classified as limnophilous, rheophilous or semi-emergent. In the context of GDEs, limnophilous species are typical in fens, pools, ponds and other water bodies where the water is stagnant or slowly moving. These plants are typically rather flaccid with large, thin-walled cells and often have a reduced midrib, such as *Drepanocladus, Scorpidium, Warnstorfia, Riccia, Ricciocarpos, Porella* (Figure 4.10.). Rheophilous mosses are characteristic of flowing water, such as streams and rivers. Some of these species show adaptations to physical stress and periodic exposure, such as stiff wiry stems, thickened leaf margins, and small, thick-walled cells. Example genera are *Fissidens, Fontinalis, Racomitrium, Schistidium, Hygrohypnum*. The semi-aquatic emergents are rarely completely submerged, the growing shoot, including the sporophyte, is exposed to the air. Such species retain water efficiently using paraphyllia and tomentum or other external and internal structures. Examples are *Sphagnum, Calliergon, Campylium, Palustriella, Paludella* that are found in springs, swamp woods and also in flood-zone of the riparian habitat. Standing open water supports a range of aquatic and semi-aquatic bryophytes, including a specialised suite of species that are not strictly aquatic, but appear as water levels fall. Mosses in general tend to be more tolerant of

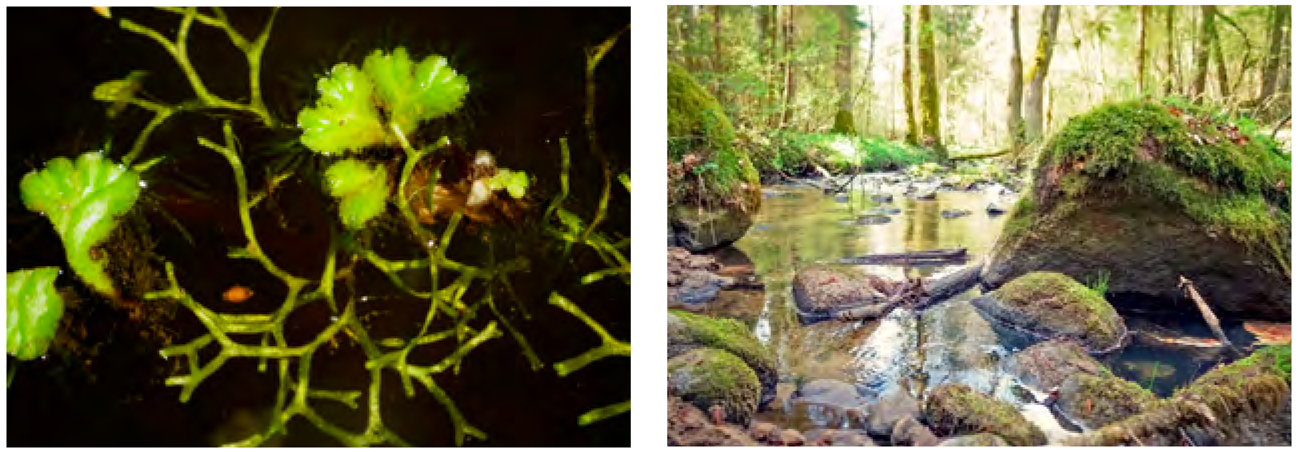


Figure 4.10. *In slow-moving ditches near swamp woodlands in Latvia, two limnophilous liverworts are found: fine filaments of Riccia fluitans between heart-shaped thalli of Ricciocarpos natans (photo on the left). Rheophilous bryophytes are common on sunken rocks and on the banks of small watercourses (on the right). Photos: L. Strazdiņa.*

desiccation than liverworts, and this goes for aquatic bryophytes too (Porley & Hodgetts 2005; Glime 2021).

It is useful to categorize environmental variables on the basis of their spatial and temporal characteristics. Stream bryophyte communities are influenced strongly by mesoscale variables (include a stream's hydrological and nutrient regime) and macroscale variables (such as climate, altitude, geology, and land use) related to substrate stability, and reflect different abilities of genera to grow under different light, nutrient, and hydrological regimes (Figure 4.11.). Mosses are typically found on stable substrates because they usually require a long time to become established. For example, bryophyte abundance is often relatively high in ecosystems with limited flow variation or bed movement, such as in spring-fed streams. Factors that control the potential source of colonists within a particular catchment are also relevant (Stream Bryophyte Group 1999).

Variable		Presence of bryophytes		Resultant community
MACROSCALE				
Land use	Modified	No (few)		
	Unmodified	Yes	Some modification	Mosses
			Natural catchments	Liverworts
Geology	High erodibility	No		
	Low erodibility	Yes	Strong sedimentary, igneous	Mosses
			Metamorphic, igneous	Liverworts
MESOSCALE				
Hydrology	Many low flows	No		
	Few low flows	Yes	Infrequent flooding	Mosses
			Frequent flooding	Liverworts
Water quality	High nutrients	No		
	Low nutrients, calcium rich	Yes	High phosphorus	Mosses
			Low phosphorus	Liverworts

Figure 4.11. *Summary of the interaction between macro- and mesoscale variables and the presence or absence of aquatic bryophytes (from Stream Bryophyte Group (1999)).*

Availability of CO_2, the water temperature, pH, and rate of flow in streams, spectral quality and water clarity, the amount of light, accumulation of pollutants in lakes are some of the most important limiting factors of aquatic bryophyte diversity. Since bryophyte leaves are only one cell thick they permit easy entry of pollutants and are therefore good bio-indicators. Whilst many species are sensitive, others are tolerant or have specific responses. The ability of aquatic bryophytes to sequester heavy metals and radionucloides in their cell walls and vacuoles enables them to survive in conditions where many vascular plants are excluded. Aquatic bryophytes are better able to survive at low light levels than vascular plants. This is because, at low temperatures, aquatic mosses have a low compensation point at which rates of CO_2 intake and output in photosynthesis and respiration are equal. The ability of mosses to grow slowly reduces their need for CO_2 and light (Porley & Hodgetts 2005; Glime 2021).

Fontinalis antipyretica (Figure 4.12.) is the best-known bryophyte of rivers and streams in the Baltic region. It occurs in oligotrophic water but is more often found in mesotrophic to eutrophic water, from neutral to basic. *Fontinalis* is sensitive to pollution and is well grown only in clean water (Porley & Hodgetts 2005).

Figure 4.12. *Greater water-moss Fontinalis antipyretica has sharply keeled leaves and trailing shoots that are able to move easily with the water flow in Koja River in Western Latvia. This moss grows in slow to moderately rapid water. Photos: L. Ķeire.*

5. Threats to groundwater and groundwater dependent ecosystems

5.1. General insight

Jekaterina Demidko, Dāvis Borozdins (Latvian Environment, Geology and Meteorology Centre)

Groundwater (GW) is an important source of water for humans and wildlife all over the world. As a result of increased human water demands and other anthropogenic factors, groundwater-dependent ecosystems (GDEs), which rely on GW for some or all of their water needs, are becoming more vulnerable across the world.

Human activities, such as agricultural practices, urban and industrial growth, mining, and forestry, are mostly responsible for changes in GW quantity and quality. GDEs are threatened by human activities that disrupt the biota and soil, deplete GW reserves, change the GW regime at a location beyond the natural boundaries of fluctuation historically seen at that site, and degrade GW quality. These concerns have the potential to have significant short- and long-term repercussions on GDEs at local and regional scales (Jakeman et al. 2016; Rohde et al. 2017). GDEs may experience considerable structural and functional alterations even with mild to moderate changes in GW quantity and/or quality. Ecosystems that are wholly or heavily reliant on GW are particularly vulnerable (Jakeman et al. 2016; Burri et al. 2019). Climate change impacts (for example, extended droughts) and some other local natural elements, such as beaver activity, may also have an impact on ecosystems ecological state. Natural succession generates structural changes and turnover of biotic communities in the ecosystem through time; nevertheless, unless the aim is a certain uncommon biotic community or species, this cannot be regarded as an undesirable shift in terms of naturalness.

More information on potential threats to GDEs is provided in the following sections (see also Chapter 6 on restoration and mitigation).

5.2. Groundwater quantity

5.2.1. Impacts to groundwater regime

The main anthropogenic threats or human activities that may affect the GW regime (quantity), posing a serious threat to GW resources and the status of GDEs, are summarized in Table 5.1. includes information on important GW parameters for GDEs.

Four factors define the impact of a changed GW regime on ecosystems: (1) the ecosystem's reliance on GW; (2) the speed at which GW levels change (drawdown rate); (3) the time span over which the change occurs; and (4) the seasonal timing of the shift. These elements often act individually or together. As a result, even minor changes within the GW regime may have a severe influence on ecosystems that are entirely or partially reliant thereon, moreover as those who occupy particularly specific ecological niches. Determining the character of this dependence, on the opposite hand, is usually difficult. Determining whether or not a threshold reaction exists is extremely complex and time-consuming (Serov et al. 2012).

Table 5.1. *Anthropogenic threats to groundwater regime and importance of groundwater parameters for GDEs (modified from Jakeman et al. (eds.) 2016)*

Anthropogenic threats to groundwater regime	
Agricultural practices	• Reduced groundwater level/pressure due to excessive groundwater abstraction to support agricultural development; • Groundwater recharge due to surface water pumping for irrigation; • Water-logging due to vegetation clearing and poorly managed irrigation.
Urban and industrial development	• Reduced groundwater level/pressure due to excessive groundwater abstraction to support urban and industrial development.
Mining activities	• Reduced level, pressure and flux due to mine dewatering; • Reduced level due to channel incision (e.g. gravel mining).
Forestry	• Reduced groundwater recharge and surface flow; • Increased groundwater discharge.

Importance of groundwater parameters for GDEs		
Depth-to-groundwater	**Groundwater pressure** (hydraulic head and its expression in groundwater discharge)	**Groundwater flux** (flow rate and volume of groundwater supply; flow direction)
• Accessible water at root zones; • Prevent water-logging; • Provide wetness or water-logged environment; • Prevent activation of acid sulphate soil; • Maintain hydraulic gradient for groundwater discharge; • Provide habitat for species; • Maintain groundwater stratification.	• Sustain groundwater discharge to springs.	• Sustain water uptake rate; • Sustain above ground wetness (wetlands); • Sustain base flow; • Prevent saltwater intrusion (estuarine/coastal environment); • Supply organic matter and oxygen.

The primary GW characteristics that may impact the status of the GDEs are depth-to-groundwater, groundwater pressure, and flux. For unconfined aquifers, depth-to-groundwater is critical, while for confined aquifers, groundwater pressure or hydraulic head, as expressed in groundwater discharge, is crucial (Jakeman et al. (eds.) 2016; Rhode et al. 2017; Serov et al. 2012).

In general, changes in GW factors have a gradual impact on biotic parameters including growth, reproduction, regeneration, mortality, and the ecosystem's structure and functioning. Organisms that have evolved to short-term stress owing to a lack of water are found in GDEs. When stress is sustained or acute, however, these adaptations become insufficient, resulting in population decrease and changes in ecosystem structure and function (Figure 5.1.).

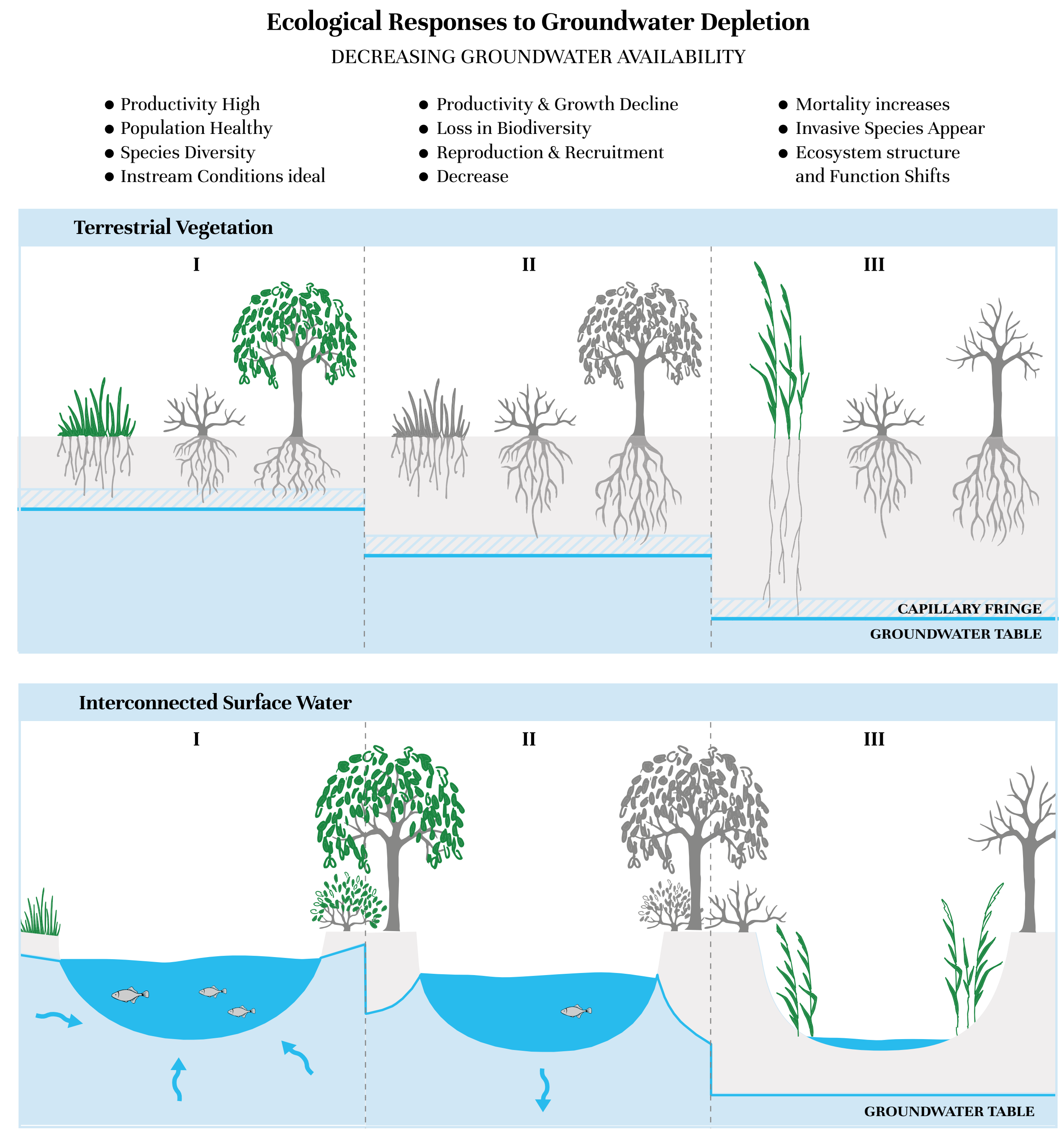


***Figure 5.1.** Ecological responses to groundwater depletion (modified from Rohde et al. (2017)).*

Depth-to-groundwater (from the land surface) is one of the most important GW parameters for GDEs. This is especially true for terrestrial ecosystems that rely on GW from the subsurface for their survival. GW availability to plants is influenced by the distance between the capillary fringe above the water table and plant roots, as well as the depth-to-groundwater. Reduced plant growth, greater mortality, and a shift in species composition may result from increasing depth to groundwater. Aquifer ecosystems may lose habitat as a result of lowering the water table. A rising water table, on the other hand, may disfavor species that are susceptible to waterlogging and result in the succession of distinct plant communities. Groundwater pollution can also be caused by changes in the water table depth, as well as other environmental causes. Lowering the water table beneath acid sulfate soils, for example, causes pyrite to oxidize and the shallow aquifer to become acidic (Jakeman et al. (eds.) 2016).

Groundwater flow is critical for GDEs because it keeps plants hydrated. Reduced **GW pressure and flow** result in lower groundwater discharge and, as a result, lower surface water availability for wetlands and GDEs that rely on springs and base flow. Reduced GW flow in estuaries and coastal locations causes seawater intrusion and pollution of coastal freshwater aquifers, lowering GW quality. Appropriate GW flow is critical for aquifer ecosystems to sustain a supply of organic matter and oxygen to the stygofauna that live there. GDEs created in naturally highly variable places (e.g., areas with substantial climatic seasonality) have usually adapted to GW regime variations and are therefore more robust to GW regime change than those developed in areas with a more stable regime (Jakeman et al. (eds.) 2016).

Daily GW level changes, as well as monthly and seasonal fluctuations, can be detected under natural situations, depending on the hydrogeological, meteorological, and climatic variables of each site. Water and land use development activities have the ability to change any of these factors, and hence the water regime required by certain GDEs. This may lead to changes in the composition and function of an ecosystem in the immediate region of the activity, as well as in ecosystems that rely on groundwater (Serov et al. 2012).

Excessive (heavy) GW abstraction is one of the most serious problems. GW abstraction aims to help agricultural operations as well as urban and industrial development. In these instances, pumping wells in confined or unconfined aquifers are usually employed to extract GW. Excessive GW pumping in a confined aquifer lowers GW discharge to springs by depressurizing the whole confined aquifer (the impact is at a regional scale). Pumping GW from an unconfined aquifer has a more localized effect. When abstraction exceeds recharging in unconfined aquifers, GW depth increases, producing a "cone of depression" around the well that can stretch over hundreds of meters (Figure 5.2.) (Jakeman et al. (eds.) 2016; Rohde et al. 2017).

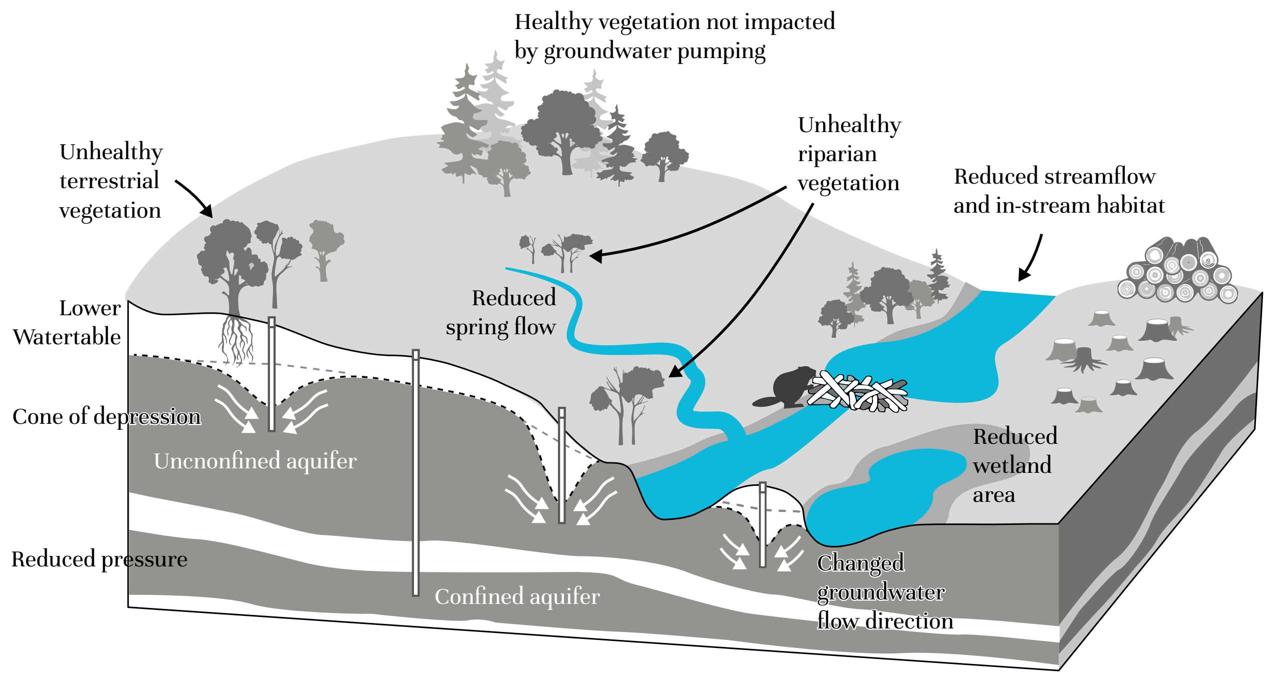


***Figure 5.2.** Diagram showing the potential impacts of groundwater pumping on GDEs (modified from Jakeman et al. (eds.) (2016)).*

Furthermore, new hydraulic gradients may affect the direction of GW flow: GW may no longer flow into the local stream, and some water from the stream may be redirected to the well, decreasing stream flow. Depending on abstraction sites (relative to the stream), abstraction volume and GW flow rate, and volume of GW supply and flow direction, the time lag between abstraction and a reduction in discharge to a stream can range from a few hours to several millennia (Jakeman et al. (eds.) 2016).

The GW regime can also be affected by **mining.** It can have a negative impact on the ecosystems and change hydrogeological settings and GW regime, such as fluctuation of water level and depletion of GW (Ozcan et al. 2012). GW and surface water changes that are caused by quarrying can indirectly affect and damage habitats by overflowing or drying out. On the other hand, quarries can also create new habitats (Řehounková et al. (eds.) 2011; Priede 2017).

Pumping out water from quarries can cause the local GW level depletion – the depression cone. When the mining proceeds below the GW table, pumping the inflow of GW out of the quarry is performed to continue mining. This pumping can lower the GW levels in areas around the quarry. Lowering GW levels can lead to the formation of a local depression cone around the quarry, which in turn can affect GW levels in nearby wells or boreholes used for household water supply (DEP 2017). An example from Estonia on how a combination of the two factors mentioned above (intensive water abstraction and quarrying) can affect a GDEs is described in Chapter 5.2.2. Mining, related drainage and water abstraction can affect both shallow GW (unconfined) and artesian (confined) aquifers.

Deforestation, especially intensive large-scale deforestation, affects hydrological processes both surface and subsurface. Extensive removal of trees leads to reduced evapotranspiration, increased surface water runoff, GW recharge, as well as reduced infiltration (at constant precipitation). These processes can result in the rise of shallow GW table and create water-logged conditions and paludification.

Forest tree cover is essential for the infiltration of water and rainfall into the soil, and thus ultimately for the recharge of GW resources. Tree cover facilitates these processes by providing shade and litter cover under trees that reduces soil evaporation and enhances downward water seepage, thereby enhancing soil organic matter. Likewise, tree roots and faunal activity enhance macroporosity, which further drives infiltration and ultimately groundwater recharge (Ellison 2018).

However, the status of GW and GDEs can be affected not only by human activities. The Eurasian beaver (*Castor fiber*) can intentionally cause habitat transformation and create a specific environment by erecting dams and establishing ponds. Regarding the situation in GW, beaver dams, by affecting hydrological conditions, can cause GW levels to rise next to the upstream ponds. Changes in GW level depend on the height of the dam, relief and geological conditions of the area. In lowlands and large depressions, the beaver ponds may have a destructive effect on spring outflows and surrounding spring-fed ecosystems and related species, while on spring discharges on slopes there may be minor effects covering small areas (Priede 2017). The beaver activity may cause changes in the vegetation and faunal compositions by supporting the establishment of species adapted to high water levels (Retike et al. 2020). However, beaver dams can also slow down water level decline during drought periods, ensuring a constant flow path both in surface waters and GW (Westbrook et al. 2006).

Changes in climatic factors, such as changes in temperature and precipitation, can also have a substantial influence on GDEs (Panwar 2020). Due to the increase of temperature, the duration of the winter ice period decreases and the summer water temperature increases. Changes in precipitation patterns impact river water levels, which are especially important during the summer when water levels are at their lowest. Though there may be significant local variances, climate change's impact on GDEs, especially when paired with other variables like land-use change, may have a detrimental impact on many GDEs (Cartwright et al. 2020).

5.2.2. Effect of the groundwater abstraction and mining on the water ecosystems – a case study of Kurtna Lake District

Jaanus Terasmaa (Tallinn University)

Northeastern Estonia is a typical case for the influence of mining on the GW table. In this region, Estonia excavates its main mineral energy resource – oil-shale, but in many cases also other natural resources – sand, gravel, and peat. In total, the former and current oil shale mines cover around 1% of Estonia's territory, at the local scale the impact is enormous: the water pumped from the mines and used by oil-shale power stations as coolant water amounts to around 90% of all water used in Estonia. As underground mines are using room and pillar mining methods, the surface water flowing into the mines must be pumped out (Väli et al. 2008). Around one quarter of the water is also pumped out from the open-pit mines. All this alters not only the circulation but also the quality of GW and releases mine water into surface water bodies (Raukas & Punning 2009).

Kurtna Lake District lies between the underground mines and open-pit mines – it is the largest lake district in Estonia, where 38 natural lakes are located in a 30 km^2 area. Kurtna kame field lays over the ancient Vasavere valley, which cuts through the Ordovician rocks and is filled with sand and gravel. The kame field over the Vasavere valley was formed by a retreating ice sheet during the end of the Weichselian glaciation around 12 200 years ago (Karukäpp 1987). Kurtna lakes are situated in and around the kames, ranging from 40 to 70 m a.s.l, separated by kettle holes between them (Ilomets & Kont 1994). The highly permeable sediments in the Vasavere valley contain an unconfined Quaternary GW aquifer, which is partly separated from the surrounding Ordovician aquifer by the low-permeability till layers (Vainu et al. 2020). The lakes are limnologically very diverse, and their size range from very small (0.1 ha) to medium (146 ha). Nowadays Kurtna is part of the Alutaguse Natural Park (Figure 5.3.) but attempts to protect this unique territory started already in 1987 by forming the Kurtna Landscape Conservation Area. Many lakes are also protected under the EU Habitats Directive. Despite all the protection effort, all lakes in Kurtna are affected by human activities, water level drop is being the main reason for ecosystem deterioration in the number of cases.

Noticeable human influence on the lake's hydrology started before oil-shale mining with ditching and peat-cutting and can be traced back to the 19th century (Kivioja 2017; Vainu et al. 2020). During World War II, the pine forests in the central part of the kame field were burnt (Mäemets 1987) causing an abrupt change in the vegetation and its evaporative capacity, which may be one reason behind the lake level changes between 1940 and 1960 (Vainu 2018). But in general, the lakes were more or less in a natural state until 1946, when the first underground oil shale mine "Ahtme" was established northwest of the lake district. "Ahtme" mine affected GW levels and flow directions, as the closest the mine

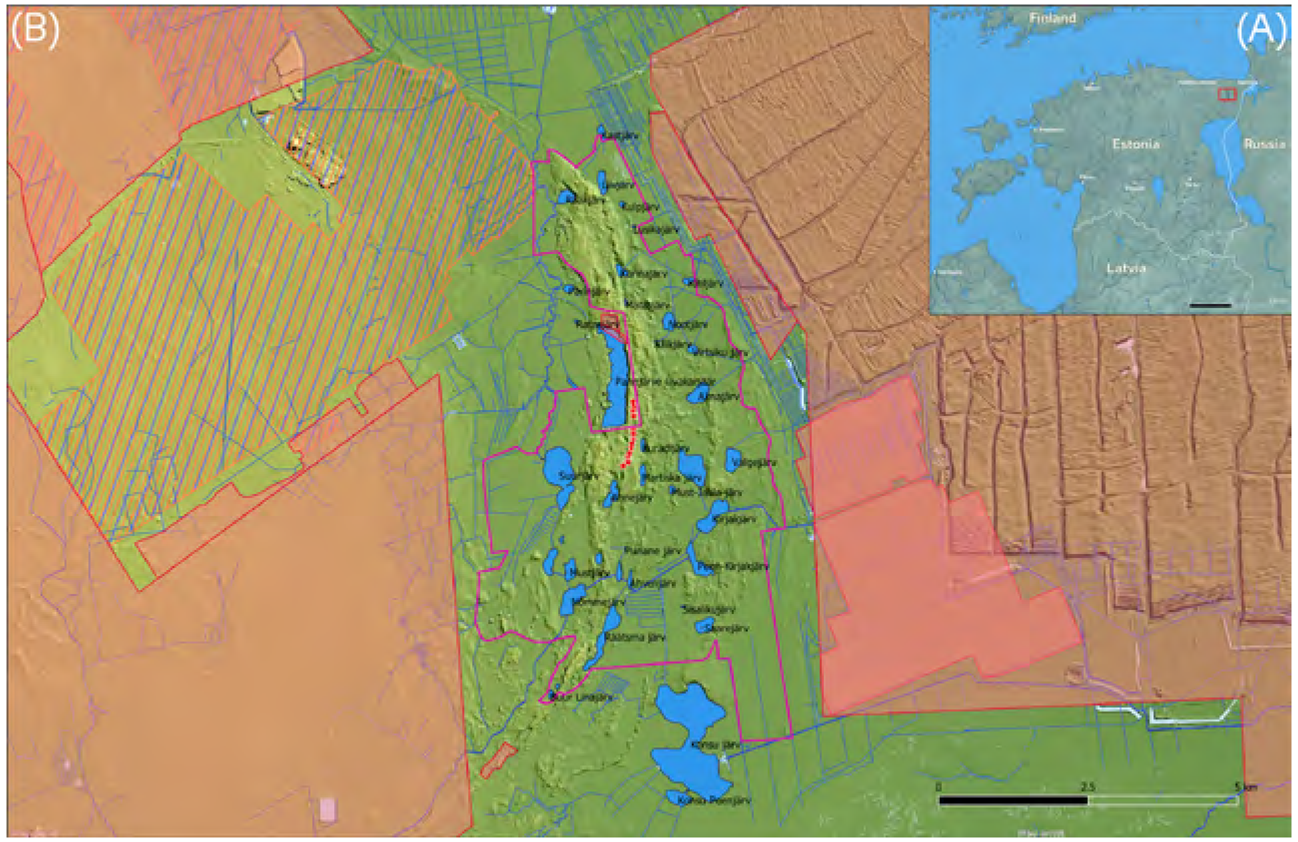


Figure 5.3. *(A) Location of the Kurtna Lake District (red box) and (B) Kurtna Lake District in Alutaguse Nature Park (inside the purple line), surrounded by mines (areas with red shading). Water bodies are blue. Red stars are groundwater intakes. Map: Estonian Land Board (2021).*

border was to the lake district was 400 m. In 2002, it was closed and refilled with GW.

In 1948, an industrial surface water intake was established in Lake Konsu to supply the oil-shale chemistry industry with water. During the following decades, the high water demand caused expansion of the system. Nowadays, only water from the "Estonia" mine is pumped into the system, and nine lakes are connected to it (Terasmaa et al. 2014).

In the east of the lake district in 1962, open-pit oil-shale mining started in "Sirgala". The water level in "Sirgala" has been lowered by more than 20 m. It is only half a kilometre from the closest lake and is still operating. Since the 1990s, the mining company is using filtration ditches and infiltration barrier for the protection of the lakes. In 1973, a new underground oil-shale mine "Estonia" was opened to the south-west of the district. That mine is currently operational and is considered to be the largest oil-shale mine in the world. Studies are showing that currently the GW level lowering caused by "Estonia" does not reach the lakes in Kurtna (Terasmaa et al. 2019; Vainu et al. 2020).

In 1964, a peat field "Oru" was established east of the lake district. The drainage system of the peat field affected the nearby lakes dramatically – the level of Lake Liivjärv dropped by 2 m in 15 years. Peat milling is continuing, but in the near future mining of the underlying oil-shale will start.

Kurtna-Vasavere GW intake was built in the centre of the lake district in 1972. The intake provides the nearby towns with drinking water. The pumping rate was designed to be over 20000 m^3/day, but in reality, it has ranged from 4000 m^3/day to 10 000 m^3/day. Vainu & Terasmaa (2016) have shown

that GW pumping has considerably affected the water levels of the closest lakes – especially lakes Martiska, Kuradijärv and Ahnejärv.

It has been estimated that since the 1950s the water level has dropped in at least 24 lakes because of drainage of the oil-shale pit and mines, peat cutting, sand extraction and GW abstraction in the centre of the region (Kink et al. 2001). Lake level changes during different time periods are given in Figure 5.4.

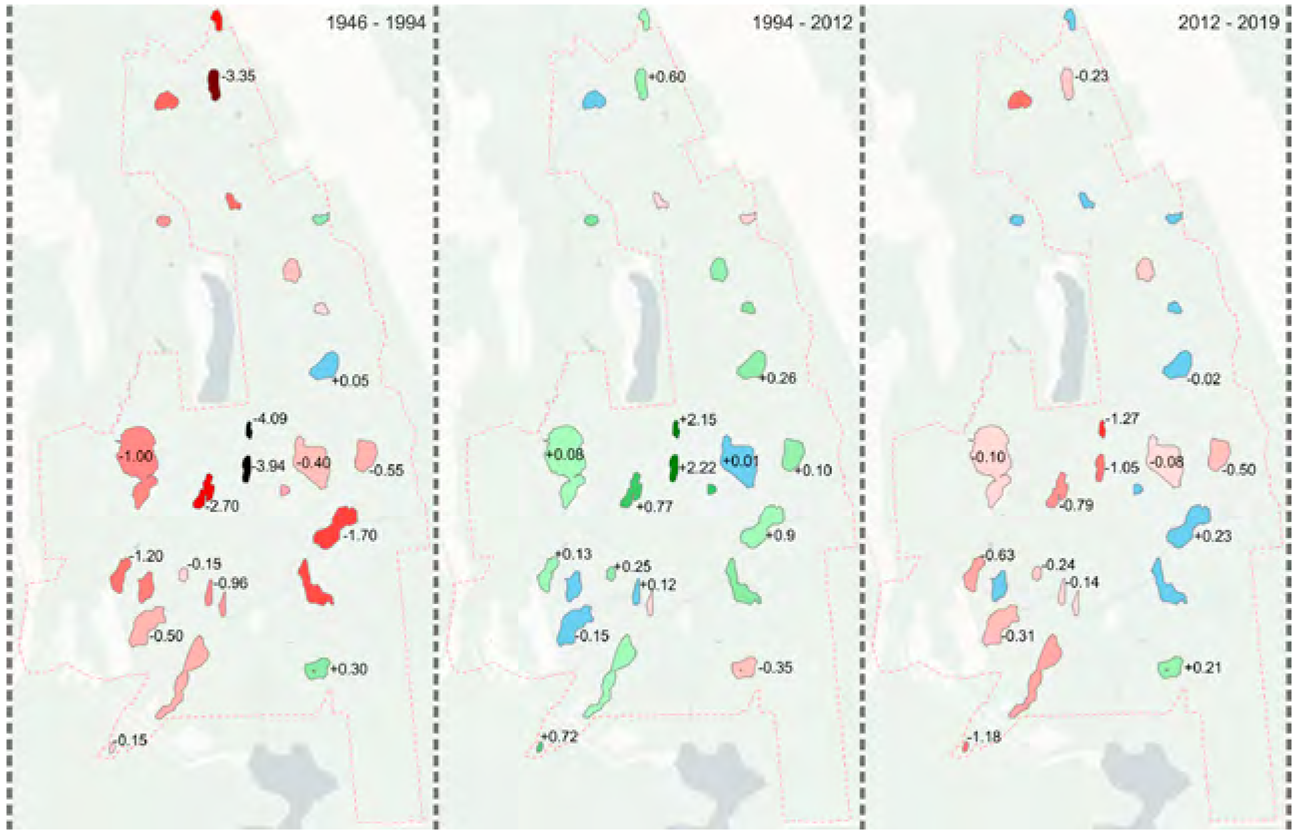


Figure 5.4. *Lake level changes (in meters, green – increase; blue – no change; red – decrease) in Kurtna Lake District (source: kurtna.ee/gis).*

Based on the ecological and scientific value of the lakes and the type of anthropogenic influence it is possible to distinguish clusters of lakes under similar pressure. In the centre of the lake district Lake Martiska, Lake Kuradijärv and Lake Ahnejärv are forming one group of lake habitats protected by the EU Habitats Directive, but under a strong anthropogenic influence due to GW abstraction.

Lake Martiska is a closed-basin GW dependent lake. It has been one of the ecologically most valuable soft and clear-water oligotrophic lakes in Kurtna, but it has suffered from large-scale water level fluctuations that have considerably reduced its ecological value (Ott et al. 1995; Põder et al. 1996; Vainu et al. 2020). In 1946 the lake area was 4.4 ha (known maximum) and in 1987 it was 1.3 ha (known minimum). During the last decade, it has been between 2 and 3 ha. The closest well of the intake is only 200 m from the lake. The natural GW flow direction in this region was from west to east. GW abstraction lowered the GW, which has caused the formation of a cone of depression around the GW intake and forced lake water to seep towards the intake. The result of the conducted research (Vainu & Terasma 2016; Vainu 2018) confirmed the previous findings – in the centre of the Lake District, the water balance of the lakes is strongly GW controlled, and therefore the most significant factor since 1972 has been GW abstraction. Meteorological conditions have either mitigated (during wetter periods) or worsened (during drier periods) its effect (Figure 5.5.).

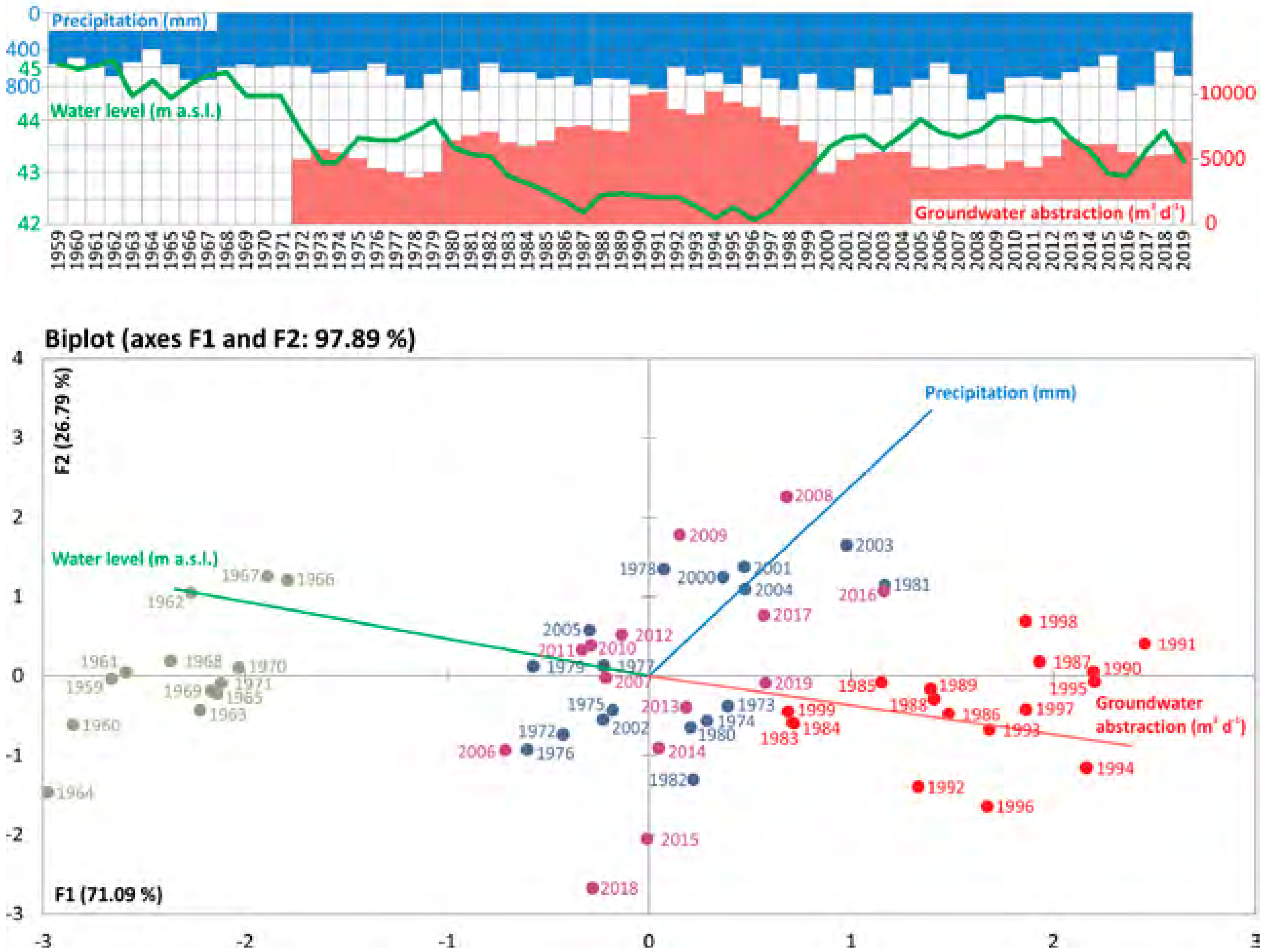


Figure 5.5. Reconstructed water level changes and their connection with groundwater abstraction and precipitation. Green dots denote the "natural" period before groundwater pumping, red indicates the period with maximum water abstraction, blue and violet are transition periods (based on Vainu 2018; renewed by Terasmaa 2019).

The GW abstraction was the highest in the period from 1980 to 1990 and dropped considerably at the end of the 1990s, staying lower until 2012. That allowed the lake level to recover to some extent. In 2012, the wells and other infrastructure were renovated and abstraction increased over critical levels, which tends to be ~4500 m3/d. The lake level reacted immediately and started to drop until 2016. The continuous drop of the lake level was supported by two consequent very dry years but have not recovered since then even during the rainy years.

It is well known and studied (Håkanson 1977; Digerfeldt 1986; Terasmaa et al. 2013), that lake level fluctuations are causing changes not only in lake morphometry, but also in lake ecosystem and its trophic status. In Lake Martiska, the regression of the lake has resulted in extensive erosion and redeposition of sediments, changes in the distance to the shore and displacement of the erosion–transport–accumulation zones (Punning et al. 2007; Vaasma et al. 2015). During the lake level fluctuations, accumulated sediments originate in principal from two different sources: in-lake concurrently accumulated sediments (mainly atmospheric input, the influx from the catchment,

autochthonous organic matter) and matter eroded in the nearshore area from where water has retreated in the course of regression. Lake level transgression will affect near-shore sediments and vegetation even further. Lake Martiska was initially inhabited by *Lobelia dortmanna* and *Isoëtes lacustris* – both species are under protection in Estonia. Before 1970, the lake and its shores were largely free of plants, and the vegetation in the lake was similar to that in 1936 (Riikoja 1940; Vandel et al. 2016). In the mid-1970s, vegetation showed slight changes, but they were not significant, as *Lobelia dortmanna, Isoëtes lacustris* and *Sparganium angustifolium* were still found but showed a decline in abundance. In 1985, *Lobelia dortmanna* and *Isoëtes lacustris* had disappeared – their initial habitats became dry and were inhabited by reeds, grasses and trees (Ott 2001). At the end of 1990s, when the water level started to recover, this region was flooded, but the habitat for protected species did not recover. The seeds of plants are transported during lowering of the water level to areas suitable for seed germination, but during water-level rise, the produced seeds will be settled in the area not suitable for germination. *Lobelia dortmanna* and *Isoëtes lacustris* require clear sandy bottoms for growing, but the newly submerged shore areas are unsuitable for re-colonisation and other vegetation, indicative of eutrophication, has taken over (Vaasma et al. 2015; Vandel et al. 2016; Vainu et al. 2020).

Vandel et al. (2016) hypothesized, based on the gradual disappearance of *Lobelia dortmanna* and *Isoëtes lacustris* in Lake Martiska, that if changes in lake level had been more subtle and extended over a longer period of time, it is possible that these oligotrophic species could have adapted to new conditions. Before the new water level drop in 2012, there was a plan to reintroduce *Lobelia dortmanna*, but currently it is not feasible.

These results emphasise the importance of identifying GW dependent surface water bodies and the mechanisms of the interactions before certified GW yields are set and water abstraction from the aquifer is started or the abstraction rates are increased (Vainu 2018).

Another cluster of lakes are lakes under the pressure from oil-shale mining. Those lakes are located closest to the mines. One representative of this group is Lake Valgejärv – the most valuable lake in the district from the ecological point of view (habitat type 3110). Lake Valgejärv is a closed-basin semi-dystrophic lake: it receives its water from precipitation, an adjacent peatland and GW. It is the only lake in the district where all the three characteristic rare and protected plants *Lobelia dortmanna*, *Isoëtes lacustris* and *Sparganium angustifolium* are still growing. According to the research (Vainu et al. 2020), the direct influence of the mine on the water level of the lake has not been proven yet, but GW level drawdown because of the nearby open-cast oil-shale mine is the most important anthropogenic

Figure 5.6. Water level changes in Lake Valgejärv (data compiled from different sources for gis.ee/kurtna).

risk for the lake. The lake level is registered historically to fluctuate in a one meter range, but in 2015–2016 it reached the lowest known level (Figure 5.6.).

In the autumn of 2015, low water levels were recorded in many of the lakes in the district, probably partially caused by relatively dry weather in 2014 and 2015. In 2015, the sum of precipitation in the Kurtna region was 471 mm (the long-term average is 736 mm). As water level drop in Lake Valgejärv was sharper and faster than other lakes in the region, Vainu et al. (2020) hypothesized that drought was probably amplified by human activities – it did not enable the mining company to pump enough water into the filtration channel between the lake and the Sirgala mine. Regardless of reasons, all these lake level fluctuations are threatening the lake's plant communities, especially *Lobelia dortmanna*, which grows in shallow near-shore water. Another noticeable change in Lake Valgejärv is the deterioration of water transparency and colour. Water transparency (Secchi depth) was 4.4 m in 1954, but only 1.5 m in 2018. This phenomenon can be explained by the high content of humic substances in water because of increased water input from adjacent peatland. Peatland influence has probably grown due to low GW level, as its share in the lake's water balance has dropped (Vainu et al. 2020).

In an attempt to find a balance between human activities and the survival of lake ecosystems, a thorough research project was conducted during 2018–2019 (Terasmaa et al. 2019). The project included not only limnological research but also hydrogeological modelling and mapping. Twelve scenarios with

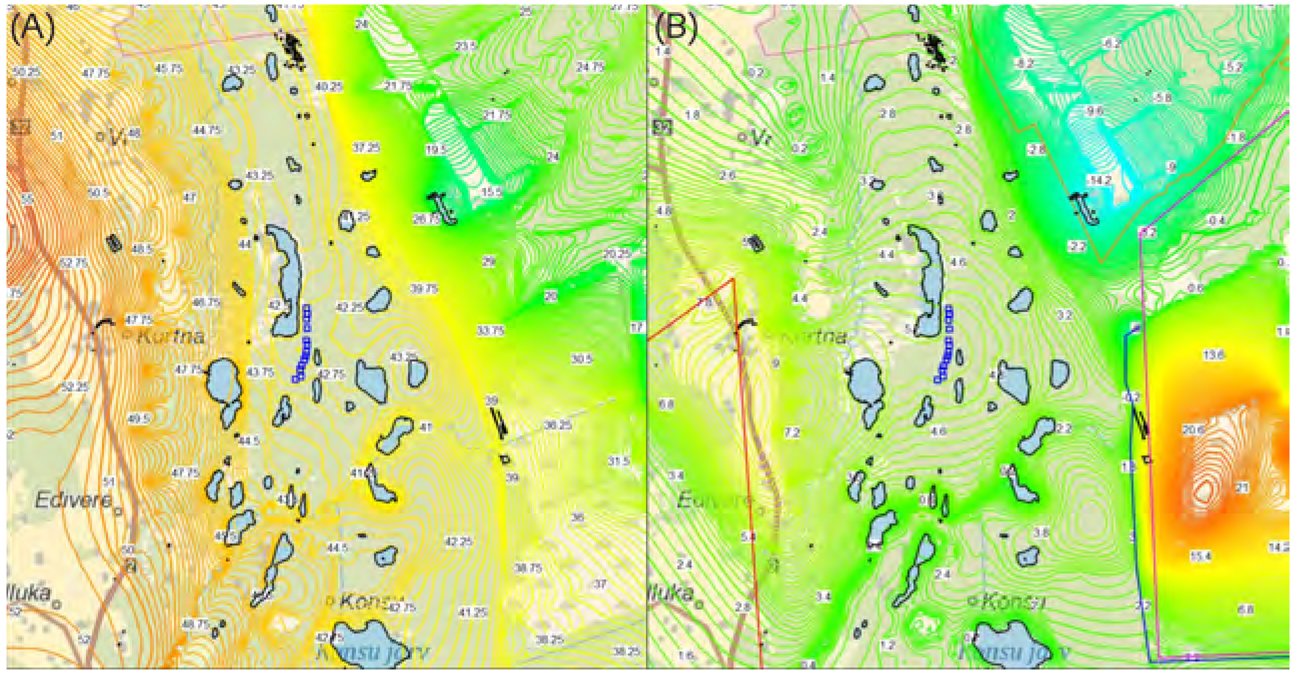


Figure 5.7. Results of the hydrogeological modelling: (A) modelled groundwater level (m, a.s.l) in 2017; (B) Hypothetical scenario #12 where isohypses describe the possible groundwater level drop in meters. This scenario will occur when all possible negative impacts (20% less precipitation, water abstraction 10000 m3/d, maximum reach of oil-shale mines without mitigation measures) emerge at the same time (hydrogeological modelling is made by Argo Jõeleht, maps are from Terasmaa et al. 2019).

different settings (human induced and climatic) and their combinations were tested (Figure 5.7.) and compared with the optimal levels of the studied lakes for ecosystem recovery or at least stability.

The results of the project are not very promising. If the current direction of human activities in the vicinity of the Lake District continues, there are not many changes to achieve needed minimal water levels. As a result, deterioration of lake ecosystems will continue, and there is a high risk that protected species and habitats will disappear in the not too distant future. In the project report (Terasmaa et al. 2019), several possible mitigation measures are pointed out, e.g. changes in the GW pumping locations, changes in oil shale mining and filling the mines with water as soon as possible. In this case, at least some of the protected lakes will have the opportunity to stabilise and the ecosystems will not deteriorate further.

During the project, the maximum possible impact was also tested as scenario #12 (Figure 5.7.,B). In this scenario, it was assumed that all possible negative impacts (20% less precipitation, water abstraction 10000 m3/d, maximum reach of oil-shale mines without mitigation measures) occur at the same time. This scenario was considered not very likely, but not impossible to happen (Terasmaa et al. 2019).

5.3. Groundwater quality

Jekaterina Demidko, Dāvis Borozdins (Latvian Environment, Geology and Meteorology Centre), Kristiina Ojamäe (Estonian Environment Agency)

GDEs are highly sensitive to changes in environmental conditions, and their viability can be significantly affected by lack of water, hydromorphological changes and the quality of the GW itself. It is an important parameter for all GDEs, because GW provides nutrients and electron acceptors (e.g. sulphate), and usually creates specific physico-chemical conditions in GDEs (Kløve et al. 2011; Jakeman et al. (eds.) 2016). Therefore, modification of groundwater chemical balance and groundwater natural condition may result in irreversible changes in the entire ecosystem (Kløve et al. 2011).

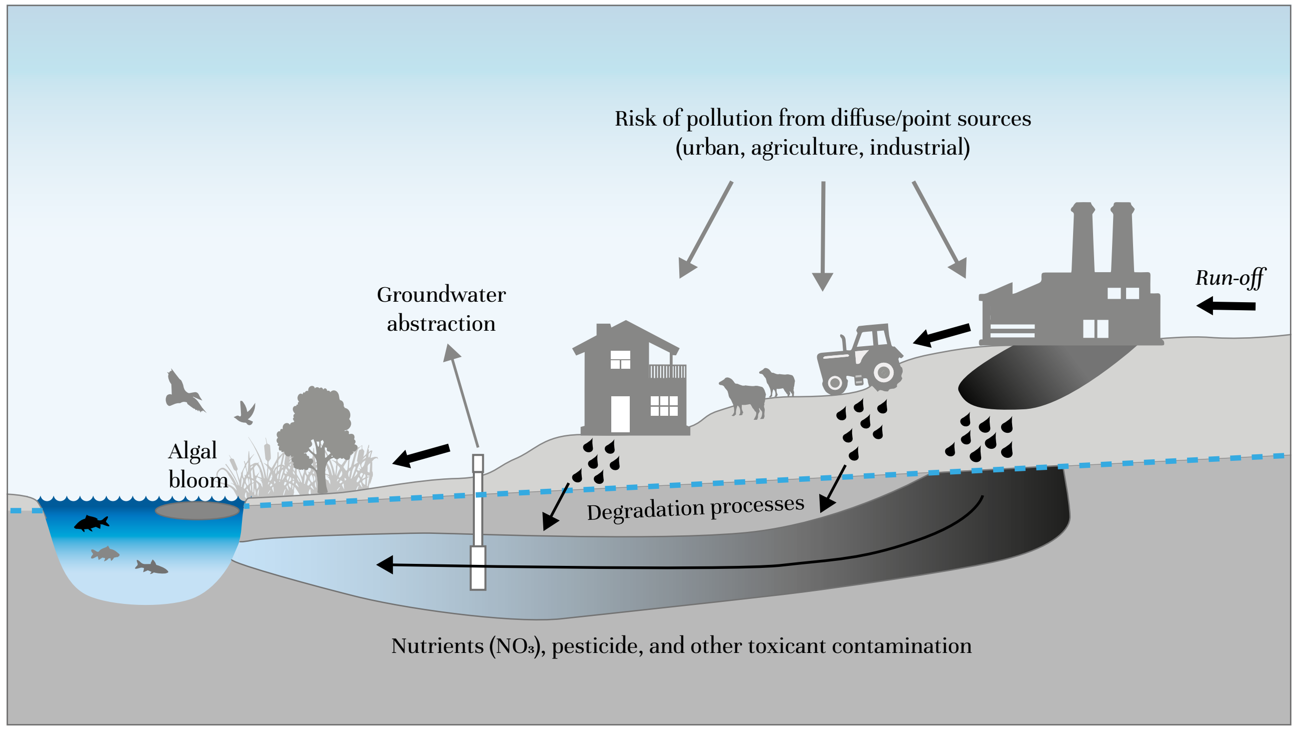


***Figure 5.8.** Major anthropogenic contamination sources affecting groundwater quality (modified from Quevauviller (ed.) (2008)).*

GW quality depends largely on the natural characteristics of its setting (geology and water residence time are the two fundamental factors controlling natural water chemistry). Changes in GW quality can occur for a variety of reasons, both natural and anthropogenic. Naturally, water quality can change due to changes in hydrogeological conditions, as well as changes in geochemical processes, such as the inflow of poor quality GW from other geological formations, seawater or surface water bodies.

GDEs, particularly those depending on unconfined GW, are highly vulnerable to contamination, and the majority of GW contamination results from human activity. Risk of deterioration caused by anthropogenic impacts depend predominantly on land use. Land development and numerous land use practices, such as conventional agriculture and urban developments, significantly determine contaminant sources and diversity of pollution (Figure 5.8.).

Agricultural areas are considered to be the main source of diffuse pollution. It reaches the soil and GW easily and persists well in an anaerobic environment until it emerges back to the surface from GDEs. Mainly nitrogen fertilizers and pesticides are used in conventional agriculture, therefore these parameters are considered to be the main indicators of GW pollution in agricultural areas.

GW pollution by nitrogen fertilizers is a global problem that is often causing deterioration of the quality of water resources. Inevitably, the increased use of nitrogen fertilizers has high potential to adversely affect GW quality and the health of GDEs and other wetlands. Nitrogen fertilizers can cause complex biogeochemical transformations to take place in GDEs (Kalvāns et al. 2021).

Nitrate leaching from agricultural lands to shallow GW has been reported in many regions around the world. Elevated nitrate levels in GW are marked in areas with intensive agricultural pressure and high vulnerability of GW. Severity of contamination is modified by other factors such as lithology, dissolved oxygen levels and land use. Discharge of nitrate enriched GW can alter nitrogen concentrations in the receiving water and hence increase the risk of eutrophication and algal blooms (Jakeman et al. (eds.) 2016).

Eutrophication or increased nutrient enrichment is the biggest threat to freshwater quality, which can be affected not only by run-off from agricultural land, but also by the high number of

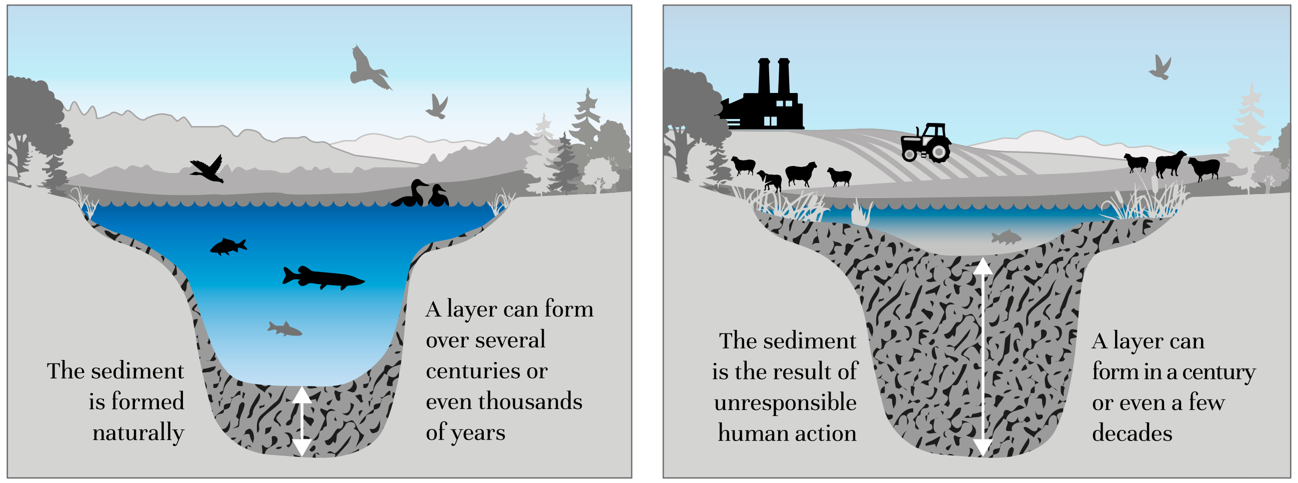


Figure 5.9. Overgrowth of water bodies as a result of eutrophication (modified from RMB Lakes Monitoring Program (2022)).

households not connected to centralized sewerage, treated wastewater and intensive forestry, and other activities (e.g. small gardens, dry toilets, compost piles, water protection zones, etc.). The declining quality of the freshwater ecosystem is indicated by the rapid overgrowth of the lake, for example with reeds, as well as the accumulation of sludge and turbid water, low species diversity, the occurrence of pollution and indicator species (free-floating water flowers). Over time, each lake transforms or overgrows and becomes a swamp, but man-made pollution accelerates this process many times over (see Figure 5.9.).

Pesticide contamination also can be a problem mainly for shallow GW. Using poor quality pesticides with low degradation rates, incorrect application of pesticides and inappropriate disposal methods can all lead to GW contamination, among which herbicides are the most frequently detected in GW (Jakeman et al. (eds.) 2016). Characterizing the dynamics of these products in the subsurface is complicated due to the differences in degradation and sorption rates (the two most important processes governing pesticide persistence), which are a function of individual pesticide compounds as well as sediment and aquifer matrix characteristics (Retike et al. 2020; Kalvāns et al. 2021). Porous aquifers are generally better at filtering pesticides from the GW, while karstic aquifers are more prone to long-term pesticide contamination issues due to rapid flow and low sediment reactivity (Burri et al. 2019).

As an example, in Latvia, elevated nitrate concentrations have been detected mainly in shallow Quaternary GW up to a maximum depth of five meters, but the nitrate content above the background values is found up to a maximum depth of 15 m. This is facilitated by a natural and intense denitrification process, which results in the conversion of nitrates to molecular nitrogen (N_2) in anoxic environment and return to the atmosphere (LEGMC 2021). The highest concentrations of nitrate have been found in the nitrate sensitive areas and adjacent areas where intensive farming takes place (Nitrātu ziņojums 2020; Retike et al. 2016). Groundwater (confined aquifers) located in areas of low natural protection (dominated by sandy sediments) or in areas with a karst process distribution (cracks in sediments) is at risk of diffuse agricultural pollution. This is also confirmed by studies (example of Kazu leja (Kalvāns et al. 2021) and the results of national spring monitoring: higher nitrate content occurs in springs flowing from fractured aquifers).

In Latvia over the last twenty years, agricultural activity has increased significantly. The use of nitrogen fertilizers in Latvia has doubled from 2005 to 2019: from 40.9 to 80.7 thousand tonnes of nitrogen equivalent (Kalvāns et al. 2021). In Estonia, the use of mineral fertilizers decreased significantly in the early 1990s due to changes in the economic model, reaching their lowest value (22 thousand tonnes per year) in 1996. Since Estonia's accession to the EU, the use of fertilizers has been on the rise, increasing by 60 thousand tonnes a year by 2018, but still significantly lower than in the Soviet period. To explore the interaction between nitrogen pressure and GDEs, a study was conducted in Latvia in an area of suspected high nitrate (NO_3^-) vulnerability due to its geological settings (Kalvāns et al. 2021). However, studies have shown that, in this case, the biota does not indicate deterioration of the ecosystem due to the increased content of nitrogen compounds in the water, as the agricultural pollution is partially absorbed in GDEs (Retike et al. 2020; Kalvāns et al. 2021). However, this result from one site cannot be generalized, as the GDEs are highly variable, bearing in mind that a limited range of bioindicators was used (see also Chapter 6.2).

In Estonia, the Nitrate vulnerable zone is located in the central part of the country, and it consists of northern Põltsamaa and southern Adavere area. It was designated by a 2003 regulation of the Government of the Republic (Vabariigi Valitsus 2003), which was renewed in 2019 (Vabariigi Valitsus 2019). Various scientific research, inventory, mapping, environmental assessment and analysis, monitoring of groundwater and water bodies were used to deliniate and characterize the nitrate-sensitive area. The main focus was on natural conditions, including to address the specificities of the water cycle (area with increased infiltration), as the realization of the pollution load depends the most on it. Agriculture is the most important cause of nitrate pollution in economic activities, which is why the agricultural pollution load has been mainly analyzed (Maa ja Vesi 2000).

In Estonia, a nitrate vulnerable zone is an area where agricultural activities have caused or may cause nitrate concentrations in groundwater above 50 mg/l or where surface water bodies are eutrophic or at risk of eutrophication due to agricultural activities. In Estonia, one of the most vulnerable groundwater is located in the nitrate vulnerable zone because geologically there are mostly limestones and karst areas with unprotected groundwater. About one fifth of the area is unprotected and this northern Pandivere part is also an important groundwater supply area for the whole country. On the other side, there are one of the most fertile soils in the country, which promotes agricultural activity in this area.

In the nitrate-sensitive area, the annual average nitrate concentration surpassed the threshold value of 50 mg/l in 14 percent of monitoring stations (wells, springs, karst) in 2020 (Keskkonnaagentuur et al. 2020). Total nitrate level in GW has grown by 60% in comparison to 2020, and has dropped in 30% of monitoring stations, according to long-term (2001–2020) averages. Because multiple GDEs in the region draw water from the same aquifer, the nitrogen concentration of the surface water rises, promoting eutrophication of surface water bodies. In the case of pesticides, the highest time for their usage was in the 1980s, when the number of active compounds used per year was between 1600 and 2200 tonnes (Keskkonna-agentuur et al. 2020). While use fell dramatically in the 1990s owing to a shift in the economic model (the collective farms of the time failed), it began to rise again in 2000, reaching 245 tonnes of preparations in 2004. Due to subsidies, the amount of used agricultural land in Estonia grew when the country joined the European Union in 2004. Simultaneously, agricultural activity has been better restricted since then, limiting the detrimental environmental impact of increased agricultural production.

GW quality may also be harmed by **urbanization**. Petrol stations, landfills, transportation infrastructure, chemical and petroleum industrial facilities, as well as historically polluted locations, are all possible pollution sources. The spectrum of pollution emitted into the environment can be different due to the huge quantity and diversity of polluting items in cities and other residential areas: petroleum products, chemicals, organic waste, plastics, and numerous inorganic compounds. The presence of anthropogenic organic pollutants in the environment, such as medications and personal care items, is causing significant worry across the world (Burri et al. 2019; Lapworth et al. 2019). It is important to mention that urban pollution is mostly detected in shallow GW, with deeper aquifers only appearing on a few occasions.

As a result, dangerous substances may directly poison GDEs. The effects of these risks are most likely to be felt in aquatic ecosystems, including aquifers, wetlands, and base flow-dependent streams. Nutrients, pesticides, and other toxicants contaminating alluvial aquifers can harm dependent ecosystems in base-flow streams, particularly aquatic habitats. Upstream land uses in karst systems can have an impact on surface and GW quality, especially if the catchment comprises agricultural

or timberland. Water quality has an impact on karst systems' natural erosion and sedimentation processes, as well as their aquatic habitats (Serov et al. 2012). The negative consequences on biotic ecosystems are not well understood due to their complexity. Hazardous compounds have the potential to infiltrate the food chain, causing direct and indirect harm to human and animal health, as well as long-term consequences that are difficult to detect.

GW level alterations may affect water quality by causing processes such as seawater intrusion and increased acidity in the soil and deeper rocks. Such changes in GW levels, which also affects GW quality, are usually caused by anthropogenic processes such as pumping, amelioration, land clearing, excavation or mining. GW level lowering due to pumping may also result in the seawater intrusion in freshwater aquifers or the leakage of brackish water in these freshwater aquifers. Similarly, GW levels are also affected by climate change e.g. increasing periods of drought. Accordingly, such a decrease in GW levels can directly affect the functioning of ecosystems, for example, leading to the drying of wetlands. Also, rising levels can affect the water quality. Rising GW levels or changes in the flow direction can also lead to poor quality- or contaminated GW flow into wetlands or streams. In any case, such hydrodynamic processes affect local hydrogeochemical conditions, contributing to soil salinisation and the accumulation of metals closer to the surface, thus posing a potential threat to the associated ecosystems.

6. Restoration and mitigation of unfavourable effects

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6.1. Basics of restoration planning

The conservation of natural ecosystems, including GDEs, is always best ensured through non-intervention and prevention of potential unfavourable impacts. The potential or existing unfavourable impacts can be prevented by applying special protective regimes or other strategies at the catchment level.

Targeted actions aimed at the recovery of deteriorated natural environments may include various strategies and various aims. The actions may be classified as restoration, creation, rehabilitation, reclamation, mitigation and maintenance. This guideline book focuses on restoration and mitigation as the main strategies for GDE conservation and improving the conditions of natural or near-natural GW-related ecosystems. Often, maintenance of the post-restoration result is needed to secure the long-term effect.

Often, the aim is to return a degraded ecosystem to a former natural condition that is called restoration. On a broader scale, the term restoration means the re-establishment of pre-disturbance ecosystem functions and related physical, chemical, and biological characteristics. Restoration is a holistic process that cannot be achieved through an isolated manipulation of individual elements (National Research Council 1992).

It is essential to re-establish the overall functionality of the ecosystem that requires reinstating conditions for the recovery of both abiotic and biotic components. Depending on the types of impacts, that requires one or more of the following actions: reconstruction of antecedent physical conditions, chemical adjustment of the soil and water, and biological manipulation, including the reintroduction of native biota, most commonly vegetation containing typical species (National Research Council 1992). The long-term maintenance of biodiversity depends on the survival of appropriate typical biotic communities which may require further natural disturbance, e.g. grazing by herbivores (National Research Council 1992). It requires understanding the ecosystem functionality and mutual relation among its components.

For example, in spring ecosystems, a functional ecosystem requires a constant flow of water without sharp fluctuations in the amount of water and without significantly altered chemical and physical properties of the water. A considerable proportion of species related to this ecosystem require a stable microclimate. Microclimate stability is related both to constant water flow and the absence of rapid changes in the surrounding forest that may cause unfavourable effects on vegetation composition.

In certain situations, one may put a landscape to a new or altered use to serve a particular human purpose (reclamation or rehabilitation) (National Research Council 1992), for example, creation of water bodies for recreative purposes or establishment of agricultural land in strongly modified areas. Reclamation and rehabilitation are terms that are usually applied in cases of severely altered environments, such as post-mining areas. It is likely that the original GDE if damaged by a mineral or peat extraction may not recover, as the original abiotic and biotic environments are fully destroyed. However, in the long term, if suitable conditions are created by reclamation measures, a new ecosystem may be established (Figure 6.1.). For example, in a former mineral extraction area where a new environment is created, a new fen may develop. However, it may take a very long time, and the outcome is highly unpredictable.

Figure 6.1. *Spontaneous development of alkaline fen with small sedges and the great fen sedge Cladium mariscus on a shore of a former dolomite quarry (Latvia, near Sloka, 2021). The photo is taken ca. 40 years after the post-mining area is filled up with water. Photo: A. Priede.*

In cases when the unfavourable impacts cannot be excluded, mitigation of the effects may be an acceptable strategy. That may allow promoting the existence of the ecosystem, though not fully in the desirable condition. That may apply to cases when the impacts are difficult or impossible to prevent at the local level, such as climate change, transboundary pollution, etc. Then, mitigation may be used as the best alternative to achieve the aims partially instead of losing a valuable ecosystem.

The key issue in restoring and maintaining GDEs is always the GW recharge area, not the point or area where the GW is flowing out.

Although the GDEs can be classified using one or another classification scheme (vegetation, spring types, etc.), still they are extremely diverse. **It is impossible to have fixed restoration schemes for all disturbed GDEs situations considering the variety of site conditions.** General principles in planning and implementation can be used; however, individual solutions for the particular sites must always be sought.

Setting restoration aims is essential before beginning any GDE restoration or mitigation measures. Prioritization may help to focus on the actions that may best help to restore the ecosystem.

The restoration should be aimed at both functionality and biodiversity of the GDEs, as they are mutually related. Often, the focus strongly depends on who is the initiator of the restoration or mitigation actions. The primary interests may be groundwater management, biodiversity conservation or spring-related cultural heritage that each have their own motivations and points of view. In optimal planning, the views should be integrated as much as possible by respecting all aspects and involving the stakeholders and experts from other fields where relevant.

Restoration intentions aiming at restoring the biodiversity of GDTEs without respecting and understanding the water cycle may fail, and vice versa. For example, a spring-fed fen may suffer from GW table dropping, followed by increased tree encroachment and loss of highly specialised light-demanding and GW dependent plant and invertebrate species. In such a case, primarily aimed at biodiversity conservation, the GW management and restoration measures to re-establish an optimal GW table are of primary importance. Only then, as the next step or parallel to the GW table management, removal of woody vegetation and similar biotechnical measures can be effective. Solely removal of trees and mowing would not help to achieve the desired condition.

The most common potential problems to be solved in GDE restoration in the focus region (Latvia, Estonia) are:

1. Human activities affecting the GW quantity or quality that deteriorate GDEs (see Chapter 5):

- Extraction of peat, tufa or other spring sediments (direct deterioration of both abiotic environment and biotic communities);
- Modifications of GW table (excessive GW extraction, mining) that induce ecosystem response including biotic changes;
- Land use change, e.g. creation of water bodies, arable lands;
- Chemical pollution, including nutrient leaching in GW and surface runoff, both point sources and diffuse sources, e.g. agriculture, forestry operations.

2. Human-induced changes in GDEs that are not related to modification of GW or quality but may result in loss of species or typical biotic communities:

- Invasive alien species;
- Increased trampling (recreation, grazing);
- Various artificial constructions that alter the natural environment in springs and deteriorate the natural vegetation (tubes, well lining rings, stone walls, garbage, etc.)
- Beaver-caused modifications in the water table, beaver ponds.

The questions that should be answered during the preparation period for GDEs (including GDTEs) restoration or/and mitigation (modified from Stevens et al. 2016a):

1. What is the problem (problems in GW quantity, quality, other)?
2. Can the problem be fixed?
3. Who is the landowner or manager?
4. What are the funding options in both the short term and long-term?
5. What is the urgency of restoration or mitigation?
6. What are the desired future conditions?
7. What kind of actions are needed?
8. What is the timeline?
9. What are the potential obstacles (e.g. legal constraints, land ownership, technical solutions)?
10. Who will be responsible for restoration?
11. Who will be responsible for maintenance of the results and further management if considered necessary?
12. How will the results be monitored (methods)?
13. Who will finance monitoring, who will carry out the monitoring and analyse the data?
14. Who will be responsible for adjusting the management if considered necessary according to the monitoring results?

There may be additional aspects to be taken into account such as cultural heritage, recreational and tourism value of the sites. Many springs are known as ancient scarification sites and popular visitor destinations including drinking water collection. Preservation of this value must be respected when planning restoration measures unless this creates significant conflicts with the conservation of GDE.

The restoration and mitigation always require the engagement of the stakeholders, respecting the national regulatory enactments and planning documents. Depending on the site, its status and the scale of restoration measures, the preparatory stage may include various inventories and assessments including fulfilling specific requirements by the national or local authorities to assess the potential environmental impact. Large scale restoration projects within Natura 2000 areas or areas neighbouring to them may have an obligation to carry out the environmental impact assessment procedure. All restoration projects of this type usually must include identification of potential conflicts and their solutions and careful assessment of potential impacts to the neighbouring human settlements, water sources, agricultural and forest lands, biodiversity, cultural heritage, and others.

The planning of GDE restoration measures should include collected knowledge about the nature of disturbances in the GW recharge area on the surface and the flow paths feeding the GDEs, since disturbances can be caused by land use types and related intensity that occur far away from the immediate seep area. Therefore, delineating and understanding the temporal and spatial variability of the hydrological catchment area of the seep can be an important first step towards protecting the target GDE (see Chapter 2.3.2).

Studies of long-term precipitation and temperature changes at Estonian peatland sites have demonstrated the importance of weather changes knowledge for at least the last 50 years in planning the restoration or/and protection of any types of wetlands (Lode, pers. comm.). Therefore, collecting of weather data and proper hydrological analyses, especially of (summer) minimum or vegetation period conditions, are recommended for an area extension that would cover both the GDEs recharge and discharge areas.

6.2. Example for restoration and mitigation actions in spring ecosystems

An increased depth-to-GW is an important threat to the spring ecosystems. Increased depth-to-GW could be observed as a result of GW abstraction for human use and increased discharge (reduced recharge) (Stacey et al. 2011). A similar effect may be caused by a long-term decline in annual rainfall across the region that results from climate change.

The drop of GW table causes a response in the typical vegetation – most commonly, loss of species that depend on constant water supply and stable microclimate. In the Baltic region, often valuable spring ecosystems lie in forested landscapes, thus being highly dependent on shaded or half-shaded conditions, although there are also open spring fens with specific communities. Many typical species (e.g. bryophytes and invertebrates) occurring in spring-fed forest patches and spring fens are attributed with low dispersal ability (Ilmonen et al. 2013) being vulnerable to sharp changes in GW table and microclimate. The best way to ensure the conservation of such ecosystems is a non-intervention regime, i.e. prevention of any potential unfavourable changes in GW quantity and quality in the catchment. Similarly, any damage to the GW quality should be ensured by the catchment approach by preventing the use of pesticides and fertilisers in a way that results in the leaching of the pollutants into GW.

Since many spring-related species are known as shade-loving, strongly associated with stable microclimate, it is essential to ensure a non-intervention regime or adaptive management of forests in and around spring discharge areas. Forest clear-cuts around springs (Figure 6.2.) are not acceptable, as such forestry operations cause rapid change in biotic communities. Removal of trees induces desiccation and rapid turnover of vegetation, with a high risk to lose rare, threatened and highly specialized species.

Figure 6.2. *Spring pool in a forest clear-cut in Western Latvia, an example of mismanagement without respecting the presence of the spring. Photo: A. Priede.*

Reducing or stopping the use of fertilisers and pesticides is highly important in carbonate and karst areas where the chemicals may easily reach the aquifer. A thin layer of Quaternary sediments covering the karst aquifer provides conditions favourable to nitrate conservation in GW due to the well-aerated state. The vadose zone within the dolomite aquifer may cover several kilometres from the spring discharges. Such conditions may be found in numerous river valleys in Latvia (Kalvāns et al. 2021), e.g. Gauja and Abava Valleys, as well as in the Pandivere Upland in Northern Estonia, composed of carbonate rocks and surrounded by extensive wetlands and at the Estonian Klint (Kalvāns et al. 2021).

The biotic effects of fertilisers and pesticides on the spring ecosystem communities are still not well understood. For example, in Kazu leja area in Gauja National Park, Latvia, no direct evidence of the elevated nutrient impact was found using plant and ground-dwelling mollusc communities, though increased nitrogen concentrations were found in the spring water (Retike et al. 2020; Kalvāns et al. 2021). The reasons may be the high fen ecosystem resilience and a time lag between the impact and evidence of degradation. Thus, the increased agricultural pollution using a limited range of bioindicators may seem “harmless”.

There is still insufficient experience on restoring the original or desirable GW table and GW dominance in spring ecosystems damaged by GW lowering, significant reduction of GW supply and drainage. In general, the GW-dominated conditions should be restored. The GW table should be restored whenever possible by damming or blocking the ditches and preventing the impacts caused by GW drop at the catchment level. Similarly, the surface water influence (inflow from ditches) should be diminished or prevented. Reduction of the surrounding GW pumping rates can help to improve the overall ecosystem health in the watershed (Stacey et al. 2011).

As shown by a broad-scale study in boreal spring ecosystems in Finland, the restoration of springs affected by forest drainage brings a positive effect on spring hydrology deteriorated by forest drainage. The effect may be detectable already few years after restoration (Lehosmaa et al. 2017).

Some other measures may be beneficial for the spring ecosystem, such as rehabilitation of spring outflow and restoration of discharge channel and floodplain morphology (Stacey et al. 2011). That may include removal of dams, berms, pits and other artificial constructions (usually in severely altered sites), as well as the removal of beaver dams that may damage the spring fens. The beavers may create ponds in sites that were originally open or forested fens; the permanent inundations may cause disruption of spring outflows (Figure 6.3.).

Figure 6.3. *An inundated forest with permanently submerged spring pools; the inundation is caused by beaver dams on the spring brook downstream. Surroundings of Renda, Western Latvia, 2021. Photo: A. Priede.*

The spring outflows may be equipped with pipes, well lining rings and other artificial water abstraction structures that may disturb the natural processes and development of typical spring vegetation. In such cases, if the constructions and the spring equipment cannot co-exist, such constructions should be removed.

The geomorphic restoration includes channel and slope stabilization to reduce erosion (Figure 6.4.) by revegetation (only native, site-typical vegetation) – planting vegetation plugs, pole planting native phreatophytes, and covering bare soil with netting, straw, or wire fencing (Stacey et al. 2011).

Several other restoration methods may be used to improve the condition of the spring ecosystems:

- Removal of trees and shrubs to restore open fens (see also Chapter 3.3);
- Mowing;

- Removal of non-native invasive and native highly competitive species (e.g. reed);
- Revegetation and reintroduction of native species.


Figure 6.4. *Sloping spring-fed fen with a roadside ditch at the slope foot. The soil instability and spring activity cause erosion. Kandava, Western Latvia, 2021. Photo: A. Priede.*

Removal of woody vegetation and mowing is sometimes practised in the spring fens (Figure 6.5.). That helps to improve the conditions for light-demanding plant species and invertebrate communities that may suffer from tree encroachment and subsequent change in vegetation (Priede (ed.) 2017). That may be combined with mowing of herbaceous vegetation that was practised in the past for hay collecting purposes. Mowing and biomass removal must be done with care and only manually using trimmers or scythes, as it causes trampling of the wet ground with fragile plant communities and spring deposits, thus, most probably, may not be done every year but can be applied biannually.

Grazing as an alternative to mowing is not considered a suitable method in spring fens, as it causes trampling and damage to the ground (Priede (ed.) 2017).


Figure 6.5. *Spring fen management site in Gauja National Park, Latvia, in the pre-restoration condition (2013) and after (2021). In 2014, the trees and shrubs were removed from the semi-open spring fen, excluding the forested part. Additionally, mowing with hay removal was established with an aim to reduce the cover of reed for a rare plant species. The management has promoted the establishment of several light-demanding fen species and increased their number, e.g. Dactylorhiza russowii (Figure 3.6.) and Primula farinosa. Photo: A. Priede*

Eradication of invasive non-native species must be always done with care. The use of herbicides is not acceptable, because it may cause GW pollution and loss of typical native species. Thus, only manual removal methods may be used, such as digging out and repeated mowing, pulling out with roots, covering with black plastic sheeting, and similar methods. The eradication must be always done in the plant development stage when they still do not have developed seeds. All viable parts (roots, stems, flowers) must be always removed from the site not to cause secondary establishment (Priede (ed.) 2021).

Damage to spring ecosystems may be caused also to **trampling** by visitors in popular tourist destinations that may be diminished or prevented by appropriate, well-planned infrastructure elements such as boards, platforms, railing, etc. and informative signs.

A heavy trampling impact can be caused by livestock in grazing areas. The best solution is to exclude such areas from the grazing grounds by installing fencing.

Recovery of desirable vegetation may occur spontaneously. In complicated cases, **revegetation and reintroduction of native plant** species could be achieved through seeding and planting transplants. That can be aimed at native spring specific vegetation as such, or selected target species, e.g. rare, threatened species (Stacey et al. 2011).

As suggested by observations in the field, also by the study by Lehosmaa et al. (2017), there are some **NOT TO DO measures**. For example, the spring pools shall not be dug deeper or cleared of mosses and deadwood unless there is an excessive amount of fallen trees that require special considerations. Sometimes that is done to improve the visual appearance of the springs. The conditions should be kept as natural as possible because the deadwood serves as a substrate for numerous specialist species, and the mosses are the target vegetation (Priede (ed.) 2017).

Detailed restoration and management guidelines for Annex I habitats including the habitat types considered GW-dependent (see Chapter 3.1)[8] were developed for Latvia earlier. They include the decision-making schemes, an overview of the threats and practical solutions. The guidelines or at least some ideas may be used not only in Latvia but also in Estonia and neighbouring countries in the Baltic region: Priede (ed.) 2017.

6.3. Basics of monitoring the GDE restoration success

Monitoring is an essential component of ecosystem restoration. The monitoring programmes, methods and their complexity may vary depending on the capacity of the initiator or responsible authorities and the funding available. Here, only the basic principles are explained.

To cover both the GW and biodiversity aspects, the monitoring should include at least the minimum set of indicators that are well linked with the restoration objectives. It means that the

[8] Mineral-rich springs and springfens (7160), 7220* Petrifying springs with tufa formation (7220*), Calcareous fens with *Cladium mariscus* and species of the *Caricion davallianae* (7210*), Alkaline fens (7230).

basic questions are: what did we want to achieve? Did we achieve it (fully or to what extent)? Thus, monitoring should be an integral part of restoration planning and implementation and continue after the measures. Monitoring may highlight a need for improvements in the restoration performance or after-restoration maintenance – in that case, it should be fixed as soon as possible.

Generally, the hydrogeological set of methods may follow the guideline given in Chapter 8, bearing in mind that there are no unique monitoring or research schemes for all cases and there are no best hydrogeological investigation methods that fit all cases. To follow the recovery of biota, perhaps the most common way is vegetation monitoring (permanent sample plots, transects or other methods). However, other specific groups of organisms may be chosen, as they may provide better or more precise results than vegetation monitoring, for example, algae or invertebrate communities in spring pools.

The ecosystem recovery is rarely a linear process. In the long term, especially if the recovery takes several decades or centuries, as in areas damaged by peat cutting, the recovery may be interrupted by disturbances, e.g. beaver activity or fires in extreme drought periods.

In the short term, success is indicated by a stable (several years) positive trend, e.g. stabilization of GW table and establishment of habitat specialist species. The restoration initiator shall not expect that a heavily disturbed ecosystem will fully recover in a few years, even though optimal conditions are re-established or created. The result after some years may not qualify as one or another protected habitat type, as shown by the case study in Kazu leja, Latvia (Retiķe et al. 2020; Kalvāns et al. 2021) with a heavily disturbed spring-fed peatland. Meanwhile, the GDE may provide significant ecosystem services, such as excess nitrogen uptake from the catchment and carbon accumulation through peat accumulation.

7. Ecosystem services provided by groundwater dependent ecosystems

Līga Strazdiņa (Nature Conservation Agency, Latvia)

7.1. Insight

Although the role of ecosystem services (ESS) provided by any natural ecosystem has been discussed for almost the last three decades, the clarity of the topic is still quite vague, and this concept is rarely implemented in nature conservation practices. ESS are a variety of benefits that people derive free of charge from natural or altered ecosystems and use them to guarantee and increase well-being. ESS have a significant impact on landowners' behavior and willingness to pay for land, diversify and expand income streams (Caparrós et al. 2013). A service that is produced but not used is not a service, and it refers more to potential ecosystem services (Cameron 2018). The most important ESS, such as fish production and recreation, as well as the key ESS, such as water purification, water retention and climate regulation, are related to aquatic ecosystems (rivers, lakes, GW, coastal waters, seas) (Grizzetti et al. 2016). However, GDEs also supports the provision of important ESS, including provisioning services such as drinking water or agricultural water; regulating services, such as flood and drought and GW recharge and discharge regulation; supporting services such as soil formation and nutrient cycling; and cultural services such as recreational, aesthetic value and other intangible benefits (Table 7.1.).

It is concluded that both the market and non-market economic benefits and the total economic value of wetlands, including GDEs, tend to be higher compared to converted land (Millennium Ecosystem Assessment 2005). However, determining the exact value and scope of services remains a challenge. Underestimating the range of ESS that an area can provide is likely to be used inappropriately, which in turn can lead to even lower economic benefits (Barbedo et al. 2014). Yet, the analysis of ESS values can be a strong argument for improved management and greater land conservation (Caparrós et al. 2013). Society is intuitively aware that both the market price and the ecological value of water, whether drinkable or not, increase as the population grows and prosperity increases (Heal 2000). If the amount of available water were to fall, its value would increase greatly, and yet more would be paid just to get enough for society's needs. But there are still ESS whose market value is almost or completely impossible to calculate, and have been underestimated, including water ecosystems, such as the value of a beautiful lake view. By mistake, the value of an ecosystem is assumed to be the cost of replacing all the services that could be lost if the ecosystem in question were removed. However, the value of natural ecosystems and the services they provide is usually valued as a small change (positive or negative) in their availability.

Water demand will increase in the future, precipitation will decrease, temperatures will rise and longer periods of drought are expected. Therefore, water scarcity is a major concern and, like many environmental problems, it is closely linked to social, economic and political problems (Garau et al. 2020).

Table 7.1. Ecosystem services provided by or derived from GDEs (modified after Millennium Ecosystem Assessment (2005) and Griebler & Avramov (2015)). ESS directly related to GDEs are marked with asterisk (*).

Services	Comments and examples
Provisioning	
Food	production of fish, wild game, fruits
Fresh water*	storage and retention of water for domestic, industrial, and agricultural use
Fiber and fuel	production of logs, fuelwood, peat, fodder
Biochemical	extraction of medicines and other materials from biota
Genetic materials*	genes for resistance to plant pathogens, ornamental species
Energy*	
Regulating	
Climate regulation	source of and sink for greenhouse gases; influence local and regional temperature, precipitation, and other climatic processes
Water regulation (hydrological flows)*	groundwater recharge/discharge
Water purification and waste treatment*	retention, recovery, and removal of excess nutrients and other pollutants
Erosion regulation*	retention of soils and sediments
Natural hazard regulation	flood control, storm protection
Pollination	habitat for pollinators
Disease control*	
Drought attenuation*	
Alleviation of climate-warming*	via provision of refugia for cold-stenotherms
Cultural	
Spiritual and inspirational*	source of inspiration; many religions attach spiritual and religious values to aspects of wetland ecosystems, including GDEs
Recreational	opportunities for recreational activities
Aesthetic*	many people find beauty or aesthetic value in aspects of wetland ecosystems, including GDEs
Educational	opportunities for formal and informal education and training
Bioindication*	
Supporting	
Soil formation	sediment retention and accumulation of organic matter
Nutrient cycling*	storage, recycling, processing, and acquisition of nutrients
Biodiversity*	
Support of groundwater-related food webs*	
Provision of a habitat*	for species that cannot survive anywhere else

7.2. Regulation of water cycle

One of the main ESS maintained by GW is the regulation of water flow itself to prevent flooding (Barbedo et al. 2014; Geneletti et al. 2020), especially in the context of urban sprawl. To reduce the risk of floods, as set out in the WFD, the best solution is "to work with nature rather than against it". For the current urban landscape, this involves reverse development and restoring altered river systems, which could lead to a conflict between progress and conservation. The other approach is to find innovative drainage-related concepts in the urban landscape to reduce the negative impact on the hydrological cycle, infiltration and water storage processes. In both cases, a clear dialogue is needed between conservationists and policy makers.

7.3. Provision of drinking water and other resources

One of the most commonly used provision services by groundwater is drinking water. GW provides 75% of Europe's drinking water and is used by a third of the population globally (Griebler & Avramov 2015). Rising demands from population growth and urban sprawl are having a negative impact on water quality and show declining trends in aquatic ecosystem health, so we as a society could pay even more attention to source water protection, water supply infrastructure and treatment systems (Derrington 2011). It is important to note that only 20% of the total water intake is still consumed by households and cities, while 69% is diverted to agriculture (Lalonde 2019). In areas with limited water availability, land management is essential to improve water quality, erosion control structures, and the management of streams and ponds for wildlife (Caparrós et al. 2013).

In most freshwater ecosystems, including artificial ecosystems, the primary productivity service supports provision services such as fiber (wetland reeds, papyrus, riparian ecosystem trees) and food (fish and crustaceans, aquatic birds, edible unicellular algae). Almost all species cultivated in artificial freshwater ecosystems have wild ancestors and close relatives in 'natural' freshwater ecosystems (Safriel 2011).

7.4. Pollution filters

GDEs have a huge potential to naturally deplete and degrade a huge variety of pollutants released underground in an aerobic, anaerobic and syntrophic manner. The subsurface systems are effective in preventing, inactivating and destroying pathogens, such as water carrying pathogenic viruses (for example, emerging from sewage treatment discharge). In addition, groundwater invertebrates can play an important role in the removal of organic contaminants from aquifers (Griebler & Avramov 2015). The relatively slow movement of water in GDEs like wetlands allows suspended matter to settle and provides time for the mineralization of organic compounds and the biodegradation of toxic chemicals (Safriel 2011). Water in lakes can last from hours (for example, in some water dams) to several years. This time period provides an opportunity to reduce the amount of incoming pollutants, especially nutrients, through a variety of physical, chemical and biological processes. Thus, nutrients can be trapped in the bottom sediments of the lakes or released into the atmosphere, to some extent purifying the lake water flowing downstream. This self-cleaning process is the principle of using some artificial lakes to protect sensitive downstream ecosystems from potentially harmful levels of nutrients, sediments or other pollutants (such as heavy metals and acids from mines) (Schallenberg et al. 2013).

7.5. Peat and carbon accumulation

The aquifers are continuously fuelled by a significant amount of organic carbon, as a result of which GDEs significantly promote carbon circulation and thus water purification (Griebler & Avramov 2015). As a result of metabolic processes, GW and especially the springs become saturated with CO_2 after GW filtration. In turn, after CO_2 fixation and chemolithoautotrophic primary production, GDEs can also act as a source of carbon, which is one of the most important processes on the Earth's surface.

Due to carbon storage, minerotrophic peatlands, which are GDEs, are also important as peat-forming areas. Peat is stored with water saturation. As long as the peatlands are wet, the carbon remains practically eternal. Carbon storage grows slowly but steadily with the addition of fresh plant material that is converted into peat. In this way, thick layers of peat are deposited over time. Under the humid conditions required to form peat, part of the plant material decomposes anaerobically, resulting in the release of methane (CH_4) into the atmosphere. Methane is 25 times more potent than CO_2 as a greenhouse gas. If the peat bog is drained, the peat is no longer saturated with water and oxygen enters the peat. Under the now established oxic conditions, CH_4 emissions stop, but the aerobic decomposition of peat releases CO_2 and often nitrous oxide (N_2O), a GHG that is 298 times stronger than CO_2. These emissions continue as long as the peatland is drained, which usually takes decades to centuries (Joosten et al. 2015).

Lakes also mitigate the effects of climate change in two main ways: carbon sequestration and hydrological buffering. Globally, inland waters are estimated to sequester about 20% of land-borne carbon, reducing carbon losses from inland waters by about one third. Lakes can therefore play an important role in ecosystem services to reduce the effects of global warming. However, lake sediments can also produce significant amounts of methane. It is highly oxidized to carbon dioxide in lakes when it forms in deep hypolimnia. When methane escapes in the form of bubbles or from shallow sediments, it is mostly lost as methane to the atmosphere (Schallenberg et al. 2013).

7.6. Unique biodiversity

GW and aquifers provide habitats for high diversity of microbial communities and metazoan fauna (stygofauna), including many living fossils and endemic species (Griebler & Avramov 2015). The biodiversity provided by lakes (not necessarily dependent only on GW) can be measured in terms of genetic, species, population, functional group and food web diversity (Schallenberg et al. 2013).

Not only does the water ensure the development of a specific flora, but in return – the plants contribute to the preservation of water quality, not necessarily only GW, but water in general. The vegetation cover of the soil, made up of a myriad of plant species with very different sizes, structures and functions, regulates one major flow from the biosphere to the atmosphere through the ecosystem function of evaporation. Part of this flow is driven by the physiological process of plant transpiration, while evaporation from the soil surface is regulated by the degree of protection against sunlight entering the plant cover (Safriel 2011).

7.7. Interconnection with other ecosystems (landscape functioning)

Many other ecosystems are directly and indirectly regulated by GDEs, but they also affect GDEs retrospectively, both positively and symbiotically and more often negatively. For example, increasing the use of fertilizers on agricultural land to increase crop production affects water quality. Land use change in forest land after the removal of ESS, such as timber or charcoal, can also often increase the input of water pollutants or remove the natural water filtration service provided by the GDTE (Millennium Ecosystem Assessment 2005). Water supply systems are often directly linked to the presence of forests, especially dense canopy cover, as they prevent rainwater (green water) runoff and also absorb drinking water reservoirs (Caparrós et al. 2013).

7.8. Recreational and scenic value

Most economic activities, including tourism, are highly dependent on water and have a significant impact on its use and consumption. Demand peaks during the summer months, when the aquatic services are most vulnerable. Although the coastal wetlands are one of the most degraded ecosystems in the world, they are an important tourist attraction, offering a variety of recreational activities such as walking, bird watching and fishing (Garau et al. 2020). The recreational services provided by the lakes are also many and include a variety of activities such as boating, fishing, swimming, hiking, kayaking and waterfowl hunting (Schallenberg et al. 2013).

7.9. Cultural heritage

Natural resources like space, scenery, water, skies, have profound and diverse meanings to people. Such ESS have unquantifiable cultural and spiritual meanings, and our knowledge is limited to value their impact on human well-being (Caparrós et al. 2013). Historically sacred springs and wells have been associated as sources of spiritual knowledge and wisdom due to magical properties of water to heal and confer vitality (Griebler & Avramov 2015).

8. Hydrogeological research methods

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GDEs can be supported by GWBs temporarily and spatially in various ways and in different proportions. Generally, prolonged monitoring periods and extensive study of hydrogeological conditions of particular areas are needed to fully clarify the connection between the terrestrial or aquatic ecosystem and the subsurface hydrological systems. Sometimes it takes even years to observe ecosystem partial dependence on GW, for instance, during drought or high-level events when the dependence is more obvious. To ease the procedure, monitoring design and hydrogeological research methods should be chosen based on either type of GDE, habitat (EC 2013), or GW temporal supply regime to GDEs (if determined) or supporting aquifer attributes (Kreamer et al. 2014).

The persistence of GDEs relies on suitable GW attributes. Identifying these attributes is essential, as this can help establish GW management targets and monitoring strategies (Kreamer et al. 2014). The following GW attributes are important for GDEs (Clifton & Evans 2001):

1. GW level for unconfined aquifers. Depth to GW, with particular reference to the distance between the capillary fringe above the water table and plant roots, directly determines GW availability to vegetation.

2. GW pressure – hydraulic head and its expression in GW discharge, for confined aquifers;

3. GW flux – flow rate and volume of GW supply; flow direction

4. GW quality – the concentrations of nutrients and pollutants.

There is no unique algorithm of activities and best hydrogeological investigation methods to choose to fit all cases, however, following subsequent steps, screening approach or step-wise approach will give a better understanding of which activity is needed and when it should be planned to avoid repetitiveness, pointless work and unnecessary expenses. Consulting with those with relevant expertise (e.g. hydrologists, hydrogeologists, environmental modellers, agronomists, ecologists) is advisable at every stage. Stakeholder engagement is also a valuable process in mutually educating, reducing conflict and building trust among researchers, decision makers and other stakeholders (Eamus 2016).

8.1. Define problems, key issues and objectives

It is crucial to define the right questions to be answered using hydrogeological methods before initiation of the study. Probably it is already known what tasks should be done or what must be explored about hydrogeological properties of the study area. It is not always needed to investigate all aquifers on site and setting too large study areas. Setting too many objectives can lead to unneeded expenses and excess work. In the case of GDE connection with GWB, the primary objective is to characterise the GDE. In cases when potential threats to the GDE are already clarified, it is easier to pose the right questions, however, it is useful to investigate possible future threats or estimate the impact of climate or land use change, additional GW abstraction from the GWB or mining impact on particular GDE.

According to the WFD, one of the objectives is to achieve good quantitative and good chemical status of GW: "The chemical composition of the groundwater body is such that the concentrations of pollutants: [..] *are not such as would result in failure to achieve the environmental objectives specified under Article 4 nor in any significant damage to terrestrial ecosystems which depend directly on the groundwater body…* ". This does not apply to naturally occurring increased substances, but only to anthropogenic impact (EU Water Framework Directive, 2010).

Accordingly, the level of GW is not subject to anthropogenic alterations such as would result in: "*… failure to achieve the environmental objectives specified under Article 4 for associated surface waters; - any significant damage to terrestrial ecosystems which depend directly on the groundwater body…*" (EU Water Framework Directive, 2010).

A technical report on GDTEs (Schutten et al. 2011) suggested that step-wise or source-pathway-receptor approach could be used to conceptualise the GW flows and anthropogenic pressures.

The first step when using the screening approach, is to determine the likely dependency of the GDE on GW (ecological features described in Chapter 3.1). When a spring or seepage is clearly visible direct dependency on GWB is more obvious, however, it is not always the case, therefore better understanding of the hydrogeology of the area is needed. The next step is to identify GW levels in the area and their possible discharge near potential GDE. Simple conceptual models are useful to perceive geological structure and therefore estimate possible connections between GWBs and GDEs (Chapters 8.2 and 8.3). Further steps include development of the model with additional features, updated information, and monitoring results. Building and application of conceptual models for GW are described in detail in the Common Implementation Strategy Guidance No. 26 (EC 2011).

8.2. Preliminary assessment

To build the conceptual model or simply estimate the study area of the GDE basic geological and hydrogeological information is needed. In Latvia and Estonia, hydrogeological information can be gathered from governmental and research institutions (in Latvia – Latvian Environment, Geology and Meteorology Centre, https://videscentrs.lvgmc.lv; in Estonia – Geological Survey of Estonia, www.egt.ee).

There are various available data types that can be used:

- **Geological maps.** The coverage of regional scale (scale 1:500 000 or 1:200 000, in best cases resolution can be even 1:50 000) completely covers Estonia and Latvia, although the resolution will be insufficient for local studies, geological maps will be useful to establish spatial limits of conceptual models both horizontal and vertical.

- **Hydrogeological maps**. Similarly, hydrogeological maps come in various scales, depending on the site they might be available in better resolution (mines, GW abstraction sites, etc.). GWB maps are useful to determine the GW levels, percolation areas, recharge and discharge areas, as well as to determine the main GW flow directions. Some hydrogeological maps contain information on the baseline GW chemical composition.

- **GW level monitoring and abstraction data.** Every country has a GW monitoring programme. GW monitoring sites, i.e. their spatial distribution is more or less even and covers both unconfined and confined aquifers. Shallower and unconfined GW monitoring wells are more widespread, as their installation is less expensive and less time consuming. Usually but not necessarily unconfined aquifers will be those involved in feeding GDEs. In Latvia and Estonia, the monitoring data, including GW level observations, are available on request or at the particular monitoring performer institution. GW monitoring data records generally starts from 1960-ies in Latvia and Estonia, while the measurement frequency changes in a wide range: earlier decades contains measurements recorded at frequencies of few times per week to once per month, but since ~2010 automatic level loggers has been installed in some of the wells which provides recordings up to twice per day. State monitoring sites certainly cannot cover the whole territory, therefore, it is useful to inspect available additional data from private boreholes and dug wells in the archives and on site. It is useful to explore if there is a larger factory or farm nearby, likely they will have their private decentralised GW supply. All larger GW consumers must perform GW monitoring and provide it to the state institutions.

- **GW chemical composition**, similarly to GW levels, can be gathered from the hydrogeological maps, from monitoring programmes and private boreholes and wells, yet the chemical properties of GW are monitored in smaller magnitude and less frequently.

- **Developed geological and hydrogeological mathematical models** are another good data source. Regional mathematical hydrogeological models are available both in Latvia and in Estonia (e.g. https://www.puma.lu.lv, http://virumudel.ut.ee/avaleht/). It is possible to use them as a data source of geological settings (properties of aquifers and aquitards), GW levels, interaction of aquifers, and GW flow system in different water bearing formations, interaction with surface water bodies. Some hydrogeological models include GW chemical composition and migration of pollutants. It is also possible to implement a study site in an existing mathematical model with better resolution or cut out fragments of it. It is helpful when a conceptual or new numerical model is developed, as larger versions can serve as boundary conditions for new projects. Most common mathematical models are developed and maintained by state surveys or academic institutions.

- **Reanalysis, meteorological observation data, surface runoff.** Meteorological data are crucial for determining GDE dependence or vice versa, independence on precipitation as well as assessing drought risks to GDTE. There are several sources of good quality meteorological data: 1) meteorological monitoring by state survey (basic ground observations); 2) gridded, interpolated meteorological datasets, such as the E-OBS database (Cornes et al. 2018) developed by the European Climate Assessment & Dataset (ECAD) project that covers also Latvia and Estonia and can be freely downloaded from ECAD homepage (https://www.ecad.eu/); 3) reanalysis of gridded datasets. European Centre for Medium-Range Weather Forecasts has developed several reanalyses of the global climate products including meteorological data monthly, daily near real time and even with back extension. The latest products are ERA5 and ERA5-Land with horizontal resolution 0.25° x 0.25° and 0.1° x 0.1°, respectively. It is a good source of evenly distributed meteorological data and without discontinuations in time series. Reanalysis data can be downloaded online for free from The Copernicus

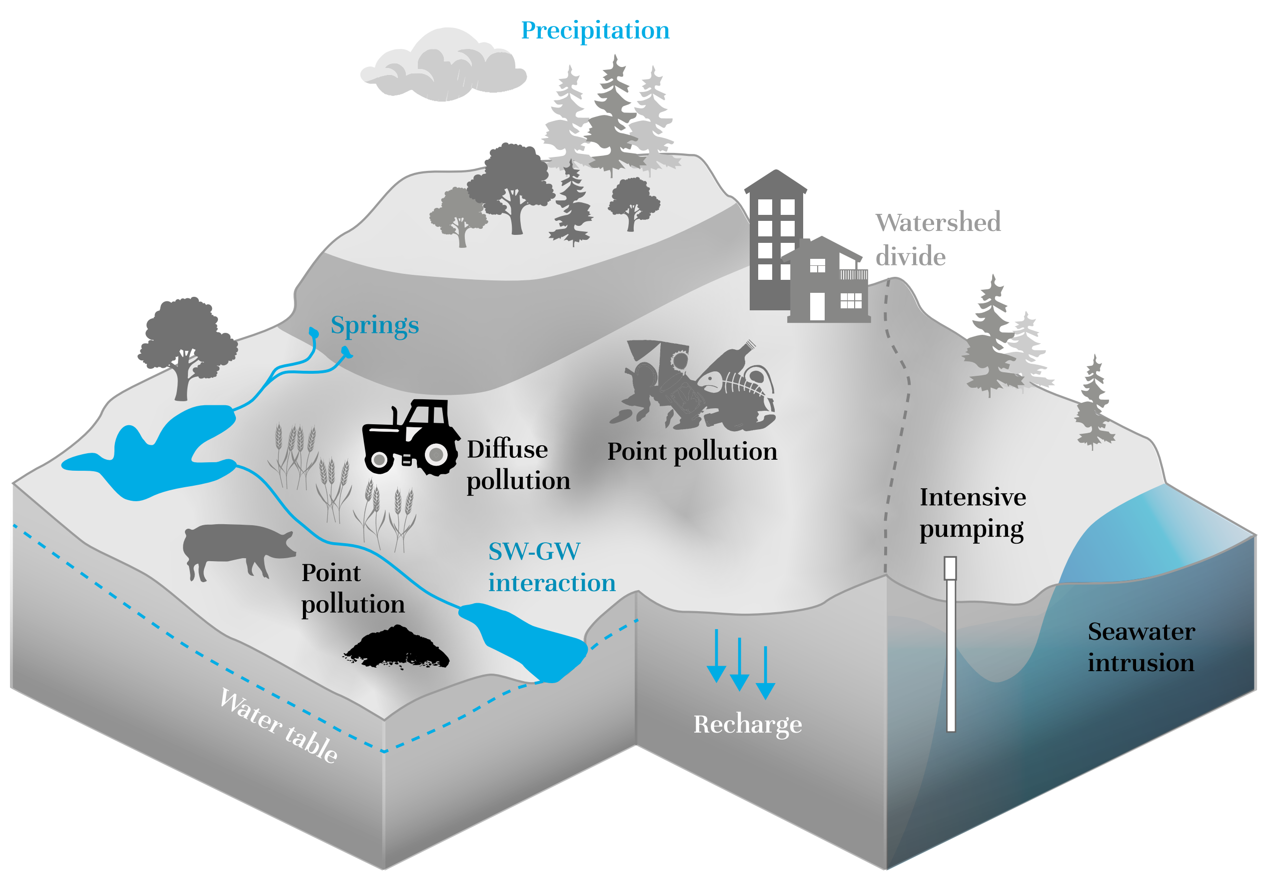

Figure 8.1. *Hydrogeological conceptual model. Author: I. Retiķe.*

Climate Change Service homepage (https://cds.climate.copernicus.eu/). All data sources have their advantages and disadvantages. In case of meteorological observations, they are the primary source for all other datasets, however, they are sparse, discontinued and can represent only local situations. Also, available parameters are limited to observed ones. In case when a meteorological station has a long time series and is closely situated to study the area, ground observations should be first to be used. Reanalysis has some advantages as it is not interrupted neither in time, nor spatially. The dataset includes other useful parameters: snowpack, surface runoff, subsurface runoff, solar radiation, soil moisture and other important parameters.

- **Mining data** must be gathered if there might be significant GW lowering close to the study site. This also relates to mining of other geological resources, as large mining sites might disrupt natural hydrogeological systems. State geological surveys collect and store information about mining sites and the extraction volumes.

- **Land use, soil properties.** For a larger scale, soil types can be obtained from ISRIC — World Soil Information, which has developed SoilGrids digital soil maps (https://soilgrids.org/) or national soil maps. Land use is available from The CORINE Land Cover current and past situations (https://land.copernicus.eu/).

- **Topography or elevation of territory** can be drawn from topographic maps of various scales, mostly 1:50 000, 1:25 000 or 1:10 000. In the Baltic countries, the most accurate topographic map is at scale 1:10 000 – these maps were developed during the Soviet times. DEM (digital elevation models) is another good source of surface elevation and can be available at national as well as European levels (e.g. EU-DEM available at https://land.copernicus.eu/).

- **Global and regional climatic models** have been developed that might be used to forecast climate change in the near future. Such models can be helpful to assess possible climate change impact on GDEs, as the models can predict the future pattern of extreme droughts and wet periods above normal condition.

- **Point and diffuse pollution sources** can also be outlined in maps or found in pollutant data bases mainly maintained by national governmental institutions.

8.3. Conceptual models

In the Groundwater Directive and in several CIS Guidance Documents, the use of conceptual models is mandatory or recommended (EC 2010). The design of a GW monitoring programme requires basic knowledge with respect to the hydrogeological framework and the GW flow systems within the relevant aquifers, aquitards and aquicludes. The description of the understanding of the hydrogeological framework and the hydrological and hydrochemical processes occurring is called a conceptual model (Jousma et al. 2016).

A hydrogeological conceptual model may describe and quantify the relevant geological characteristics, flow conditions, hydrogeochemical and hydrobiological processes, anthropogenic activities and their interactions. The degree of details is based on the given problems and questions. Monitoring results can be used to improve the understanding of the system and the effectiveness of measures. If necessary, for a better understanding, or for a selection of the most appropriate measures, the conceptual models may evolve into more complex numerical models. The starting schematic model can definitely be called a conceptual model (for example as in Figure 8.1.). A complex numerical model is definitely not a conceptual model anymore.

8.4. On-site reconnaissance

The next step, after basic information is gathered, is visiting the study site and collecting additional information on site. If the study area is well examined previously or continuous observations are available, the necessity to further described techniques will be less needed. On-site hydrogeological methods are: measuring GW level, discharge of springs and wells, sampling of existing wells, sampling of springs, new temporary or permanent well installation, tracer methods, etc.

Before going to the site, always a list of takeaway materials and spare sample containers must be taken. It is advisable to take photos of the site and not to forget the GPS device.

8.4.1. Groundwater sampling

Here, conventional methods and one sophisticated technique are briefly reviewed with respect to their capability of providing representative samples as follows:

- Bailers or depth samplers are the grab samplers that operate by lowering the device to a known depth in the well's water column, closing the valve at the depth of the screen interval or at the bottom and raising it to the ground surface.

- Suction devices lift the water sample by applying suction directly to the water or via a collection bottle. Suction can either be generated manually or by a pump (e.g. peristaltic or centrifugal type) but the sampling depth is limited to 8 or 10 meters.

- Gas-driven devices apply gas (air) creating positive pressure directly on the water that drives it from the borehole – backflow being prevented by check valves. Usually compressed air is pumped down the borehole through a delivery tube. The air then forces water up through a second tube (acting as an airlift pump) and the air water mixture emerges at the head of the well.

Sophisticated techniques

To take a representative sample, the sampling procedure should meet the following requirements:

- allows removal of stagnant water from the well (called purging) by means of a submersible pump so that the sampled water represents the water in the aquifer and not the water that has stayed in the well for a prolonged period;

- avoids degassing of the sample and volatilisation of components in if volatile compounds are to be analysed;

- prevents oxidation caused by contact with the atmosphere and avoids contamination of the sample and the well.

Submersible pumps are lowered into the borehole and water is driven out continuously at the surface. The following three principles are used to drive out the water: gears or rotor assembly (electric centrifugal pump), gas-operated plunger (piston pump) or a gas operated diaphragm (bladder pump). Submersible pumps are rated acceptable for sampling GW for all parameters, including volatile organic carbon, trace metals and dissolved gases.

For manual sampling of hand-dug wells (which cannot be purged) all that is required is a weighted sampling can with a rope attached to its handle. The can is then carefully lowered down the well until it fills with water and is then brought out of the wall. The simplest form of a water sampling device is a bottle attached to a string. To lower a plastic or glass bottle in a body of water it is necessary to use a bracket or holder of sufficient weight to overcome the buoyancy of the bottle and allow it to sink as rapidly as desired to the required depth.

The preferred type of sampler in the field for GW sampling is the submersible pump. The sampler should be cleaned and rinsed frequently. Sampler should also be briefly checked for functioning, closing of caps, if applicable, and condition of the cable by which the submersible pump will be lowered inside the well.

Field parameters

Some GW quality parameters are likely to change significantly after sample collection as a result of temperature change, degassing, mineral precipitation, and other chemical, physical, and biological reactions, and should be measured immediately in the field rather than in the laboratory. There are five physico-chemical parameters which are usually measured in the field (also called, field parameters). These parameters are temperature (T), pH, specific electrical conductance (SEC), oxidation reduction potential (ORP) and dissolved oxygen (DO). Digital multimeter is the best option for measuring field parameters, however, pH can be measured with litmus paper and temperature can be estimated with a thermometer. In addition, the concentration of nitrate and alkalinity may help in GDE assessment. Techniques in detail are described in Chapter 5.2.

Sampling

Water samples from wells should be taken only after field parameters are stabilised (do not change at least 5 minutes), it does not apply to springs and conventional well sampling. In case of springs, take the sample as close as possible to its discharge, for dug wells, submerge the bailer sampler or bucket as deep as possible and take the sample.

Before taking a sample it is recommended to rinse the container with sample water, exclusion applies to preserved bottles with acid or preservatives. If samples are kept in the freezer, only 70-80 % of the container should be filled to avoid crackings. If analysis will be performed in the near future, the container must be filled full. Always secure the caps tightly.

Bottles which are to be used for collecting microbiological samples must be thoroughly washed and sterilised before use. Sterilising can be carried out by placing the bottles in an autoclave at 121°C for fifteen minutes or, if the caps of the bottles do not contain plastic or rubber materials, in an oven at 170°C for at least two hours. Bottles to be used for the collection of pesticides are to be rinsed with organic solvent prior to use. This should be done in the laboratory. It is better to use HDPE plastic bottles, teflon or glass (preferably dark) with a tight seal. Volume of containers depends on analysis. The best option is to ask the intended laboratory to provide already prepared vials, bottles or containers or describe their requirements.

Sample labelling

Label the sample container properly, preferably by attaching an appropriately inscribed tag or label and secured with transparent film or tape. Alternatively, the bottle can be labelled directly with a marker. Information on the sample container or the tag should include:

- sample number;

- date and time of sampling;
- source and type of sample;
- pre-treatment or preservation carried out on the sample;
- any special notes for the analyst;
- sampler's name, location.

It is a good practice to ask the intended laboratory how to label samples and what information should be included in the provided sample list.

Sample preservation

If samples collected for chemical oxygen demand (COD) analysis cannot be determined the same day they are collected they should be preserved below pH = 2 by addition of concentrated sulphuric acid. This procedure should also be followed for samples for ammoniacal nitrogen and total oxidised nitrogen analysis.

Samples which are to be analysed for the presence of metals should be acidified to below pH = 2 with concentrated nitric acid. Such samples can then be kept up to six months before they need to be analysed. Analysis of bacteriological samples should be started and analysed within 24 hours of collection.

Some samples need to be preserved or fixed in the field. For dissolved oxygen fixing, every field operative should bring three pipetted glass or plastic stoppered 500 ml bottles containing the DO fixing solutions.

For other parameters (e.g. COD, NH_3, NO_2, NO_3), addition of concentrated sulfuric acid should be done in the field after sampling. For heavy metals, addition of nitric acid needs to be done in the field after sampling. Therefore, the field operative should be equipped with two pipetted glass or plastic stoppered 100 ml bottles containing the two acids.

Sample transportation and storage

After labelling and preservation, the samples have to be packed for transport, preferably in an insulated cool box. After sampling, many water quality parameters undergo chemical or biochemical reactions in the sample bottle causing the concentration to change from that which was present in the watercourse. To prevent this alteration of parameter values, ideally, all samples should be kept at a temperature below 4°C and in the dark until they are analysed. If this is not possible, then at least samples for BOD, coliforms, pesticides and other organics that are likely to volatilise must be kept at 4°C, and dark. Samples should be transported to the laboratory as soon as possible, preferably within 48 hours.

8.4.2. Groundwater quantity and dynamics

The quantity of water in the subsurface system can be estimated by various methods, mainly GW balance calculations (described in more detail in Chapter 2.2.1) or with the use of numerical GW models. GW balance calculations can be applied to quantify the different components of GW recharge and discharge as well as the accompanying changes (increase or decrease) in groundwater storage in a selected period (Juosma et al. 2006). Numerical GW flow models may contribute considerably to the understanding of the GW flow and GW quality development in the area considered. The models can be extremely useful in determining the directions and rate of GW flow.

Both methods require considerable and accurate data collection from the study area. If required data gathered in the preliminary stage is insufficient, for instance, for building conceptual models or assessment of changing conditions of the environment, then additional field works must be performed.

There are two types of additional data that is required to estimate or improve GW quantity and dynamic assessment, i.e. GW level and GW discharge or yield in springs and wells.

Both manual-operated and automated-recording instruments are available to measure piezometric levels in boreholes. The accuracy of piezometric level measurement is a function of the selected methodology and of the operational conditions. For many purposes, an accuracy of 0.01 m can be considered sufficient to measure groundwater level (Jousma et al. 2006).

Groundwater level is a crucial parameter of aquifers as it allows us to determine interaction between elements of the hydrogeological system and monitor GW response to conditions environmental and anthropogenic.

There are various methods developed, manual and automatic. All have their pros and cons, therefore it should be considered cost-effectiveness versus precision and data regularity.

Most frequently used manual methods are:

- Dipper and popper. Long tape with popping sound indicating water level;
- Manometer or pressure gauge. Suitable only for artesian wells. Easy readable, but calibration needed;
- Wetted-tape. Tape with changing colour when it reaches the water surface. Not recommended for quality monitoring, as it may contaminate water in the well;
- Two electrode methods. Sound or light indicate when GW level is reached. Regular maintenance is needed but a very accurate method;
- Inertial devices.

Automatic methods:

- Pressure data logger. Measures pressure well below the water table. High accuracy, but calibration needed. Temperature effect. Can be used for springs;
- Float recorder. Expensive method and large well diameter needed. The most accurate method. Can be used for springs;
- Ultrasonic sensors. Can be used for springs.

Combined methods:

- Air-line or bubble-in method. Low accuracy, but can be used for springs. Expensive, regular maintenance;
- Acoustic resonance sensor. The most expensive method. Suitable for complicated wells.

Methods to measure groundwater discharge or yield

- Volumetric method. Is the simplest method. Large tank and chronometer are needed. Applicable to low discharge and discontinuous flow. High precision can be achieved if accurately used. Not suitable for ascending springs, springs with large outflow and high discharge;
- Discharge measurements using a weir. Low cost method using a weir tank with the water discharging over a V-notch or a rectangular notch. Good option for ascending springs that form pools and larger surface water objects. Afterward calculations are needed;
- Orifice bucket method;
- Volumetric meter method;
- Rotameter tube method;
- Orifice plate method;
- Venturi tube method.

Selected methods for measuring spring discharges are described in detail in the WaterAct Spring monitoring guide (https://allikad.info/manuals/volunteer_monitoring_manual_ENG.pdf).

8.4.3. Water properties/quality

The frequency of sampling or monitoring plan depends on the rate of expected changes in the GW quality. For a general reconnaissance of water quality, including water types, origin of the water and first indications of contamination, the following parameters should be analysed: Ca, Mg, Na, K, NH_4, Fe, Mn, SiO_2, HCO_3, SO_4, Cl, NO_3, PO_4 (major ions). Analysis should also include direct measurements of field data. Groundwater quality parameters and field parameter techniques are covered in more detail in Chapter 5.2.

Regarding the chemical status, Article 3 of the Water Framework Directive (EU 2000) states the following criteria for assessing GW chemical status: "(a) groundwater quality standards as referred to in Annex I, (b) threshold values to be established by Member States in accordance with the procedure set out in Part A of Annex II…".

Where a GW quality standard is not sufficient to ensure that the environmental objectives set out in Article 4 of the WFD will be met, then a more stringent value is needed. This also considers any significant damage to GDEs. These more stringent values are defined as Threshold Values (TVs) and are to be established at an appropriate scale (national, river basin district or at GWB scale). They should be applied to the GWB (or relevant scale) during the relevant classification tests and if exceeded at relevant locations in the GWB, appropriate investigation need to be undertaken, including investigations for significant damage to GDEs, in order to confirm that the achievement of the WFD objectives is not compromised (EC 2012).

Characterisation should determine what GW chemical pressures may be acting on the GDE, and information about anthropogenic pollutants present in the GWB should be obtained from the GWB chemical monitoring network. For example, in an intensive agricultural setting, it is likely that the most important substances affecting GDEs are nitrate and phosphate. The substances which are identified as important should be considered as the starting point of the risk assessment.

8.5. Data follow up

After the first assessment of the gained data before fieldworks and after processing sample results, sometimes there is information still missing to completely solve the predefined tasks.

Further data collection and processing is needed to determine the GWB interaction with the GDE – usually regular monitoring is needed, as the GDEs can be fed by GW seasonally or proportion of water sources can be changing depending on availability of water in the system. For GDTEs at risk, damaged or with poor status, monitoring of the GW quantity and quality should be performed for a longer period of time (EC 2012).

It is common that monitoring should be performed at least every meteorological season. It is advisable to take samples, measure GW level and discharge on a monthly basis. To cover the full cycle of monitoring, the duration must be at least one year, however, it depends on aquifer properties feeding the GDE and meteorological conditions, as during a wet year or unusually dry periods the local water cycle can act differently. For further reading on monitoring: CIS Guidance No. 7 (Monitoring) and No. 15 (Groundwater Monitoring), Technical Report No. 3 (Groundwater Monitoring) and CIS Guidance No. 26 (Risk Assessment and Conceptual Models)

The criteria required for the establishment and application of thresholds is set out in the CIS Guidance No. 18: Guidance on groundwater status and trends assessment (EC 2009). One of the specific criteria is that TVs should "*…aim to protect … groundwater dependent terrestrial ecosystems.*"

Collecting new groundwater samples should be performed at least every meteorological season (McColl, 2005). Again, if the financial resources are limited, in sampling points where previous results are not changing in time or reflect stable recharge conditions, sampling can be skipped.

Automatic GW data logger is a good option to be installed in distant study areas, as it can provide continuous level readings without the need to visit the site very often.

Drilling new wells is expensive, therefore the best place to install them should be carefully considered. The considerations must include how deep and what kind of materials will be used. It is better to consult the drilling companies before planning so that it can be cost-effective.

What else can be done:

- Establishment of wells to measure water levels in the near-surface deposits and their hydraulic relationships with the underlying GWB;
- Remote sensing data to predict water levels in GDEs;
- Geophysical surveys, e.g. resistivity, ground-penetrating radar to confirm stratigraphic relationships and location of water-bearing/conductive strata;
- Drilling of deeper piezometers into the underlying GWB;
- Measurement of other sources of chemical pressure, e.g. atmospheric nitrogen deposition that may cause a critical loading to be exceeded (EC 2012);
- Including additional substances or parameters to be analysed, e.g. stable or radioactive isotopes and trace elements.

All additional data can be implemented in a developed conceptual model and upgraded into a more complex numerical flow model.

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