Interreg Estonia-Latvia project No. Est-Lat62
“Joint management of groundwater dependent ecosystems in transboundary Gauja-Koiva river basin (GroundEco)”

FINAL REPORT

2020

Table of Contents

ABBREVIATIONS

EC – European Commission

EU – European Union

GDTE – groundwater dependent terrestrial ecosystems

GWB – groundwater body

GWD – Groundwater directive

TVs – threshold values

WFD – Water Framework Directive

INTRODUCTION

Groundwater dependent terrestrial ecosystems (GDTE) are ecosystems that rely upon groundwater, e.g. fens, spring flushes, swamp forests. Significant changes in groundwater chemical composition (quality) and/or groundwater levels (quantity) can damage the quality of GDTE or even destroy it. GDTEs provide important ecosystem services such as regulation of water cycle, drinking water and other resources, pollution filtering, peat and carbon accumulation, unique biodiversity, interconnection with other ecosystems (landscape functioning), recreational and scenic value, and cultural heritage.

According to the EU Water Framework Directive (2000/60/EC), the whole groundwater body (GWB, a management unit set by each Member State) is considered in poor status if anthropogenic pressure on groundwater causes significant damage to GDTE. Then, specific actions (in WFD – programmes of measures) must be planned and carried out to improve the status of GWB and thereby, restore the quality of damaged GDTE.

Groundwater is a dynamic system which cannot be divided by human drawn boundaries such as country borders, and many valuable GDTEs depend on the quality of these water resources. From a hydrological viewpoint, the transboundary Gauja-Koiva river basin has a joint water cycle. It means that human induced activities (e.g. water extraction or pollution) on one side of the border can affect groundwater and associated ecosystems on the other side. Thus, without joint activities and cooperation it is impossible to sustainably manage transboundary natural resources in the long term.

A theoretical approach on how to identify, assess, and monitor GDTEs has been previously developed in Estonia. Similar climatic and hydrogeological conditions allowed us to adapt the methodology to Latvia and jointly update it to fit the needs of both neighboring countries – Estonia and Latvia. During Interreg Estonia-Latvia project Est-Lat62 "Joint management of groundwater dependent ecosystems in transboundary Gauja-Koiva river basin" or GroundEco project, joint methodology for identification and assessment of GDTEs was developed which was used to identify GDTEs in transboundary Gauja-Koiva river basin. The methodology and best GDTEs assessment techniques were tested in two pilot areas – Matsi spring fen in Estonia and Kazu leja valley in Latvia. Finally, recommendations for better GDTEs monitoring and future works were developed. All results will be used to support the development of 3rd cycle River Basin management Plans (2021–2027) for Estonia and Latvia.

We hope that this report will be useful for many habitat and hydro(geo)experts working with assessment of GDTEs in line with European Water Framework Directive as well as serves as a good starting point for better understanding of GDTEs functioning and linkage with groundwater resources.

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1. Description of GroundEco project

Project "Joint management of groundwater dependent ecosystems in transboundary Gauja-Koiva river basin (GroundEco)" is being implemented with the support of Interreg V-A Estonia-Latvia cross-border cooperation programme 2014-2020.

Total budget of the project is EUR 411,764.72. Interreg V-A Estonia-Latvia cross-border cooperation programme contributes EUR 350,000.00 EUR and partners' self-contribution is EUR 61,764.72. Project duration is 26 months (10.05.2018-09.07.2020). Main materials developed during the project are available at the project homepage https://www.bit.ly/GroundEco.

The objective of GroundEco project is to enhance sustainable management of common groundwater resources and associated ecosystems in the transboundary Gauja-Koiva river basin.

The project's main activities are:

- identification and exchange of data necessary for project implementation;
- development of a joint methodology and its implementation in Gauja-Koiva river basin;
- pilot studies and development of conceptual models (Matsi spring fen in Estonia and Kazu leja valley in Latvia);
- recommendations for better management of transboundary Gauja-Koiva river basin and project result dissemination.

The project partnership consists of seven partners from Latvia and Estonia:

1. Leading partner is Latvian Environment, Geology and Meteorology Centre (LEGMC) (Latvijas Vides, ģeoloģijas un meteoroloģijas centrs; https://www.meteo.lv/), the project coordinator is Inga Retiķe (inga.retike@lvgmc.lv);
2. Tallinn University, Institute of Ecology (TU) (Tallinna Ülikool, Loodus- ja terviseteaduste instituut; https://www.tlu.ee/eco), the coordinator from TU side is Jaanus Terasmaa (jaanus.terasmaa@tlu.ee);
3. University of Latvia, Faculty of Geography and Earth Sciences (UL) (Latvijas Universitāte, Ģeogrāfijas un Zemes zinātņu fakultāte; https://www.geo.lu.lv/), the coordinator from UL side is Andis Kalvāns (andis.kalvans@lu.lv);
4. Ministry of Environment of Estonia (MEE) (Keskkonnaministeerium; https://www.envir.ee/et), the coordinator from MEE side is Kersti Türk (kersti.turk@envir.ee);
5. Nature Conservation Agency, Latvia (NCA) (Dabas aizsardzības pārvalde; https://www.daba.gov.lv/public/), the coordinator from NCA side is Ieva Veikšina (ieva.veiksina@daba.gov.lv);
6. Vidzeme Planning Region, Latvia (VPR) (Vidzemes plānošanas reģions; http://www.vidzeme.lv/), the coordinator from VPR side is Maija Rieksta (maija.rieksta@vidzeme.lv);
7. Geological Survey of Estonia (GSE) (Eesti Geoloogiateenistus; https://egt.ee/et), the coordinator from GSE side is Siim Tarros (siim.tarros@egt.ee).

GroundEco activities in a nutshell:

WP	Description of activities	Main results
WP1 Data exchange, mobilisation and collection	• Data exchange and mobilisation; • Pilot studies and data collection (Matsi spring fen and Kazu Leja valley).	Data exchange report provides an insight into the data identification, specification, exchange and storage processes during the GroundEco project. During pilot studies tools and approaches were tested to find best parameters and data analysis techniques for GDTE assessment, as well to collect data necessary for GDTE identification and assessment methodology testing (including assessment schemes).
WP2 Development of joint methodology and implementation	• Development of common structure of groundwater and habitats data; • Development of joint methodology; • Identification of groundwater dependent ecosystems in Gauja-Koiva river basin; • Development of conceptual models and assessment of pilot groundwater dependent ecosystems.	As a result, a joint methodology for GDTE identification and assessment was developed including quantity and quality assessment schemes. According to the methodology, GDTEs were identified in the Gauja-Koiva river basin. Conceptual models were developed for pilot areas (Matsi spring fen and Kazu Leja valley) and their status was assessed according to developed joint methodology. All results are incorporated in this report.
WP3 Recommendations and dissemination of results	• Recommendations for transboundary groundwater monitoring and groundwater dependent ecosystem assessment; • Development and dissemination of informative materials for target groups; • Workshop "Introduction to development of conceptual models"; • Seminar on groundwater dependent ecosystem assessment; • Special conference session for groundwater dependent ecosystems; • Knowledge exchange and training at European Geosciences Union General Assembly 2019.	The recommendations for better GDTE assessment and monitoring, and future prospects are incorporated in this report. During the project 3 main public events were organized to introduce target groups with project progress, main results and to train to use the developed methodology. Experience exchange and presentation of main results to the experts all around the world was done during the EGU2019 conference in Vienna, Austria. Additionally, materials were developed for target groups to raise awareness of groundwater and associated nature protection. Main project materials and organized/ attended events are available at **bit.ly/GroundEco**

2. Background for identification and assessment of groundwater dependent terrestrial ecosystems (GDTEs) according to the Water Framework Directive (WFD)

2.1. Role of groundwater dependent terrestrial ecosystems in the Water Framework Directive

Water Framework Directive (WFD, 2000) aims to protect all water resources, including groundwater. For groundwater, environmental objectives are set in Article 4 and the main goal is to achieve good groundwater status which is a combination of both, good quantitative and chemical status.

Terrestrial ecosystems that are directly dependent on groundwater (GDTE) can affect the status of groundwater body (GWB). If a significant damage to GDTE is caused by anthropogenic alterations of groundwater level or chemical composition (groundwater chemistry or concentrations of any pollutants), a whole GWB is in poor status.

In order for the terrestrial ecosystem to be a part of GWB status assessment, it must be directly dependent on GWB. This means that existence and quality of such terrestrial ecosystems relies on groundwater input (flow, level, certain chemical composition). Critical dependence upon a GWB is most likely where groundwater supplies GDTE for a significant time period of a year (continuously or for most part of the year) (EC 2011).

2.2. How to determine whether a terrestrial ecosystem directly depends on groundwater body

There are cases when the dependence of GWB is easy to identify, for example where a spring is clearly visible. Still there are many situations where terrestrial wetland ecosystems depend or partly depend on other water sources such as surface water, precipitation or distance from the sea.

Member States might use available knowledge base from localized site assessments for the Habitats Directive (1992). If it is unclear whether the site is GWB dependent, a staged approach for screening out the ecosystems using **ecological and hydrogeological information**, as well **expert judgement** may be used to rank them in terms of likely dependence on groundwater.

First step in such a screening approach is to identify whether certain terrestrial ecosystems theoretically can be dependent on groundwater resources. Member States may use lists of vegetation communities and habitat types under Habitats Directive to rule out sites which do not have groundwater dependent vegetation. Then a conceptual model of a GWB may be used to identify if groundwater in a GWB is likely to be discharging to and supplying a GDTE.

Conceptual models (EC, 2010) may include information on habitat type, the ecosystem and the GWB and their linkages, or approximate understanding of the linkages, as well as the degree of dependence on the GWB. Considering the large number of GDTEs, conceptual models may never be developed for all sites. The priority should be given to significantly damaged ones and to GDTEs in GWBs with significant pressures. The conceptual model remains open to revision, as more data becomes available.

When no monitoring results or conceptual models are available, expert judgement may be used. Member States may have lists of plant communities occurring in the particular country, and their dependence on groundwater can rely on expert knowledge using certain principles. If no data is available, adaptation of existing knowledge base from neighboring countries with similar climatic and hydrogeological conditions may be considered.

If an ecosystem is currently no longer groundwater dependent, due to lowering of groundwater levels, but has been dependent on groundwater in the past, it should also be considered as GDTE. GDTEs can be damaged also by rising groundwater tables, for example, if a wetland is flooded after construction of a dam that obstructs the discharge of groundwater. Article 4 of WFD (2000) requires that all bodies of groundwater are protected, enhanced and restored. However, priority should be given to preventing deterioration of sites which are currently considered groundwater dependent, to avoid damage in the future (EC, 2011).

UKTAG (2004) suggest two simple complementary techniques which may be adapted to identify GDTEs:

- identify ecosystem types that depend upon groundwater and then identify which of these ecosystems are most ecologically significant in the particular country;
- identify points where groundwater interacts with the land surface and assume that any terrestrial ecosystem present is utilizing this water.

2.3. How to determine whether there is a significant damage to groundwater dependent terrestrial ecosystem

According to EC (2003) expression “significant damage” is based upon the magnitude of the damage and the ecological or socio-economic significance of the terrestrial ecosystem. The magnitude of damage depends on whether GDTE continues to fulfill its ecological or socio-economic function. A GDTE which is crucial for regional tourism might be considered to have socio-economic significance, thus any changes in groundwater quality or quantity supplying the GDTE may lead to its damage and reduced visitor numbers.

If a GDTE belongs to the Natura 2000 network, the failure to meet its conservation target can be interpreted as significant damage (as far as groundwater status is concerned). For GDTEs that are not part of Natura 2000 network, formulation of targets and derivation of groundwater requirements (chemical and quantitative) may be developed by Member State and then comparison between actual and desirable situations can be performed.

Member States are strongly encouraged to carry out analysis of significant damage to GDTEs in multidisciplinary teams as both good ecological and hydrogeological understanding of the sites is required.

2.3.1. Threshold values and trigger values

To protect groundwater, quality standards and threshold values (TVs) should be established, and methodologies based on a common approach developed, in order to provide criteria for the assessment of the chemical status of GWBs. The requirements for setting up of threshold values (TVs) are fixed in Groundwater Directive (GWD, 2006) or so called “daughter directive” of WFD. The criteria required for establishment and application of TVs is set out in European Commission guidance document (EC, 2009).

TVs must be established in such a way that any exceedance at representative monitoring point will indicate a risk that environmental objectives under Article 4 of WFD are not being met in GWB. When establishing TVs Member States must consider not only pressures, hydrogeological conditions and geochemical pathways, but also the extent of interactions between groundwater and associated aquatic and dependent terrestrial ecosystems. When establishing threshold values, it is particularly important to consider pathways as for example, reducing conditions may remove nitrate, but high calcium content may co-precipitate phosphate.

For surface water such quality criteria are called “environmental quality standards” (EQS), but for GDTEs the expression “trigger values” are most commonly used. If trigger values are exceeded in the GDTE (not GWB), significant damage to it could be caused. To determine trigger values, it is necessary to understand the levels that different types of GDTEs can tolerate in terms of chemical inputs which do not result in significant damage. After that TVs for GWB can be determined by taking into account dilution and attenuation between the GWB and the dependent terrestrial ecosystem through understanding of the likely pollutant linkage.

Annex III of GWD (2006) states that for the purposes of investigating whether the conditions for good groundwater chemical status are met, Member States should (where relevant and necessary) on the basis of relevant monitoring results and developed conceptual model for GWB, assess: a) the amounts and concentrations of pollutants being or likely to be transferred from the GWB to GDTEs and b) the likely impact of the amounts and concentrations of the pollutants transferred to the GDTEs.

2.3.2. GDTEs that depend on concentrations of naturally occurring substances above threshold values

Some ecosystems are adapted to higher concentrations of naturally occurring substances and according to WFD this is not pollution. For example, in a region of the Netherlands, zinc naturally occurs in high concentrations and local ecosystems have adapted to those conditions and differ from ecosystems elsewhere. Such ecosystems may also develop if high zinc concentrations come from anthropogenic pollution, but such a case should be considered pollution and damage to the currently existing ecosystem.

2.4. How to determine the groundwater quantity and quality needs of groundwater dependent terrestrial ecosystems

When GDTE has been identified, there more likely will be key periods during a year when groundwater input is critical to the functioning of the individual GDTEs. Ecologists in Member States may understand the water quantity needs of specific GDTEs, but as quantitative requirements are specific to each GDTE, the necessary knowledge base may be absent. Where quantitative information is available, hydrological modelling can be used to link groundwater resources to a GDTE.

Characterization process should determine what groundwater chemical pressures may be acting on the GDTE. The information from chemical groundwater monitoring should be used to identify pollutants. If such data are lacking, then potential pressures from the initial GWB characterization process may provide necessary information. For example, if the area is intensively used for agriculture then most likely pollutants will be nitrates and phosphates.

However, different GDTE categories can have large tolerance ranges to specific pollutants. Some GDTE vegetation types only occur in certain locations due to the unique chemistry of water supply in the area, still others are more or less tolerant to elevated concentrations of substances such as nutrients. Further investigation is needed to understand the effect of elevated pollutant concentrations on each GDTE category. This information can further be used to develop trigger values and help to establish TVs for GWB. The information exchange between Member States is strongly encouraged to identify both, quantity and quality requirements of specific GDTEs.

2.5. Characterization and risk assessment

Typically, characterization of GWB is a two-step risk assessment procedure. During initial characterization the uses of GWBs and the degree to which they are at risk of not meeting the environmental objectives set in Article 4 of WFD are assessed. If GWB is considered to be "at risk" of not meeting the objectives, then further, more detailed characterization process is needed. In many cases it would be appropriate to combine both risk assessment procedures into one stage.

Initial characterization may gather available information on diffuse and point source pollution, characterize overlying strata (define vulnerability to pollution) and assess the pressures from water abstraction and artificial recharge. One of the outcomes of this process should be identification of those GWBs for which there are GDTEs. It is recommended to use GIS techniques in cases when there are many possible GDTEs.

If a GWB is at risk due to negative effects of GWB pressures on GDTEs, an inventory of terrestrial ecosystems with which the GWB is dynamically interlinked is needed. Even though WFD prioritizes ecosystems, in practice it might be necessary, if large numbers of GDTEs are connected to GWB. Guidance (EC, 2003) provides practical guidelines for prioritizing terrestrial ecosystems. Stepwise approach suggests firstly focusing on Natura 2000 sites and those ecosystems which may experience significant damage (either ecological or socio-economical). Inclusion of anthropogenic pressures analysis can focus the risk assessment, for example for further characterization a source-pathway-receptor approach could be used to conceptualize groundwater flows and anthropogenic pressures.

Classification of the status of GWBs under the GDTE test, for both quantitative and chemical status, is a decision based on the information and data from the monitoring networks, gathered in the previous 6 years (reporting period under WFD). There might be different levels of certainty in the risk assessment depending on the amount of available evidence available about each site, linkages to GWB and pressure action on GWB. Only those defined as being at risk of failing WDF environmental objectives under characterization process should be considered as potentially putting GWB at poor status. For those GWBs where there is significant damage of a GDTE after the characterization process, additional investigation and monitoring is required according to WD and GWD.

2.5.1. Stepwise approach

Firstly, a screening process should identify those terrestrial ecosystems which are (likely) interacting with GWB, which are significantly damaged and the damage is caused by pressures on GWB. Investigations may be used to provide evidence if GDTE is significantly damaged and if this is caused by pressures on GWB. During investigations a magnitude of damage (assess damage magnitude in combination with ecological studies), linkage between GWB and GDTE and relative importance of groundwater for particular ecosystems should be identified. Where there are many GDTEs, it is suggested to use already available information, undertake simple ecological walkover surveys of the site or use expert judgement. Only for significantly damaged sites a more detailed investigation should be carried out to design measures for status recovery.

There may be GDTEs interacting with the GWB, where there is not sufficient information to allow for clear distinction whether these are significantly damaged or not and/or environmental supporting conditions are not quantifiable with a significant degree of confidence. In that case GDTE will remain at risk of being significantly damaged and could be prioritized for further investigation. Where the information and knowledge are insufficient, the status assessment procedure can be stopped and GWB is classified as of good status under this test, but will remain to be at risk.

If the first step confirms that GDTE is significantly damaged and interacting with GWB, then the second step should identify whether environmental supporting conditions are not met due to anthropogenically induced changes in the GWB and if that is the case, which ones are not being met. If the departure from the environmental conditions is the result of groundwater abstraction, then GWB is classified to be at poor status. For chemical assessment developed GWB TVs can be used which may be based on GDTE trigger values.

Many GDTEs may have not been subject to rigorous status assessments and could be damaged by a wide range of pressures such as afforestation and land development, many of which may be unrelated directly to the quality or quantity of groundwater supply (UKTAG, 2004).

2.5.2. Consideration of groundwater threshold values

When ecosystems with different sensitivities for a certain substance depend on one GWB, the TV used is proposed to be based on the most sensitive GDTE (EC, 2009). If in the first step of status assessment all monitoring results fall under TV, all GDTEs in a GWB are deemed protected. If there are exceedances at some monitoring points, this does not automatically lead to poor status. Appropriate investigations can confirm that the objectives of WFD are still being met and suggested procedure is:

- Derive TVs based on most sensitive GDTE based on GDTE trigger values and dilution and attenuation along the flow path;
- Compare monitoring results from points where the GDTE is dependent on the GWB to this TV (based on conceptual understanding);
- If the average of concentrations in each of these monitoring points is lower than the TV, the GWB is of good chemical status for that particular substance in relation to the GDTE test;
- If the average of concentrations in these monitoring points are higher than the TV, an investigation is needed. Depending on the result, assign either good or poor chemical status for this substance for the GDTE test;
- If a single GWB contains ecosystems with high sensitivity for certain substances as well as ecosystems with low sensitivity for the same substances, the investigation should focus on the high sensitivity GDTE.

If a GDTE has been identified as significant and is significantly damaged from groundwater pressures, GWB is at poor status independent of the size of GWB or GDTE.

2.5.3. Investigations

For GWB where TV is exceeded, but significant uncertainty exists, more detailed site-specific assessment or expert judgment may be used. It is also important to carry out detailed site investigation in the most cost-effective way. The investigation should always be guided by the conceptual model (EC, 2010).

Some methods that are considered to be cost-effective are:

- targeted ecological surveys (vegetation surveys to detect changes in abundance of species indicative of chemical damage such as nutrient enrichment or significant drop in groundwater table);
- walkover hydro-ecological surveys to determine the locations of springs and seepages in relation to critical groundwater dependent ecological features and hydraulic relationships between groundwater and surface water (e.g. ditches, ponds, watercourses);
- establishment of shallow dip 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 GDTEs.

Other suggested techniques are geophysical surveys (resistivity, ground-penetrating radar to predict stratigraphic relationships and location of water-bearing/conductive strata; drilling of deeper piezometers into underlying GWB and measurement of other sources of chemical pressure such as atmospheric nitrogen.

3. Methodologies used for assessment of groundwater dependent terrestrial ecosystems in the European Union

3.1. Results from joint questionnaires

In 2012 a questionnaire was developed and distributed by a European Commission's working group among the member states of the European Union in order to gather information on how the member states have assessed their GDTEs. The purpose of it was to share experiences and to learn from best practices (EC, 2014). Neither Estonia or Latvia responded to the questionnaire because GDTEs had not been determined nor assessed yet. In the GroundEco project these experiences and best practices were taken into account for the development of the joint methodology for GDTE identification and assessment.

A total of 21 countries responded to the questionnaire. From these 19 were Member States and two were non-Member States, who indicated that they have national GDTE designations and have an interest in sharing their knowledge and experience. Although the majority of Member States responded (19 from 28), the compilation of answers does not currently provide a complete overview of the situation in Europe and the work should be continued as there are still many uncertainties and knowledge gaps among the countries (EC, 2014).

In TABLE 1 a compilation of the most significant results from the joint questionnaire and approaches used in Member States are presented. Full answers from each country are available at EC (2014).

TABLE 1

Summary of responses to joint questionnaire on national GDTEs identification and assessment approaches (EC 2014)

No.	Question	Summary response
1	*How GDTEs were identified in your country?*	● Using Natura 2000 (90% or 19 respondents) ● Using additional protected sites (48% or 10 respondents) All 19 Member States rely on Natura 2000 designations when identifying GDTEs. Approximately half of the countries indicated they had additional wetland designations in addition to those declared under Natura 2000 and that these were also considered for GDTE assessments. Seven countries identified the Annex I habitats typologies that they used when selecting GDTEs for assessment.
2	*How has your country identified that terrestrial ecosystems were groundwater dependent?*	● Expert judgement (100% or 21 respondents) All countries indicated that expert judgement played a key role when determining groundwater dependence. Few countries struggled to determine groundwater dependence, but included wetlands in country GDTE assessments. Four countries have a clear assessment criterion and other four countries use field assessments (bottom up approach) to carry out dependency analysis. Over half of the countries rely on the combination of field studies and general wetland characterization criteria to determine groundwater dependency.
3	*How has your country assessed if GDTEs were damaged and what criteria were used to determine the magnitude of damage?*	● Use of national monitoring network (71% or 15 respondents) ● Specific habitats studies (19% or 4 respondents) Many countries responded that this assessment was rather complicated to carry out and in most cases was subjective. Eight countries developed specific damage criteria for wetland assessment. Other countries rely on the conservation status from Habitats Directive reporting, site-specific assessments of damage (if and where available) and expert judgement. Five countries were not able to determine significant damage. Four countries have considered abiotic factors (pressures, trends) to determine if groundwater has the potential to cause damage to a GWDTE, rather than focus on damage at the wetland itself.
4	*How has your country coordinated gathering of*	● Using information coming from different organizations: (76% or 16 respondents)

No.	Question	Summary response
	relevant data and who was responsible for coordination?	● Local level (33% or 7 respondents) The assessment coordination was primarily undertaken at a national level. Seven countries indicated that the assessments were undertaken by working groups and three countries did not have any specific coordination approach.
5	***What are the lessons learned from the first assessment of GDTEs and what changes will be made?***	● Developing significant damage, groundwater dependency and prioritization criteria and/or assessment methodologies (43% or 9 respondents) ● Gathering additional monitoring data (33% or 7 respondents) ● Better coordination (19% or 4 respondents) To sum up, further work is required regarding: developing significant damage and groundwater dependency criteria; improving GDTE condition assessment methodologies and developing abiotic threshold criteria; gathering additional monitoring data; improving knowledge of the pressure source and groundwater pathways to the wetland and then linking this back to measures.
6	***What are the remaining constraints to GDTE assessment in your country?***	● Lack of shared methodologies (52% or 11 respondents) ● Exchange of experience within working groups (52% or 11 respondents) Most of the countries report that there is a lack of supporting guidance documents at EU level. Better linkages between ecologists and groundwater experts at EU level and greater sharing of experiences between countries are necessary. There were requests for sharing on how GDTEs were selected, criteria for determining groundwater dependence and significant damage, and the overall assessment approaches, the use of screening criteria and monitoring data for GDTE assessments. In particular there were requests for the establishment of baseline conditions for GDTEs – what are the reference conditions in terms of ecology and the abiotic factors that countries should achieve.

The main outcomes from the questionnaire can be summarized as follows:

- All countries rely on Natura 2000 designations when identifying GDTEs. About half of the countries also consider additional wetland designations;
- Expert judgment plays a key role in GDTE identification process and only a few countries have clear criteria for identification process;
- Most countries acknowledge that the assessment of the significance of damage to GDTE is subjective. Countries rely on the conservation status from the Habitats Directive reporting, site-specific assessments of damage (if and where available) and expert judgement. Some consider abiotic factors (pressures, trends) to determine if groundwater has the potential to cause damage;
- Majority of countries admit that there is a lack of guidelines at EU level on how to perform assessments and not enough cooperation between ecologists and hydrogeologists. The necessity to share knowledge and good experience between countries is also highlighted. Some countries request to establish joint baseline conditions for GDTEs.

3.2. Methodology for identification and assessment of groundwater dependent terrestrial ecosystems in Estonia

In Estonia a preliminary methodology for identification and assessment of groundwater dependent terrestrial ecosystems was developed in 2015 (Terasmaa et al., 2015). The report on the methodology was written in Estonian. Therefore, the following section presents the translation of the relevant parts of the report, which have not been published in English before. The pre-existing Estonian methodology was taken as a starting point in the GroundEco project for the development of joint methodology for Estonia and Latvia.

3.2.1. Identification of terrestrial ecosystems depending on groundwater bodies

Two principal mechanisms for the dependency between GDTEs were considered (FIGURE 1).

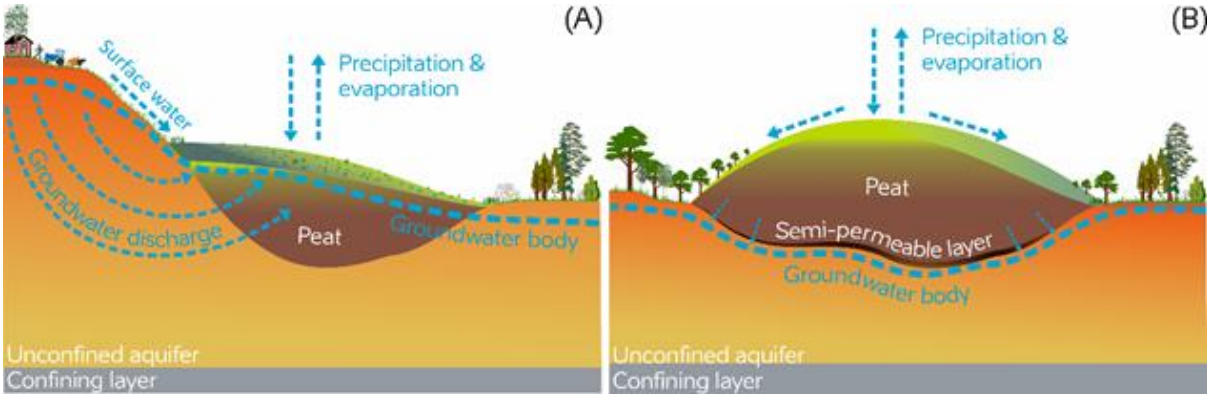


FIGURE 1 **Conceptual models for two main types of terrestrial ecosystem dependence on groundwater in Estonia (and also in Latvia): (A) the ecosystem is fed by the groundwater body, (B) the ecosystem is supported by the groundwater body (Terasmaa et al., 2020)**

These two mechanisms may be combined and may be absolute or partial and can affect different parts of the ecosystem (wetland) differently. For example, a mire that is mostly fed by precipitation may have a section that is fed by springs.

In the case of Estonia, the ecosystems considered depending on groundwater are mires, including paludified forests and grasslands, but not alluvial forests and grasslands). Ecosystems that depend fully on groundwater are spring fens, which have formed on the locations of perennial seepage-springs. Although some spring fens are probably fed by Quaternary aquifers that do not belong to groundwater bodies[1], most larger spring fens are totally or partially dependent on water coming from GWBs. Spring fens may be affected both by changes in the quantity and quality of the groundwater bodies. All the largest spring fens and spring fen groups were selected to the list of Estonian GDTEs.

Minerotrophic mires also have a significant connection with GWBs, as their water regime and hydrochemistry depend mostly on groundwater. The dependency is more significant in the alkaline fens of western and northern Estonia, as the Quaternary cover is thinnest there. Transition mires and bogs are usually not fed by groundwater, and therefore they are assumed to not have a direct dependency. However, the water in the peat layer may not be fully isolated from the groundwater aquifer. A drop in the groundwater 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. That theory is supported by modelling results in some bogs in the vicinity of oil-shale mines in northern Estonia (Kalm & Kohv, 2009; Marandi et al., 2013). The list of GDTEs includes all mires (including bogs) with the size over 20-30 ha that are located on GWBs in a bad and endangered status (Quaternary Vasavere and Ordovician Oil-Shale basin). On the Quaternary Männiku-Pelguranna groundwater body all mires, including bogs, were included. On Ordovician-Silurian Pandivere and Põltsamaa-Adavere GWBs (designated nitrate vulnerable zones according to Nitrates Directive (1991)), larger minerotrophic mires with higher ecological value were included. On other GWBs only larger minerotrophic mires (also some transitional mires) with higher ecological value were included in the list of GDTEs.

[1] Since 2019, all Quaternary aquifers have been joined to groundwater bodies in Estonia

Identification was fully based on the expert judgement. Altogether 71 mires, mire systems or groups of mires were identified as GDTEs depending on GWBs (FIGURE 2). Some of them could be dependent on several GWBs. In many cases the dependency is hypothetical because of the lack of hydrogeological data.

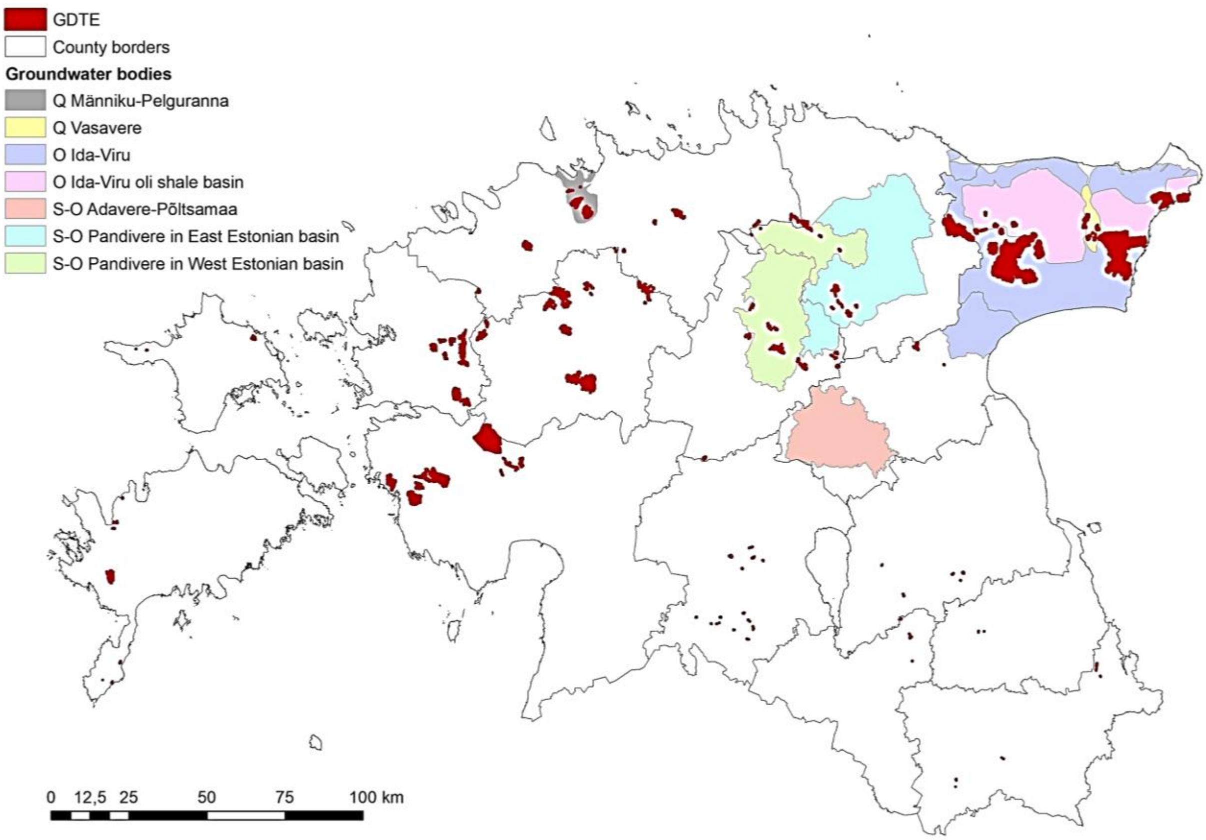


FIGURE 2 **Identified terrestrial ecosystems dependent on groundwater bodies in Estonia (only the groundwater bodies mentioned in the text are shown) (Terasmaa et al., 2015)**

3.2.2. Indicators and criteria for the evaluation of quantitative and qualitative effects of groundwater bodies (GWBs) on groundwater dependent terrestrial ecosystems (GDTEs)

According to the WFD (2000) criteria have to be set for GDTEs in order to evaluate the effect of GWB on the ecosystem. In order to maintain favorable ecological status, GDTEs need to maintain groundwater discharge. If a GDTE is in an unfavorable ecological status and it is caused by the pressures on GWBs, then it affects the status of the whole GWB.

Groundwater body can have a negative quantitative and/or qualitative effect on the GDTE:

- Quantitative effect – human influence has lowered groundwater levels, so that the GWB does not provide enough water to sustain the GDTE in its natural state.
- Qualitative effect – human influence has affected the GWB in such a way that its chemical composition causes the deterioration of the ecological value of the GDTE.

3.2.2.1. Indicators and criteria for the assessment of the quantitative effect

The first step to assess whether the contributing GWB could have a negative effect on the GDTE, is to determine the status of the GDTE. If the status is unfavorable, then potential other causes have to be ruled out first. Often the unfavorable status is caused by surface drainage and not by unfavorable changes in groundwater body. In the case of spring fens, the impact of surface drainage is usually smaller than in minerotrophic mires. In spring fens, the effect of drainage is mitigated by the constant inflow of groundwater.

The negative quantitative effect of a GWB on a mire may be expected if there have been no changes in the surrounding surface drainage network and other surface water bodies, but the mire water level has dropped, and discharge from the mire has decreased. It is important to also take into consideration meteorological conditions, which for some period of time may affect mire's water regime considerably.

Groundwater dependent mires are affected by the amount of water in the GWB. Suitable indicators for a potential change are mire water level and annual minimal water level compared to long-term average levels (TABLE 2). For spring fens, the amount of groundwater flow is also an important indicator. It can be assessed by measuring the discharges of surface water bodies (streams) that originate from the fen. The water level dynamics in wells of the GWB that feeds the mire can also be used as an indicator. If the groundwater level drops, then it could result in a decrease of groundwater inflow to the GDTE and the deterioration of its status.

3.2.2.2. Indicators and criteria for the assessment of the qualitative effect

There is no data suggesting that groundwater quality has led to the deterioration of the status of terrestrial ecosystems in Estonia. Generally, the amount of nutrients in spring fens and minerotrophic mires is relatively high. The chemical composition of mire's water is affected by decaying processes and the consequent release of nutrients, assimilation of nutrients by vegetation and outflow through surface water discharge, also by the dilution of mire's water through precipitation.

Threshold values (TVs) for nitrates in GWBs have been developed in the United Kingdom to assess their effect on various types of wetland ecosystems (UKTAG, 2012). In terms of phosphates, the development of TVs is still ongoing. In Estonia, there is not enough relevant data on the hydrochemistry of groundwater dependent mires to develop TVs. The role of nitrates and other substances in groundwater in the functioning of ecosystems and their effects on biota is also unclear. Therefore, thorough hydrochemical and ecological studies have to be conducted if these TVs are to be developed (TABLE 2).

TABLE 2

Criteria for assessing the status of GDTEs

Effect	Terrestrial ecosystems		Comment
	Indicator	Criteria	
Quantitative	**A) Mire water level** A piezometric well is installed to the center of the mire massif, where mire water level is measured.	A) Annual mean or minimal water level compared to long-term average water level.	In order to determine possible long-term effects, changes in biota have to be monitored even before permanent monitoring of water level is started. If the status of the habitat is deteriorating, the nature of the influencer has to be determined. The effect of the drainage system, changes in meteorological conditions have to be assessed. If there are no long-term measurements, then proxy methods have to be used to determine approximate historical water levels (maps, photos, written sources).
	B) Groundwater level at the same location If there is known anthropogenic influence to the GWB (e.g. mine), then groundwater levels should be measured also outside of the mire massif in the direction of the influencer.	B) Annual mean water level compared to long-term average water level.	
Qualitative	**A) Nutrient content** (nitrates, P_{tot}, N_{tot}) both in surface and groundwater	Currently none	Based on current knowledge it is not possible to define criteria for determining the effect of changes of groundwater quality to GDTEs. Relevant studies to produce such criteria have to be carried through.

3.2.3. Evaluation schemes of quantitative and qualitative effects of groundwater bodies on groundwater dependent terrestrial ecosystems

It is not possible to develop a simplistic and universal evaluation scheme that gives a high-reliability answer without the acquisition of additional data. Therefore, the developed schemes enable to pinpoint the ecosystems for which the effect of GWB cannot be ruled out as the cause for the unfavorable status. In these cases, more thorough studies have to be performed to determine the actual effect of the GWB, the size of the effect and suitable mitigation measures.

3.2.3.1. Quantitative effect

The assessment scheme for the potential quantitative effect of a GWB on GDTEs consists of the following steps (FIGURE 3):

1. GDTEs depending on the evaluated GWB have to be determined.
2. National monitoring and inventory programmes for terrestrial ecosystems do not include water level monitoring. Therefore, there is no data on their water level and its dynamics. Generally, the deterioration of the status of mire habitats is caused by a decrease in the mire water level. Therefore, the only possible indicator currently available for determining the potential water level decrease is the deterioration of the conservation status according to the assessment of habitats listed in Annex I of the Habitats Directive (hereafter Annex I habitats). As repeated inventories have been performed only in some mires and permanent monitoring is also carried through only in some areas[2], then it is difficult to find trends in the status of a specific mire. Therefore, the most reasonable solution in this step is to select areas, whose ecological status is less than good (B). These areas move to the next step.
3. The prerequisite for changes caused by groundwater is the occurrence of intensive groundwater abstraction in the vicinity of the GDTE and the subsequent drop in the groundwater head. As the size of the cone of depression depends on the properties of the aquifer, intensity and time since commencing the pumping and depth of the wells, it is impossible to give a universal radius of threat. For the evaluation scheme the following solution is proposed: if in a 10 km radius of the GDTE at least 1000 m^3/d of groundwater is abstracted, then the effect of abstraction on the water level drop in the ecosystem cannot be ruled out. The limits are rather conservative in order to assure that no potential case of the negative effect is ruled out in this step.
4. The next step is to assess if the annual average groundwater level in the aquifer feeding or supporting the GDTE is lower than its long-term average water level. A groundwater level drop in the recharge area of the GDTE will likely cause a drop in the amount of water reaching the ecosystem. On the other hand, a drop in the groundwater level downstream of the GDTE, could cause an increase in the amount of water seeping out from the ecosystem. In both cases, a drop in the ecosystem's water level will probably follow. If the ecosystem does not feed from groundwater, but its water level is kept high because of the pressure from the underlying aquifer, then also, in that case, a drop in groundwater level could cause a drop in the ecosystem's water level (mire water level).
5. If the assessment shows that the water levels in the ecosystem and in the groundwater aquifer that the ecosystem depends on, are lower than they should be, and groundwater abstraction is taking place in the vicinity, then it is possible that the groundwater level drop is caused by the abstraction. To prove or deny it, a preliminary, field-work based study should be carried out, in order to evaluate

[2] Since 2017 permanent monitoring in Estonian mires has been suspended (see Chapter 4.3.4)

whether the water level drop in the ecosystem could be caused by groundwater abstraction and a decrease of the water level in the GWB.

6. If the preliminary study proves that groundwater abstraction and a water level drop in the groundwater body most likely have affected the ecosystem, a thorough study, including field-work and analysis of meteorological data, should be carried out to determine the functional connections between the GDTE and the GWB.
7. After that a monitoring and action plan should be prepared and applied in order to survey possible further changes in the ecosystem's status and to find out means for improving it. That step is especially important for GDTEs situated in nature protection areas of national importance and Natura 2000 areas.

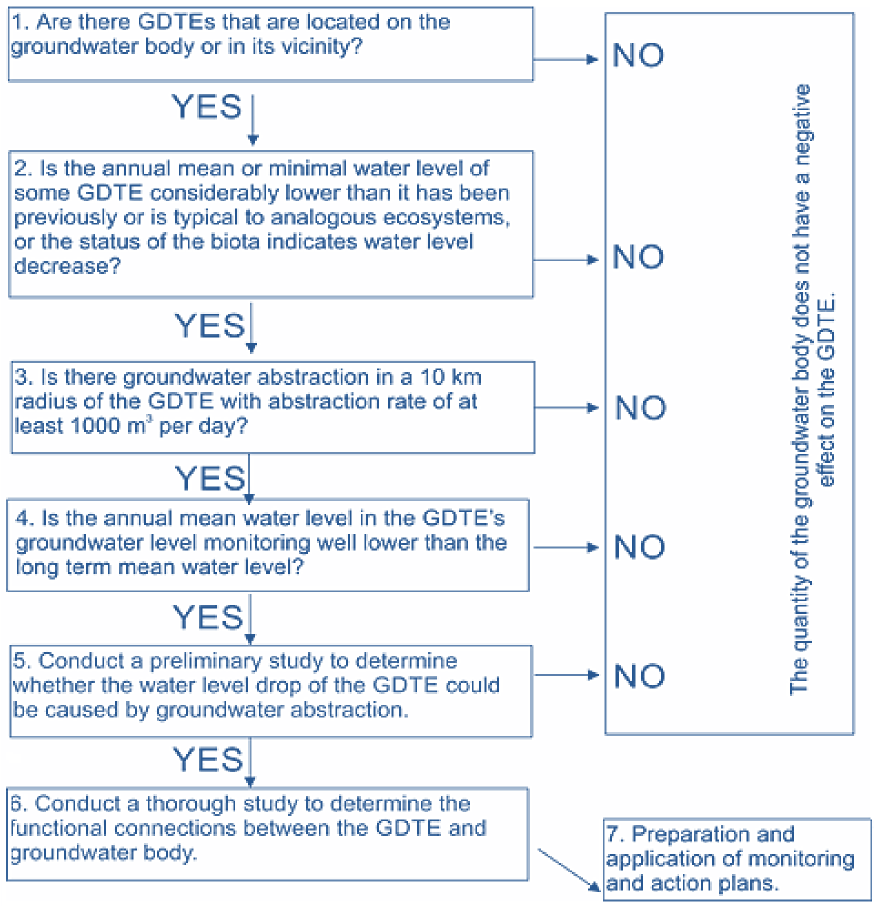


FIGURE 3 **Assessment scheme for the quantitative effect of groundwater on groundwater dependent terrestrial ecosystems**

3.2.3.2. Qualitative effect

The assessment scheme for the potential qualitative effect of a GWB on GDTEs consists of the following steps (FIGURE 4):

1. GDTEs feeding from the evaluated GWB have to be determined. GDTEs that are only supported by the groundwater body are left out in this step.
2. National monitoring and inventory programmes for terrestrial ecosystems do not include water chemistry monitoring. Therefore, there is no data on water and peat chemistry and its dynamics. The only possible indicator currently available for determining the potential negative effect of the water chemistry, is the deterioration of the conservation status according to the assessment of Annex I habitats. As repeated inventories have been performed only in some mires and permanent

monitoring is also carried through only in some areas[3], then it is difficult to find trends in the status of a specific mire. As the quality of GWB cannot have had a negative effect on the ecosystem, if the ecosystem's (Annex I habitat) status is at least "good", therefore the most reasonable solution in this step is to select areas, whose ecological status is less than good (B). These areas move to the next step.

3. The next step is to assess whether the deterioration of the status of the ecosystem could be caused by changes in the water chemistry (increased concentrations of N_{tot} or/and P_{tot}, nitrates etc.). That step is impossible to perform, because of the lack of data. Additionally, there are no TVs for nutrients set for mires and other terrestrial ecosystems. Also, the relations between the chemistry of different types of mires and their status are not clear enough. If such TVs existed, then ecosystems where they are exceeded would move to the next step. But even without the TVs, the ecosystems whose worse than good status could be explained by other effects then changes in groundwater body (drainage, peat extraction, alkaline pollution etc.), may be removed from the assessment in this step.
4. The groundwater quality in the ecosystem's catchment area has to be determined. If the concentration of TotN, TotP, nitrates or other relevant substance in the groundwater monitoring well of the GDTE is two times or higher than the threshold value for the specific ecosystem type (2*TV), then the GDTE moves to the next step. The double threshold value is proposed to take into consideration potential dilution of groundwater, as groundwater might not be the only water source for the GDTE.
5. If it has been determined that the water quality in the ecosystem and in the feeding aquifer, is inadequate, then it is possible that the inadequate quality of the water in the ecosystem is caused by the quality of the groundwater. To prove or deny it, a preliminary, field-work based study should be carried out, in order to evaluate whether the worse than the good status of the ecosystem could be caused by loading from the groundwater.
6. If the preliminary study proves that the water quality of the groundwater most likely has affected the ecosystem, a thorough study, including field-work and hydrochemical analysis should be carried out to determine the functional connections between the GDTE and the GWB.
7. After that, a monitoring and action plan should be prepared and applied in order to survey possible further changes in the ecosystem's status and to find out means for improving it. That step is especially important for GDTEs situated in nature protection areas of national importance and Natura 2000 areas.

[3] Since 2017 permanent monitoring in Estonian mires has been suspended (see Chapter 4.3.4)

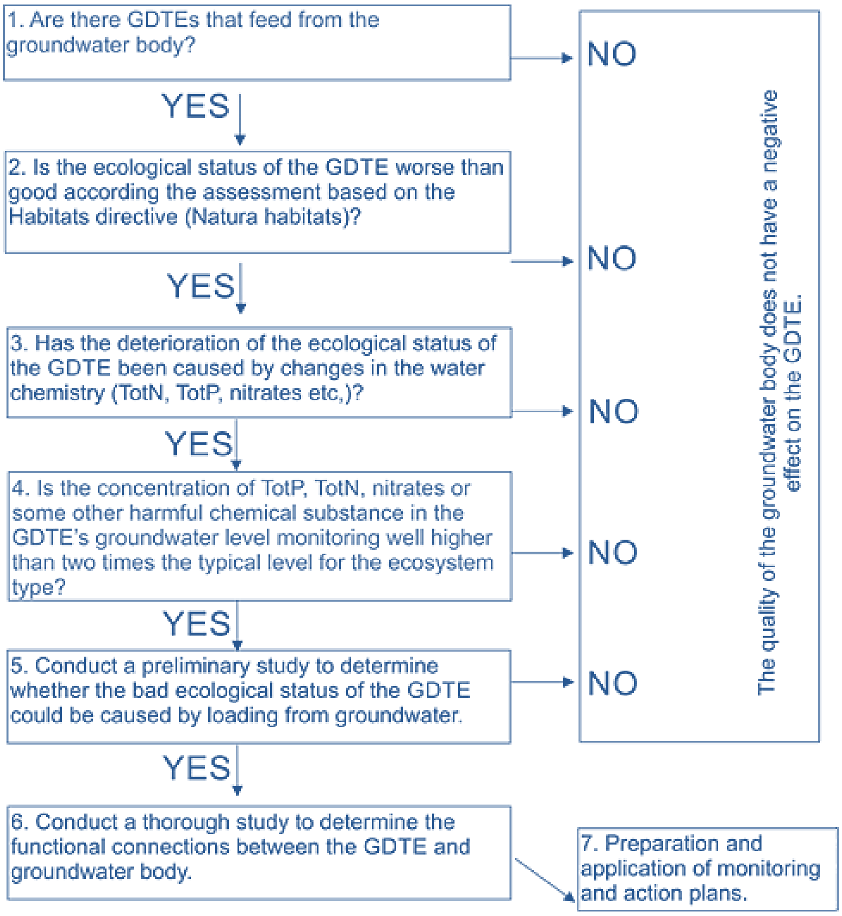


FIGURE 4 **Assessment scheme for the qualitative effect of groundwater on groundwater dependent terrestrial ecosystems**

3.2.4. Application of the evaluation schemes and conceptual models of groundwater dependent terrestrial ecosystems

The assessment schemes were tested on the 70 significant GDTEs in Estonia, but none of them reached the end of the scheme.

In some cases, groundwater abstraction was identified in the vicinity of GDTEs with worse than good status. But because of the lack of water level monitoring wells in the GDTEs and also lack of adjacent groundwater level monitoring wells, the potential effect of groundwater level lowering could not be detected. Still, most probably there are GDTEs in worse than good status because of too low groundwater level.

The qualitative effect could not be assessed at all because there are no habitat-specific thresholds of chemical substances in Estonia. Also, the chemical composition of mire's water has not been monitored. According to the existing knowledge, there are no GDTEs in Estonia whose status is worse than good because of groundwater quality.

Based on the tests, conceptual models were compiled for two mires (Selisoo mire and mire near Kurtna Suurjärv) that are probably dependent on a high-priority GWB and could suffer from anthropogenic influence via the GWB (by the water removal from oil-shale mines and groundwater abstraction from an intake).

3.2.5. Monitoring scheme for groundwater dependent terrestrial ecosystems

Determination of GDTEs and assessing the effects of GWBs on the ecosystems is complicated in Estonia, because the present monitoring networks have not been established with the goal to enable determining interactions between groundwater and surface water. Currently, the parameters monitored from surface water or terrestrial ecosystems are relevant only for these ecosystems and parameters monitored from GWBs are relevant only for groundwater. Also, groundwater monitoring points are located mostly in places that are not representative from the perspective of dependent ecosystems.

Monitoring of the quantitative dependency/effect of GDTEs (mires) should be based on observations of mire water level and their ecological status. Mire water level should be monitored in piezometers equipped with automatic loggers. That data enables to analyze water level dynamics and to compare the data with meteorological data. In addition to the mire water level, also the water level in deeper peat layers and in the underlying Quaternary sediments is a good indicator and should be monitored in piezometers equipped with data loggers. If the water levels in the deeper peat layers or in the mineral sediments are lower than the mire water level, then it is an indication that water is seeping into the underlying groundwater body.

In parallel to mire water level monitoring, the groundwater level in the influencing groundwater body should be monitored as well. That should be preferably done in monitoring wells as close as possible to the mire. In addition to the establishment of the monitoring network, the ecological status of the mire ecosystem has to be determined, based on the methodology developed for Annex I mire habitat types. If the drop in the mire water level has already been determined, the habitat status assessment has to be made regularly, in order to determine the actual effect of water level drop on the ecosystem's status. If possible, these mires should be included in the national monitoring programme for Annex I mire habitats.

The number of mires' water monitoring stations and their spatial placement depends on the specifics of the monitored mire, for example: its size, distribution of different habitat types, the location of pressure(s) etc. For a small and compact area, probably one station would suffice. If the pressure affects the mire through the groundwater body only from one direction, then the monitoring stations should be installed to the side of the mire that is closest to the pressure and to the center of the mire. For larger mires, the monitoring stations should be installed on transect(s) leading from the center of the mire towards the pressure(s).

In the selection process of mires that have to be equipped with monitoring networks for observing the effects of GWBs, areas that have the highest ecological value, are located in nature protection areas of national importance and/or Natura 2000 areas and would presumably suffer most, have to be preferred. These areas include mires that potentially depend on the Ordovician Ida-Viru Oil-Shale Basin groundwater body (Muraka mire complex, Selisoo mire, Puhatu mire's complex, Sirtsi mire). A monitoring network has already been established in Selisoo mire and in Muraka mire's complex. It should also be established to the northern part of Puhatu mire's complex, in order to monitor the effect of water removal from the Narva open-cast mine on the mire ecosystems.

As a second priority a monitoring network should be established in the mires depending on the Quaternary Vasavere groundwater body (mire near Kurtna Suurjärv, mire west from Lake Niinsaare and Lake Nõmmejärv, transitional mires and mire-forests around Lake Konsu). They are smaller and have lower ecological value, but are a part of the Kurtna Kame Field landscape complex, belong to the Kurtna Landscape Conservation area, and partly to the Kurtna Natura area. It is reasonable to develop a common monitoring network for the mires and lakes.

The mires on the Quaternary Männiku-Pelguranna GWB near and in Tallinn are likely affected by its quantitative effect. Therefore, as a third priority, a monitoring network could be established in Männiku bog, where relatively natural mire habitats still exist, and in Pääsküla bog, which has been strongly altered but is under municipal protection.

Other groundwater dependent mires are most likely not significantly affected by groundwater quantity. According to the current knowledge, there are no valuable or protected mires depending on other groundwater bodies that are threatened by groundwater level decrease caused by groundwater abstraction.

The proposed scheme for monitoring the potential effect of groundwater on a GDTE is following:

- Determine the groundwater catchment of the GDTE.
- Establish at least one groundwater monitoring well (level and quality) on the groundwater catchment and preferably also one monitoring well (only level) downstream of the GDTE. If the GDTE is only supported by the GWB, then the well should be established in the direction of the potential pressure.
- Establish monitoring well(s) for mire-water level and quality on the GDTE.
- Measure water quality in the wells four times a year. If the GDTE is not fed by the groundwater body but is just supported by it, the water quality measurements are not needed.
- Measure the water level in the wells using automatic sensors every 3 hours or manually at least once a month.

3.2.6. Conclusions

- Based on existing data, a total of 71 mires, mire complexes or groups of mires were determined that probably depend on GWBs. 26 of these depend on GWBs that had the highest priority according to the project scope statement (in northeastern Estonia and in nitrate vulnerable zones according to Nitrates Directive (1991)). A map layer and attribute table for GDTEs were compiled.
- Criteria for evaluating the effect of GWBs on the mires were developed. In the case of the quantitative effect of the GWBs, the criteria are annual average or annual minimum mire water level compared to long-term average mire water level and annual average groundwater level compared to long-term average groundwater level. In the case of qualitative effect, there are no monitoring data of mires' water chemistry, also the relations between the status of various mire types and their hydrochemistry are unknown. According to currently available data and the level of knowledge, there are no terrestrial ecosystems in Estonia that are in a bad status because of groundwater quality.
- Methodology and evaluation schemes for assessing the potential quantitative and qualitative effect of groundwater bodies to GDTEs were developed.
- Tests on the potential effect of groundwater bodies on GDTEs were performed as far as there was available data.
- Conceptual models were compiled for two mires (Selisoo mire and mire near Kurtna Suurjärv) that are probably dependent on a high-priority groundwater body and could suffer from anthropogenic influence via the GWB (by the water removal from oil-shale mines and groundwater abstraction from an intake).
- Principles for monitoring schemes for GDTEs were given.

3.3. Delineation of threshold values for groundwater dependent terrestrial ecosystems in United Kingdom

Threshold values (TVs) are needed to determine the limits for pollutants which, if exceeded, would indicate a pressure that could be or is causing damage to GDTE. A combination of damaged GDTE (such as the failure to meet conservation objectives) and exceedance of the relevant TVs may require further investigations to confirm whether damage has occurred, to define hydrogeological and hydrochemical pathways between GWB and GDTE and finally assess the status of GWB.

UKTAG (2012) have derived GDTE TVs for British wetlands and sets out how these values should be improved in the future and what kind of information and knowledge is lacking. At the beginning it was proposed to develop TVs for nitrates and phosphates as the macronutrients N and P are the major chemical pollutants that impact GDTEs, and nutrient pressures are the most widespread pressures in UK GWBs. Standards for other pollutants may be developed in the future as more information (monitoring data) becomes available.

3.3.1. Nutrients as a risk for GDTEs damage

In general, higher nature conservation value is given to wetlands characteristic of low nutrient settings as they tend to be associated with a high species diversity per unit area and support more rare species that wetlands in higher nutrient settings.

Changing nutrient conditions may change the relative competitive ability of individual plant species and can result in degradation or complete loss of high value species and communities; may change plant communities within GDTE; may increase dominance of particular plant species that are responsive to elevated levels of nutrients (for example, common reed, nettle) and may change the structure of particular plant communities (such as reedbeds) that affect their function as a habitat for birds or insects.

Nutrients can enter GDTEs by various pathways: aerial deposition, surface water (during flooding) and through lateral water movement in the soil, groundwater, direct deposition (such as spreading of fertilizer) and re-mineralized nutrients within GDTE.

Main factors affecting the sensitivity of wetlands to nutrients are: vegetation management (cutting and removal of biomass will remove nutrients); water management; reversibility impact (it is very costly and time consuming to revert ecosystem's initial status); nutrient immobilization (for example, phosphate immobilization by iron and calcium).

3.3.2. Nutrient related wetland categories

Different wetland types will have different dominant water sources and thus different exposure to nutrients. For example, wetlands fed by regular over-topping of rivers in comparatively fertile lowland situations tend to support productive vegetation dominated by a relatively few nutrient responsive species. Groundwater fed wetlands are less likely to be exposed to high nutrient pressure and will be characterized by less productive vegetation. Finally, predominantly by precipitation fed systems (such as bogs) are not naturally exposed to elevated nutrient concentrations thus will tend to be highly infertile and acidic.

UKTAG (2012) groups wetlands into 11 broad categories that take into account trophic classes and ecology according to the following criteria:

- Dominant water source (precipitation/ surface and groundwater);
- Landscape setting (valley bottom, slope etc.);
- Intrinsic wetland sensitivity (based on expert judgement).

The categories are: 1) quaking bogs; 2. wet dunes; 3. fens (mesotrophic) and fen meadows; 4. fens (oligotrophic) and wetlands at tufa forming springs; 5. wet grasslands; 6. wet heaths; 7. peatbogs and woodlands on peatbogs; 8. wetlands directly irrigated by springs or seepage; 9. swamps (mesotrophic) and reedbeds; 10. swamp (oligotrophic); 11. wet woodlands.

These categories are adapted from a functional wetland typology developed for Scotland (SNIFFER, 2009) and relate to the more detailed National Vegetation Classification and wider Natura 2000 feature types. The resulting wetland classes are still quite broad and it is therefore possible that the same category can contain a different set of species or species dominance at different altitudes.

3.3.3. Factors affecting the relationship between nutrient and wetland conditions

Large variability around the mean nitrates or phosphates concentrations and its associated condition for each vegetation category was observed. Initial analysis of such a large variance pointed out differences between countries, whereby nitrate values in GDTEs of the same type were often much lower in Scotland and Wales than they were in England.

It was decided to study if the variability was due to catchment related factors such as catchment size, land-use and geology. As the possible reasons for a variance were identified: dominant land use (arable, woodland, grassland, heath, urban, coastal); mean altitude of the wetland above ordnance datum (above or below 175 m); catchment geology (using WFD generic geology types; categories: organic, calcareous, siliceous and salt); catchment size (using WFD generic catchment size types; categories (extra small, small, medium and large)); mean annual recharge to the groundwater (used categories: low (0-200 mm/annum); medium (200-400 mm/annum); high >400 mm/annum).

Several reasons were pointed out why some GDTEs were observed to have a good status, though they were fed by groundwater containing nutrient concentrations that looked elevated at first sight:

- Wetlands at lower altitudes have changed due to historic nutrient changes;
- The wetland categories used were broad and included a range of National Vegetation Classification communities within the categories;
- Groundwater nitrate and phosphate concentrations were lowered in the pathway from aquifer to wetland in situ groundwater due to chemical processes, and the potential GWB pressure had therefore not been transferred to the wetland vegetation;
- Phosphate may be limiting plant growth. Probably GDTEs in the lowlands that experience elevated nitrate concentrations in the groundwater feeding the site were not ecologically impacted because the low phosphate concentrations curtail growth and restrict ecological change;
- Probably some wetlands under elevated nitrate concentrations were already damaged or at least approaching a point where ecological enrichment would become apparent, but this was not yet recognized in the site condition assessment.

3.3.4. Development of threshold values for groundwater dependent terrestrial ecosystems

Threshold values were developed using three sources of information: 1) empirical correlation between wetland condition and chemistry data for hydrogeologically linked GWBs across the UK; 2) site specific investigations and 3) other published databases and literature. Step by step delineation approach is described in UKTAG (2012).

TABLE 3

Proposed groundwater nitrate (NO_3 mg/l) threshold values (UKTAG, 2012)

GDTE category	Altitude		
	<175m AOD	>175m AOD	Any altitude
Quaking bog	18	4	
Wet dune			13
Fen (mesotrophic) and fen meadow	22	9	
Fen (oligotrophic and wetlands at tufa forming springs)	20	4	
Wet grassland	26	9	
Wet heath	13	9	
Peatbog and woodland on peatbog			9
Wetland directly irrigated by spring or seepage			9
Swamp (mesotrophic) and reedbed			22
Swamp (oligotrophic)			18
Wet woodland	22	9	

The analysis of phosphate concentrations was more problematic due to low variability of phosphate concentrations between wetlands in good and poor conditions and high number of values under detection limit. The main factors or their combination leading to such results are: the wetland ecology might be predominantly impacted by nitrate coming from the groundwater, whereas phosphates are more likely to be transported through surface waters and the wetlands investigated may be nitrogen limited.

4. Development of joint methodology for identification and assessment of groundwater dependent terrestrial ecosystems in Gauja-Koiva river basin

4.1. Groundwater monitoring in Estonia and Latvia

4.1.1. Groundwater monitoring in Estonia

In Estonia regular observations of groundwater regime were started in 1946, the planned development of a hydrogeological observation network began in 1960. The general coordinator of the monitoring programme is the Ministry of the Environment in cooperation with the Environmental Board, the Environmental Inspectorate, the Environment Agency and the Environmental Investment Centre.

The water monitoring programme prepared for the second period (2016–2021)[4] of the river basin district management plans helps to monitor the achievement of environmental objectives, assess the impact of human activity and gain reliable data on the actual environmental status of water bodies. The water monitoring programme is an activity plan prepared in accordance with the WFD establishing a framework for Community action in the field of water policy (*WFD*) article 8, Annex 5, and the requirements from the legislation Water Act and Minister of the Environment regulation no 49 Sampling methods. Monitoring is used to evaluate the chemical and quantitative status of groundwater. The monitoring programme provides an overview of the methods used for planning monitoring and the planned monitoring locations, monitoring frequencies and indicators to be monitored.

4.1.1.1. Monitoring of the quantitative status of groundwater bodies

The number and location of monitoring points for the quantitative status of groundwater bodies is chosen in the following manner:

1) it allows to consider the long- and short-term changes in the recharge of groundwater bodies while measuring the groundwater body water levels;
2) the number and location of monitoring points for the quantitative status of groundwater bodies at risk must additionally allow to evaluate the impact of water abstraction and water discharge on the groundwater levels;
3) the number and location of monitoring points for the quantitative status of transboundary groundwater bodies must additionally allow to evaluate the direction and quantity of groundwater flow across the state border.

On the basis of the abovementioned, the spatial density of the groundwater bodies' monitoring network depends most directly on the level of human impact on groundwater (water abstraction, intensity and nature of industrial activity, including mining and agriculture) in different regions.

For this reason, a denser network of groundwater monitoring wells has been designated for the Ordovician Ida-Viru and the Ordovician Ida-Viru oil-shale basin groundwater bodies, where the pressure on groundwater is the highest due to oil shale mining. A relatively dense monitoring network is also designated for the Cambrian-Vendian type groundwater bodies, as these are subject to the most intense consumption of groundwater which have caused the formation of extensive cones of depression in groundwater. The cones of depression may increase the risk of saltwater intrusion from the Baltic Sea or from the underlying crystalline bedrock.

The quantitative status of the Cambrian-Vendian, the Cambrian-Vendian Gdov, the Cambrian-Vendian Voronka and the Ordovician-Cambrian groundwater bodies from the West-Estonian river basin, the

[4] http://www.envir.ee/et/vesikondade-veeseireprogramm-2016-2021

Ordovician-Cambrian groundwater body from the East-Estonian river basin, the Silurian-Ordovician Pärnu and the Quaternary Vasavere body are at risk because of groundwater overdraft. Therefore, the changes in the groundwater levels of these layers need to be monitored meticulously to avoid saltwater intrusion or deterioration of GWB's quantitative status conditioned by the widely stretched cone of depression due to groundwater overdraft. This allows us to implement timely measures to maintain a good quantitative status of the groundwater body and to avoid overdraft of groundwater.

If the good status of a GWB or a group of GWBs is not at risk and the GWBs have similar hydrogeological conditions, the monitoring of the quantitative status of groundwater may be optimized. This means that not all GWBs need to have at least three monitoring points. This is why the East-Estonian river basin's Quaternary aquifers (except for the Quaternary Vasavere and Quaternary Meltsiveski groundwater bodies which are in a bad state) have one bore well each for the monitoring of quantitative status. The GWBs of Quaternary Prangli, Middle and Lower Devonian Kihnu and Middle and Lower Devonian Ruhnu located on islands also have one bore well each for monitoring the quantitative status of groundwater.

In general, the monitoring wells were chosen from the existing ones. The reliability of evaluating the status of groundwater bodies with the existing monitoring network was evaluated, considering the heterogeneity of groundwater bodies and spread of monitoring points. As a result, it was decided to restore some of the monitoring wells conserved during previous years in areas with insufficient coverage and add additional monitoring wells. The vertical coverage of groundwater bodies is ensured with groups of monitoring wells with different open intervals.

The monitoring of the quantitative status of groundwater bodies includes measuring the water levels of groundwater bodies and, if necessary, the quantity of water flow in springs and gaining watercourses.

For monitoring of groundwater bodies quantitative status there are 247 monitoring wells where water level data is gathered with the frequency of 1 time per year, 1 time per month (manual measurements) or with automatic sensors 8 times per day (in 163 monitoring wells). To ensure that the water level data collected during the monitoring of the quantitative status of groundwater bodies is comparable with meteorological measurements, the automatic sensors are set to measure simultaneously in every three hours.

4.1.1.2. Monitoring of the chemical status of groundwater bodies

The electrical conductivity of groundwater and its chemical composition need to be measured for the evaluation of the chemical status of groundwater. Before taking samples, water needs to be pumped out of the bore well at least until the values of field parameters (pH, electrical conductivity, dissolved oxygen concentration, temperature) have stabilized. This ensures that samples are taken from groundwater and not from the water that has remained in the well. However, it is desirable that at least 4-6 well volumes are pumped out of the observation well to ensure that sampled water originates from the aquifer and not from stagnant water in the well casing. According to the regulation No.49 of the Minister of the Environment "Sampling methods"[5], the following measurements must be taken on site, together with water samples: water temperature; contents of dissolved gases, e.g. oxygen; electrical conductivity; pH, i.e. the negative logarithm of hydrogen ion concentration.

The network for monitoring the chemical status of groundwater is prepared in a way that would allow a reliable evaluation of the chemical status of each groundwater body. The monitoring network must also allow to describe the natural and anthropogenic changes in the chemical composition of groundwater and the significant and constant changes in pollutant concentrations, and to evaluate the achievement of environmental objectives for areas that are dependent on groundwater and need protection.

[5] https://www.riigiteataja.ee/akt/108102019001

Furthermore, the monitoring points must be evenly spread across the groundwater body vertically as well as horizontally. The evaluation of the chemical status of groundwater depth-wise has been ensured in Estonia with groups of monitoring wells with different open intervals. The representability of monitoring points is ensured by choosing an appropriate open interval at an observation point typical of the groundwater body, i.e. with the bore well construction and water abstraction. The layouts of monitoring points must consider the size of the area that an observation point is representative of. Local sources of pollution (point source pollution) need individual monitoring.

Water samples for monitoring the chemical status of groundwater bodies are taken during the summer minimum period of groundwater levels in shallow groundwater bodies and at any time from the confined groundwater bodies formed in the Cambrian-Vendian, Ordovician-Cambrian, Lower-Middle Devonian and Silurian GWBs underlying the Lower-Middle Devonian GWBs.

According to the recommendations of the European Commission, the following elements are considered as the main elements of groundwater, which allow to designate the hydrogeochemical class of groundwater and check the quality of monitoring analysis: Na, K, Ca, Mg, Cl, SO_4, NO_3, HCO_3, Fe, Mn, NH_4. Changes in the hydrogeochemical type of groundwater is the first indicator for changes in groundwater caused by either anthropogenic or natural factors. The abovementioned ions also allow us to check whether the analysis results of water samples are reliable. This is why these indicators are monitored in each and every groundwater body.

In each groundwater body, the following data is collected at least once per year: electrical conductivity, pH, dissolved oxygen concentration (O_2), hardness, chemical oxygen demand on the basis on Mn (COD-Mn), concentration of ammonium ions (NH_4^+), nitrate ions (NO_3^-), chloride ions (Cl^-) sulphate ions (SO_4^{2}-), macro-components characteristic of the chemical composition of groundwater (HCO_3^-, Ca^{2+}, Mg^{2+}, Na^+ and Fe), which are necessary for establishing the chemical type of groundwater. Cl^-- and Na^+-ions serve as indicators of saltwater intrusion and increase of electrical conductivity.

Annex I of the Groundwater Directive establishes groundwater body threshold values for nitrates (50 mg/l) and the concentration of pesticides, including their metabolites, degradation and reaction products (0.1 µg/l and 0.5 µg/l in total) applied across the union. The concentrations of these pollutants need to be monitored in all groundwater bodies.

According to Annex I of the groundwater directive, the concentration of pollutants in groundwater bodies formed from the deeper well protected layers of groundwater, may be measured once in every two water management plan periods (12 years).

In Estonia, the threshold values for monophenols, petroleum products, benzene and sum of PAHs have been designated on the basis of pollutant concentrations in some groundwater bodies. Therefore, water samples must be taken from these groundwater bodies to measure the concentrations of these pollutants.

Pursuant to Part B of Annex II of the Groundwater Directive, at least once during the duration of a water management plan, data about the following pollutants must also be collected from each groundwater body: arsenic, cadmium, lead and mercury, and also trichloroethylene, tetrachloroethylene and other synthetic substances.

For monitoring of groundwater bodies qualitative status there are 227 monitoring wells where water chemical analysis is gathered with the frequency of 1 time per year.

4.1.1.3. Additional monitoring of groundwater bodies in nitrate-sensitive areas

Additional monitoring of groundwater bodies in nitrate-sensitive areas is carried out in the main monitoring points (in 53 locations) as well as additional monitoring points (in 72 locations) which differ in terms of monitoring frequencies. Koiva water basin is not considered as a nitrate-sensitive area.

When taking groundwater samples temperature (t°), dissolved oxygen concentration (O_2), pH and electrical conductivity is measured on site. In nitrate-sensitive areas the monitoring frequency for nitrate compounds (NH_4 and NO_3) is 1-4 times per year, for phosphates (PO_4) 1-2 times per year and for pesticides 1-2 times per one water management plan cycle (6 years). In 8 main monitoring points petroleum products, benzene and sum of PAHs analysis are taken once per one water management plan cycle. The concentrations of other relevant chemical substances are measured once per year for the main and additional monitoring points.

The concentration of pesticides, pesticide ingredients, including their metabolites, degradation and reaction products is measured every year in relevant main or additional monitoring points for groundwater bodies located in nitrate-sensitive areas.

To ensure the reliability of the monitoring data, the parties fulfilling their contractual obligations are required to hold a certificate of attestation for taking water samples in order to guarantee that water samples are taken as required. All water samples must be analyzed in a laboratory which has accreditation for the analysis of the given water type and indicators. A quality system like this shall guarantee the reliability of data from the moment of taking the samples till the analysis results of data, as the attestation and accreditation system shall guarantee control and supervision over the entire data supply chain.

Groundwater monitoring in the Koiva River Basin District is carried out in 7 wells. Of these, groundwater quality observations are made in 5 wells, while groundwater quantity is monitored in 2 wells (ANNEX 3).

4.1.2. Groundwater monitoring in Latvia

Regular surveys of groundwater quality in Latvia have been conducted since 1959. The objectives and scope of the groundwater monitoring network varied over time, mainly due to changes in regulatory documents as well as global trends in groundwater monitoring. The groundwater monitoring network was mainly set up between 1959 and 1991, initially to assess the water quality of deep pressurized aquifers and their changes, as these aquifers began to be used intensively for centralized drinking water supply not only in cities during this period, but also in populated rural areas. Gradually, it was supplemented by the addition of "level principle" monitoring stations, which consist of well-placed boreholes with filters at various intervals up to a depth of 200-400 meters, and the installation of "balance stations" with shallow boreholes. From 2004, the groundwater monitoring network also includes springs. This is an important improvement in the monitoring network, as springs with high water flow mostly represent water quality in much larger catchment areas than wells and are an important indicator of diffuse pollution.

In Latvia, groundwater monitoring simultaneously fulfills the requirements of the Nitrates Directive and the Water Framework Directive (WFD, 2000/60/EC). The primary objective of nitrate monitoring in Latvia is to detect any nitrate contamination to ensure good drinking water quality throughout the country, as well as to reduce the impact of nitrate pollution on small and large rivers whose waters flow into the Baltic Sea. In the current reporting period (report of the Nitrates Directive 2012-2015), 68% of all monitoring observation points represent confined aquifers which in Latvian hydrogeological conditions are less vulnerable to nitrates pollution compared to shallow Quaternary and fractured aquifers. Thus, regarding the Nitrates Directive, sampling of shallow wells and springs should be the priority and deeper wells could be monitored in cases when pollution is found in the upper aquifers.

Under the constraints of limited funding, the groundwater monitoring programme is being adapted to the requirements of the two directives (Nitrates Directive and WFD), which are not equivalent. The WFD requires the identification of the background level of natural chemical composition and trends in the aquifers used in the main water supply of underground water bodies, which in the case of Latvia are deeper confined water. The current groundwater monitoring programme is more adapted to fulfill the requirements of the WFD than to fulfill the requirements of the Nitrates Directive.

Groundwater monitoring provides data on the status of the groundwater body. It is the main and strategic goal of monitoring in any year of the monitoring programme period. Achieving good groundwater status in all groundwater bodies and assessing the risk of failing to achieve this goal is the main objective of groundwater resource management. Groundwater monitoring is primarily done at groundwater body level, while integrating river basin management into a common strategy for achieving environmental quality objectives.

Groundwater status within the monitoring network is observed in 311 wells at 61 stations and 30 springs. Of these, quality (chemical composition) observations are provided at 53 stations in 218 wells and 30 springs, while quantitative (water levels) observations at 60 stations in 305 wells. Accordingly, groundwater monitoring in the Gauja River Basin District is carried out at 6 stations in 23 wells and 12 springs. Of these, groundwater quality observations are made in 21 wells and 12 springs, while underground water quantity is monitored in 23 wells (ANNEX 3).

The frequency of monitoring observations during the six-year cycle of monitoring of river basin districts includes a detailed breakdown of monitoring stations by groundwater bodies and types of monitoring. Monitoring points that are monitored each year and observable parameters for groundwater quality can vary according to the annual monitoring plans developed. The frequency of groundwater monitoring is variable: the frequency of quantitative observations – two times a day (automatic level measurements) up to four times a year, and the frequency of groundwater chemical observations is four times a year, up to once a year (over a six-year period, it changes from one time in six years to one time each year).

Groundwater quality monitoring parameters are traditional field measurements, key ions, nitrogen compounds and their ionic forms, heavy metals, chemical pollutants, and pesticides (atrazine, simazine, bentazone, MCPA, promethrin, propazine, 2,4-D, MCPB, isoproturon, aclonifene, bifenox, aldrin, dieldrin, heptachlor, heptachlor epoxide, dimethoate, cypermethrin, alpha-cypermethrin, trifluralin). Chemical pollutants are monitored only in urban areas, artificially replenishing groundwater resources, as well as in the areas of water extraction and associated potential cones of depression. Pesticides are monitored in agricultural areas, including monitoring stations that coincide with the nitrate vulnerable zone. Primary pesticides are analyzed in the first sensitive and unprotected lower-lying aquifer, and in the case of pesticide detection, the deepest aquifers are sampled. Other observable parameters are observed at all monitoring points (both wells and springs).

The frequency of monitoring observations and its determination in the following years may change, taking into account the new monitoring data obtained, experience gained, developed scientific projects in connection with the implementation of the WFD, and new requirements of EU and regulatory enactments in the Republic of Latvia. This will be assessed by developing a monitoring plan for each specific year.

4.1.3. Comparison of groundwater chemical monitoring in Estonia and Latvia

In general, Estonian and Latvian groundwater monitoring programmes are quite similar, but there are some differences in the observed water quality indicators (TABLE 4). The main difference is that in Estonia until now, total phosphorus and nitrogen are not measured in the groundwater. This is also problematic from the GDE (groundwater dependent ecosystems) point of view because in surface water monitoring, those are one of the main parameters for deciding the ecological status of the habitat (Terasmaa et al., 2020). Marandi et al., (2019) have suggested that threshold values of total phosphorus and nitrogen should be established in the GWBs strongly influencing the GDEs in the area and that these parameters should be gradually integrated into the existing monitoring strategy.

TABLE 4

The aggregated list of observed water quality indicators of GWBs in Estonia (EE) and Latvia (LV) (Terasmaa et al., 2020)

Parameters	LV	EE	Parameters	LV	EE
Traditional measurements			**Nitrogen compounds and their ionic forms**		
Temperature	Yes	Yes	NH_4^+, mg/l	Yes	Yes
Conductivity 20^{0}C, µS/cm	Yes	Yes	NO_2^-, mg/l	Yes	Yes
pH index	Yes	Yes	NO_3^-, mg/l	Yes	Yes
Eh, mV	Yes	Yes	$N_{tot.}$, mg/l	Yes	No
$Fe_{tot.}$, mg/l	Yes	Yes	TOC, mg C/l	Yes	No
O_2 dissolved, mg/l	Yes	Yes	DOC, mg C/l	Yes	No
Key ions			UV absorption, cm^{-1}	Yes	No
Na^+, mg/l	Yes	Yes	Permanganate index, mg/l	Yes	Yes
K^+, mg/l	Yes	Yes	**Heavy metals**		
Ca^{2+}, mg/l	Yes	Yes	Cd, µg/l	Yes	Yes
Mg^{2+}, mg/l	Yes	Yes	Pb, µg/l	Yes	Yes
Cl^-, mg/l	Yes	Yes	Ni, µg/l	Yes	No
SO_4^{2-}, mg/l	Yes	Yes	Hg, µg/l	Yes	Yes
HCO_3^-, mg/l	Yes	Yes	As, µg/l	Yes	Yes
Mn, µg/l	Yes	No	**Chemical pollutants**		
$P_{tot.}$, mg P/l	Yes	No	Trichlorethylene, µg/l	Yes	Yes
PO_4^{3-}, mg/l	Yes	Yes	Tetrachloroethylene, µg/l	Yes	Yes
Total hardness, mmol/l	Yes	Yes	Trichloromethane, µg/l	Yes	Yes
			1,2-dichloroethane, µg/l	Yes	Yes
			BTEX, µg/l	Yes	Yes
			Pesticides	Yes	Yes

4.2. Groundwater assessment in Estonia and Latvia

Groundwater body (GWB) is a management unit established in the Water Framework Directive (WFD) to ensure good groundwater status and to protect groundwater resources and associated ecosystems from further deterioration. Due to large heterogeneity of European aquifers, there is no unified methodology on how to delineate GWBs. The approaches vary between Member States and often also between neighboring countries with similar hydrogeological conditions. According to WFD, common principles are needed to manage groundwater resources in transboundary areas. Within a transboundary river basin, water management activities should be jointly coordinated for the whole of the river basin district between involved Member States.

At the beginning of WFD implementation, it was encouraged to harmonize approaches how each Member State delineate GWBs. However, the harmonization process could lead to not only changed boundaries of GWBs, but also to changes in long-term monitoring networks, which consequently would increase the need for extra investments in new monitoring stations and review of whole RBMPs. Now Member States do not plan to harmonize GWBs, but only exchange GWB delineation strategies to take that into account when analyzing GWB status differences in transboundary areas.

This section contains a summary of the implementation of groundwater monitoring in Estonia and Latvia, paying more attention to monitoring of groundwater quality (chemical composition). The aggregated list of observable parameters for groundwater quality between countries is attached in ANNEX 3.

Currently there are 31 GWBs delineated in Estonia (FIGURE 6). From these 8 GWBs are at risk (FIGURE 5). In Estonia the GWB status assessments have been made in 2014 (OÜ Hartal projekt, Türk) and in 2015 (Perens et al). In the Gauja-Koiva river basin there are 2 GWBs that have a good status based on both quantity and chemical composition (FIGURE 5). At the moment (2019-2020) there is in progress a new GWB status assessment report that is prepared by GSE as conducted by Estonian Environmental Agency.

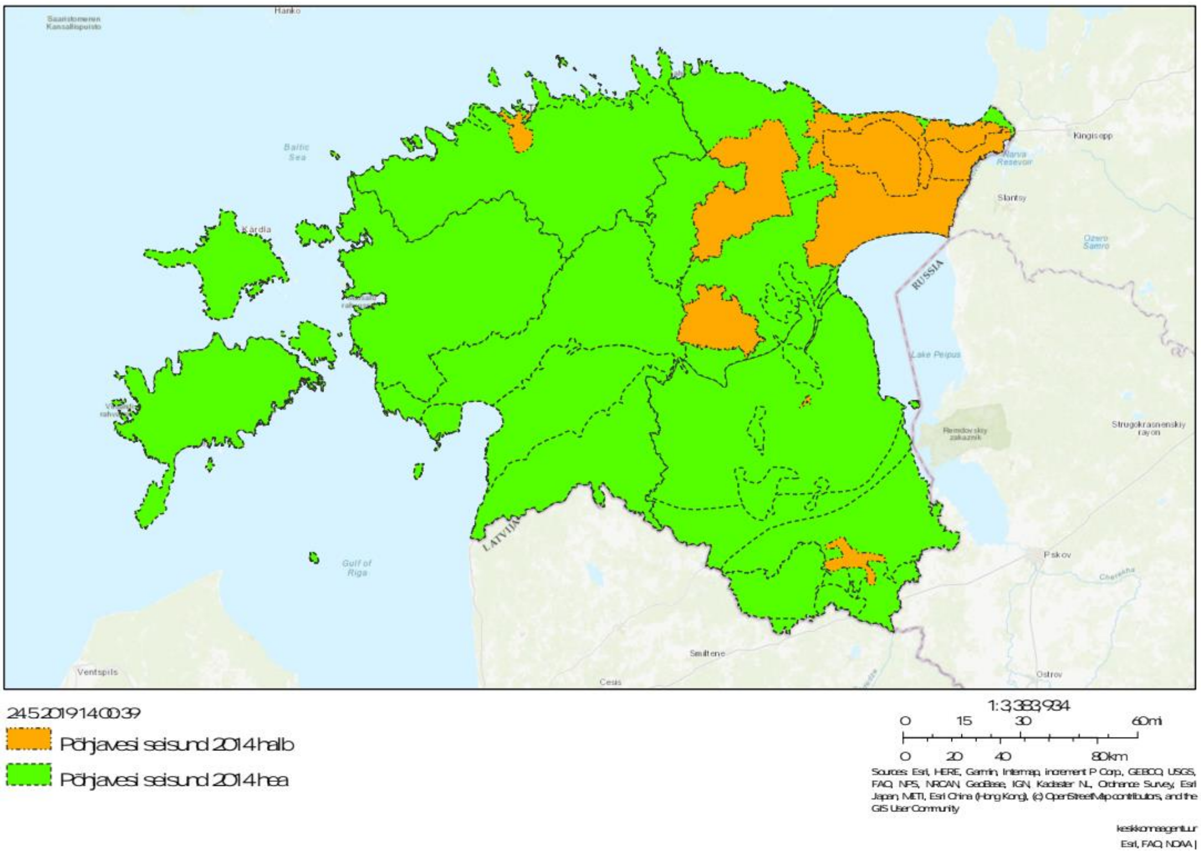


FIGURE 5 **Status of groundwater bodies in Estonia, based on 2014 assessment (orange color marks GWBs in bad status)**

In Latvia, the status assessment of the GWBs was updated in 2015, taking into account the old boundaries of GWBs (previously 16 GWBs have been identified). Based on this assessment, all GWBs were in good quantitative status and three regional areas within those GWBs (which were not identified as separate GWBs but as part of the respective Pos) were in poor qualitative status.

In 2018, the boundaries of the GWBs were reviewed and a total of 25 GWBs were delineated, of which 3 GWBs were assessed being in poor quality status (GWBs at risk) and the remaining 22 GWBs will be assessed within the 3rd period River Basin Management Plan cycle (2022-2027). Currently, there is only one GWB at risk in the Gauja-Koiva river basin, where poor quality status is noted (see Chapter 4.2.2, FIGURE 10).

4.2.1. Groundwater body delineation approaches in Estonia

The methodology used in the delineation of Estonian GWBs is based on the guidance documents by European Commission on the common implementation strategy of the WFD and Groundwater Directive. A detailed description about the Estonian delineated GWBs and their boundaries can be found in a GSE published report Perens et al (2012). In 2018 and 2019 GSE, as commissioned by the Ministry of the Environment, synthesized information collected on Estonian GWBs and updated their conceptual models according to the new dataset (Marandi et al., 2019).

In general, the delineation of GWBs in Estonia was carried out by taking into account the following general criteria: natural baseline quality of groundwater, hydrogeological parameters of aquifer-forming rocks, the extent of vertical water exchange between different aquifers, possible anthropogenic pressures and socio-economical aspects (Perens et al. 2012). The delineation of the GWBs was carried out by combining the borders of the existing hydrostratigraphic units (aquifer systems), the coastline, Estonian land border and the limits of river basin districts in Estonia (*Ibid.*). In deeper aquifer systems the chloride concentration of 350 mg/L was chosen as a limit to the extent of GWBs due to the fact that more saline waters do not comply with drinking water quality standards (*Ibid.*).

GWBs in Estonia are delimited in three dimensions, which means determining the aquifers overlying and underlying strata, surface elevations and the overall thickness of the groundwater body. The three-dimensional nature has been achieved by creating cross-sections for each GWB.

In most cases, the aquifers that form the GWB are separated by aquitards. In the absence of an aquitard between different GWBs, the boundary between them is defined by different lithological composition and the resulting water type differences. Due to the different nature of groundwater flow, a distinction is made between porous aquifers, where groundwater flows mainly through the pores of granular rocks (sandstones), and fissured-karstic aquifers (carbonate rocks), where groundwater flows along fissures and interconnected crevices. This different behavior of the hydrogeological system also results in different measurement frequencies for groundwater monitoring and determines the need to distinguish GWBs composed of carbonate and siliciclastic rocks in the hydrogeological section. As the geological setting of Estonia is characterized by wide lateral distribution of different bedrock formations and similar hydrogeologic conditions in aquifers in different parts of the country, the GWBs comprising bedrock aquifers have quite a wide lateral extent (FIGURE 6, FIGURE 7, FIGURE 8).

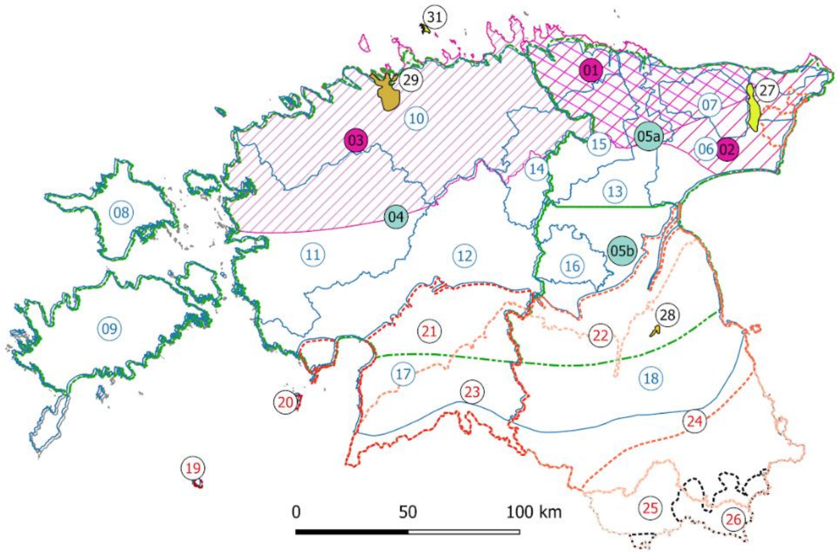
From the Estonian legislation Water Act[6] it is predefined that GWB has to be delineated if one of the following conditions is met: (1) a groundwater resource allocation has been calculated and approved for a part off or for the whole of the aquifer; (2) the aquifer supplies drinking water to at least 50 people; (3) at least 10 m^3 of water per day can be abstracted from the aquifer at present or are planned to be abstracted in the future; (4) the natural chemical composition of groundwater is such that it can be used for drinking water. Marandi et al. (2019) have suggested that due to a large number of GWBs that would have to be established (mostly in Quaternary aquifers of local extent) when these conditions are followed by the letter, the legislations should be changed so that the GWBs can be determined if one of the above conditions is met but there is no obligation for it.

Among many smaller changes to the existing GWB network in Estonia, Marandi et al. (2019) suggested two general ones to the initial GWB delineation developed by Perens et al. (2012). The first concerns the deep Ordovician-Cambrian GWB in the East Estonian river basin district. It was recommended that this GWB should be split up into two separate GWBs due to different anthropogenic pressures affecting different areas underlain by the GWB (potential effects of current oil shale and future phosphorite mining in north-eastern Estonia and groundwater abstraction in south-eastern Estonia).

The second general change to the GWB delineation is broader and concerns the Quaternary aquifers in Estonia. In the delineation developed by Perens et al., (2012), small 13 Quaternary GWBs were established based on the areas where groundwater from the Quaternary aquifers forms an important source of water supply. Other Quaternary aquifers were not considered as part of the GWB framework in Estonia. However, as Quaternary aquifers all over the country can affect the formation of groundwater quality, the infiltration rates and the transmission of anthropogenic pollution from the land surface to the subsurface, it was suggested that all Quaternary aquifers should be considered as a part of the Estonian GWB framework. The

[6] https://www.riigiteataja.ee/en/eli/527122019007/consolide

delineation of all Quaternary aquifers with a potential for water abstraction or having an important influence to the groundwater dependent ecosystems as separate GWBs would have led to formation of hundreds or even thousands of new Quaternary aquifers, which would not have been administratively manageable. Thus, a simpler approach was implemented. All Quaternary aquifers were joined with the underlying bedrock GWB. This approach is justifiable, because in most cases the Quaternary aquifers form a unified hydrogeological system with underlying bedrock aquifers and have the same anthropogenic pressures affecting them. It also greatly facilitates the calculation of groundwater budgets for different GWBs. Four Quaternary aquifers which have a more regional importance to the water supply systems or which are located on islands and do not have an underlying bedrock GWB, were kept as separate GWBs in the new delineation. In the future new independent Quaternary GWBs can be delineated when water abstraction from an aquifer increases or when it is shown that they exert an important influence to the groundwater dependent ecosystems in the area. However, before such new GWBs can be delineated, a comprehensive monitoring network has to be put in place in the area in question, so that the quantitative and chemical status of the GWB can be properly assessed (Marandi et al., 2019).



01 Cambrian-Vendian Gdov GWB
02 Cambrian-Vendian Voronka GWB
03 Cambrian-Vendian GWB
04 Ordovician-Cambrian GWB in West-Estonian RBD
05a Ordovician-Cambrian Virumaa GWB in East-Estonian RBD
05b Ordovician-Cambrian Tartu GWB in East-Estonian RBD
06 Ordovician Ida-Viru GWB
07 Ordovician Ida-Viru oilshale basin
08 Silurian-Ordovician Hiiuma GWB
09 Silurian Saaremaa GWB
10 Silurian-Ordovician Harju GWB
11 Silurian-Ordovician Matsalu GWB
12 Silurian-Ordovician Pärnu GWB
13 Silurian-Ordovician GWB in East-Estonian RBD
14 Silurian-Ordovician Pandivere GWB in West-Estonian RBD
15 Silurian-Ordovician Pandivere GWB in East-Estonian RBD
16 Silurian-Ordovician Adavere-Põltsamaa GWB
17 Silurian-Ordovician beneath Devionian in West-Estonian RBD
18 Silurian-Ordovician beneath Devionian in East-Estonian RBD
19 Middle-Lower Devonian Ruhnu GWB
20 Middle-Lower Devonian Kihnu GWB
21 Middle-Lower Devonian GWB in West-Estonian RBD
22 Middle-Lower Devonian GWB in East-Estonian RBD
23 Middle Devonian GWB in West-Estonian RBD
24 Middle Devonian GWB in East-Estonian RBD
25 Middle Devonian GWB in Koiva RBD
26 Upper Devonian GWB
27 Quaternary Vasavere GWB
28 Quaternary Meltsiveski GWB
29 Quaternary Männiku-Pelguranna GWB
31 Quaternary Prangli GWB

FIGURE 6 **Groundwater bodies in Estonia (Marandi et al., 1997)**

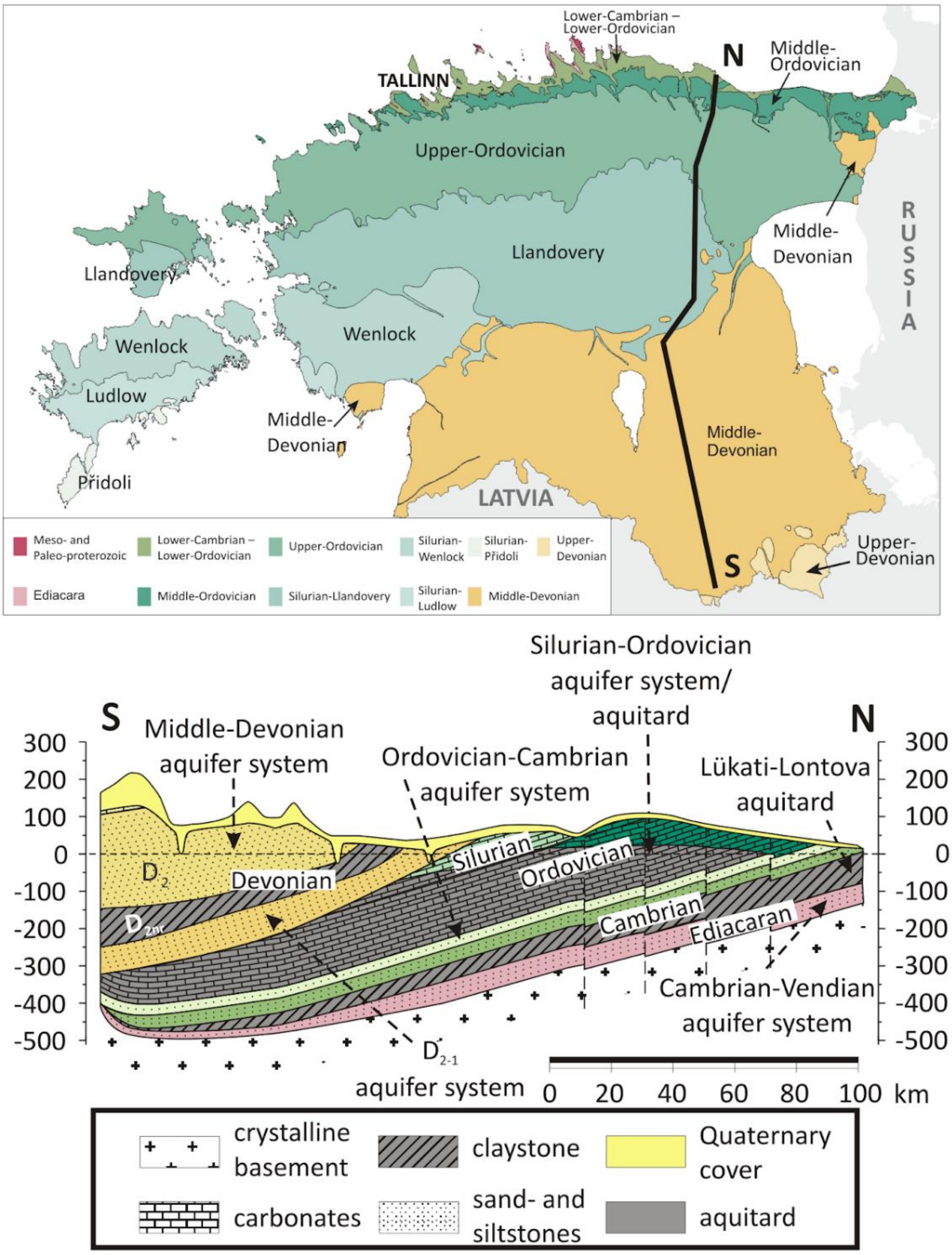


FIGURE 7 **Geological map of Estonia and hydrogeological cross-section of Estonian bedrock together with the distribution of major aquifers and aquitards. Modified after Pärn, 2018.**

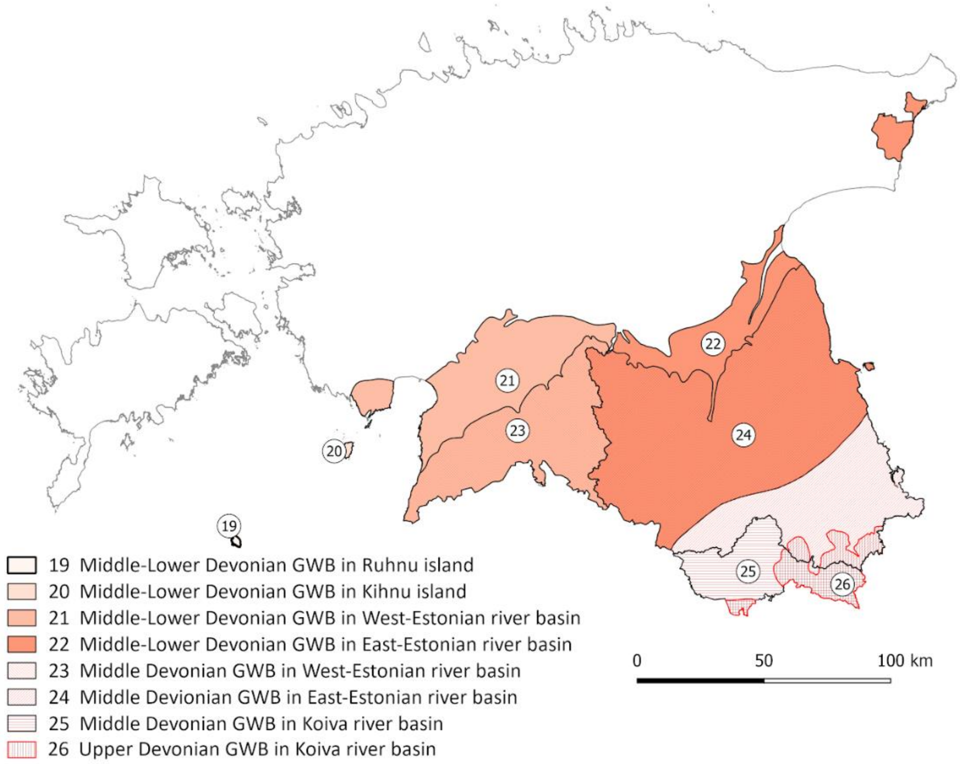


FIGURE 8 **Delineated groundwater bodies in Devonian aquifers (Marandi et al., 1997) (two GWBs (No.25 and No.26) are located in the Koiva river basin)**

4.2.2. Groundwater body delineation approaches in Latvia

In 2018 Latvia has started GWB and GWB at risk review process. According to WFD, GWB delineation process should be iterative and ongoing considering new knowledge base and new monitoring data. Step by step national GWB delineation approach is presented in FIGURE 9.

The first step (FIGURE 9, step 1) in GWB review process was initial identification of aquifers and their vertical boundaries according to the existing national knowledge – national hydrogeological stratification set in national legislation (MK noteikumi Nr.696) and freshwater distribution maps (Levins et al. 1998). This step is a common first step in the GWB delineation process in most Member States, because all national monitoring networks are built considering the national knowledge base. Ignorance of existing groundwater management principles would result in the need for installation of many new wells and monitoring stations which would be economically unreasonable and loss of long-term data series. As well basic hydrogeological conditions do not change over time and are still valid.

Second step (FIGURE 9, step 2a) was identification of aquifers in terms of WFD. This means that only aquifers providing significant quantities of water should be considered as aquifers. For that reason, two main assessments were carried out. Firstly, the water abstraction fields were mapped to identify the areas where aquifers currently provide more that 100 m^3/d of water and supply largest cities and villages. Secondly, the wells which currently supply or can provide more than 10 m^3/d of water should be mapped. As we currently could not use such data due to database development, a stricter approach was chosen to select wells which might be used for more than 10 m^3/d in the future.

Then (FIGURE 9, step 2b) aquifers were identified according to freshwater distribution. According to WFD only freshwater or water which are used for drinking water supply after the traditional treatment processes (e.g. iron removal with aeration) should be considered as aquifer and be a part of GWB. In this step the areas which contain low water quality, are not used for water supply and there are better groundwater quality aquifers above them, were not considered as aquifers and were excluded from GWBs. In some cases, low quality water areas were kept as a part of GWB as there are no better water resources available in the territory at the depth which is economically reasonable to install water supply wells.

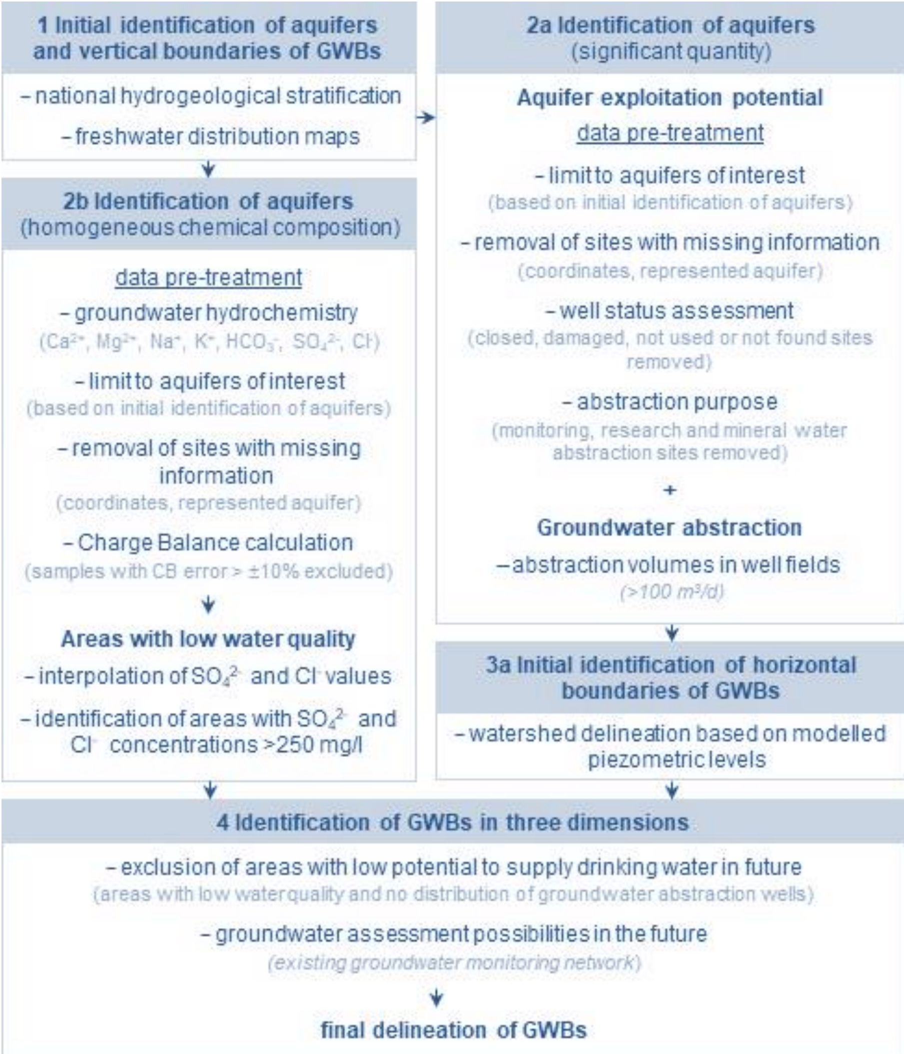


FIGURE 9 **Schematic flowchart illustrating main steps and data pre-treatment procedures for groundwater body delineation in Latvia**

In the third step (FIGURE 9, step 3a) watersheds were modelled using 3D hydrogeological modelling results (Virbulis et al., 2013). This allowed us to take into account horizontal hydrogeological conditions. And it could be observed that initial GWB delineation carried out in 2008 provided more or less similar results at least for the upper aquifers. As there was a limited amount of data in 2008 and the process was mostly manual, then more differences were observed in deeper aquifers as was expected.

Finally, GWBs in three dimensions were delineated based on all previous steps and considering existing monitoring networks. This was an essential step as delineation of GWBs should not be carried out only statistically and scientifically, but should take into account national opportunities to further monitor and assess the quantitative and chemical status of GWBs. For that reason, some of the watersheds were merged into larger GWBs only based on groundwater monitoring network distribution. In the future, when more funding is available for installation of new monitoring sites, the GWBs could be split.

As a result, initially 22 GWBs in good status were identified and after additional data analyzes also three GWBs at risk were identified; altogether 25 GWBs were identified in Latvia. Delineation approach of GWB at risk F5 (seawater intrusion) and methodology for establishing background levels and threshold values is described by Bikše and Retike (2018) and Retike and Bikše (2018). LVĢMC (2018) describes delineation approach of GWB at risk A11 (Sulphuric Goudron pollution), but LVĢMC (2019) describes delineation approach of GWB at risk Q2 (artificial recharge in Baltezers) and also methodology for establishing background levels and threshold values for two GWBs at risk A11 and Q2.

Current boundaries of GWBs in Latvia and Gauja-Koiva river basin are represented in FIGURE 10. In the Gauja-Koiva river basin altogether five GWBs are located: four GWBs (Q1, D6, A8 and P) in good status and one GWB at risk A11 - Sulphuric Goudron Pools of Inčukalns. More detailed information about these GWBs is available in ANNEX 1.

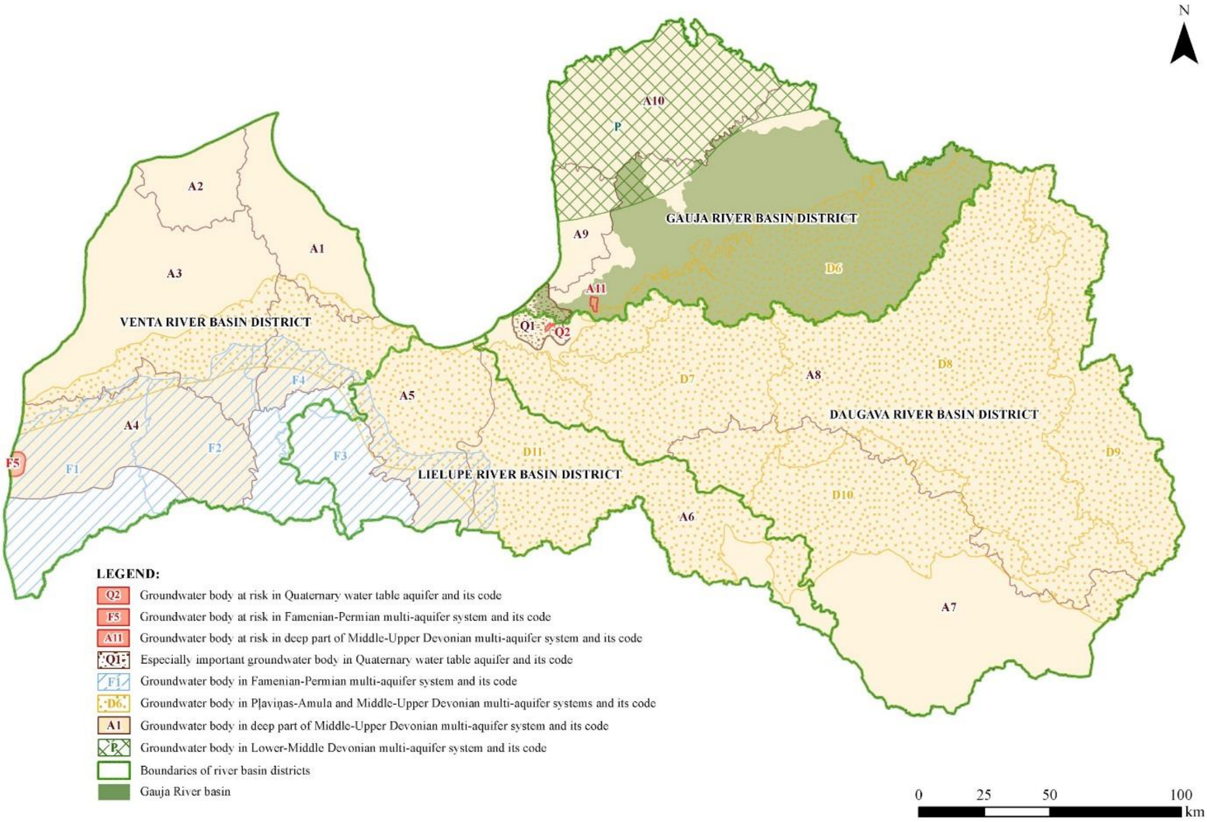


FIGURE 10 **Groundwater bodies in Latvia and Gauja-Koiva river basin**

4.3. Habitat assessment in Estonia and Latvia

4.3.1. Availability and quality of habitat data in Estonia

The description and classification of the habitats of Estonia started in the 1930s. After joining the European Union, the habitats were reclassified based on the habitat types listed in Annex I of the European Council Directive 92/43/EEC of 21 May 1992 on the conservation of natural habitats and of wild fauna and flora (Habitats Directive) according to the national interpretation (Paal, 2007) of these habitat types. The national interpretation manual (Paal, 2007) was compiled according to the Interpretation Manual of European Union Habitats (European Commission DG Environment, 2013) and the detailed vegetation site type classification of Estonia (Paal, 1997). From the habitat types listed in Annex I of the Habitats Directive, 60 are found in Estonia. Majority of habitat types are coastal, forest, mire and grassland habitats; the number of freshwaters, rocky and marine habitat types is lower. For collecting the habitat data and for inventories, the guidelines and data forms have been prepared for each habitat class (e.g. forest, mire, grassland). In addition to data relevant for the characterization of Annex I habitat types, vegetation site types according to Paal (1997) are usually determined during inventories as well.

A number of nationwide inventories have been conducted since re-establishing independence of Estonia in order to ascertain the occurrence and condition of various habitat types[7]. For example, national-scale inventories on the distribution and status of old-growth and natural forests; valuable forest habitats; coastal, floodplain, alvar and wooded meadows; and wetlands were performed in the 1990s. For the designation of Natura 2000 network, this information was used in addition to a national-scale inventory of Annex I habitat types performed in the early 2000s.

After the establishment of the Natura 2000 network, the process of inventorying continued, new areas important for biodiversity were identified and the already known areas were re-inventoried. For example, marine habitats were inventoried within inventories performed from 2007 to 2011 and 2014 to 2016. A national-scale mire habitat re-inventory was made from 2008 to 2011 and a follow-up project started in 2017 to amend and refine collected data. That inventory will be finalized in 2020. A national-scale coastal habitat inventory was carried out from 2014 to 2016.

The main aim of the inventories is to elaborate the protection regime and management plans for the protected areas. Therefore, habitat inventories have been performed in a number of protected areas since 2009. For example, over the last ten years, the forest habitats were re-inventoried in 84,000 hectares.

The datasets of Annex I habitats are stored in the Estonian Nature Information System (EELIS). EELIS is the main database in Estonia for nature-related data, which was initiated in 1999 and it is maintained by the Estonian Environment Agency. In its essence, it is a geographical information system that consists of a desktop application working with Microsoft Windows, which is linked to a PostgreSQL geodatabase in a central server. Most of the data are added and managed via the desktop application by the specialists in the Environment Agency. The main users of EELIS are environmental officials, but rights for using the software for viewing and downloading spatial data may also be obtained by scientists and specialists working in the field of nature via specific contracts.

Habitat data in EELIS is being constantly updated in cooperation between the Environmental Board and the Environment Agency. The Board orders inventories and the Agency enters the data to the database. The data from habitats' inventories is stored in the "natura_elupaik" dataset (the national dataset). If a habitat polygon or its attributes are updated according to a newer inventory, then the old polygon is archived and also the adjoining polygons are updated, therefore there are no overlapping polygons in the dataset.

In addition to the "natura_elupaik" dataset, there is a specific dataset of semi-natural Annex I habitats to help site managers to manage these habitats in protected areas. This dataset differs from the "natura_elupaik" dataset. These two datasets may have overlapping polygons, different habitat boundaries and/or habitat code. Currently, the work to include the semi-natural habitats' dataset into the "natura_elupaik" dataset is in progress. However, for the purpose of identifying GDTEs, the inconsistency of these two datasets is manageable and combining these does not create significant problems. Generally, habitat polygons in the national and the semi-natural communities' datasets contain full information on their status (representativity, degree of conservation of structure, degree of conservation of functions, restoration possibilities and global assessment). Occasionally, though, no status information at all or only the representativity is given. Lack of status information is more typical for patches that are located outside of protected areas and have been inventoried a long time ago, as re-inventories have mostly been made in protected areas.

Besides these two datasets, there are datasets from different non-governmental organizations (NGOs), e.g. ELF, PKÜ, containing habitat data that are also stored in EELIS. The main NGO-maintained dataset that contains groundwater dependent habitats is the dataset of mires. It includes data from various national-

[7] Table 5. Estonian habitat inventories from 1991 to 2007 in https://www.keskkonnaagentuur.ee/publications/329_PDF.pdf

scale mire inventories. Most of the habitat data from that dataset has not been transferred to the national dataset yet. That issue complicates the usage of the dataset of mires with the national habitat dataset for identifying GDTEs. Specific methodology would have to be developed in order to combine these two datasets, as polygon configurations, coverage and sometimes also habitat information differs significantly between them.

Altogether, knowledge on the distribution and status of Annex I habitats in Estonia is quite good and is constantly improving. Also, the data is readily available for users.

4.3.2. Availability and quality of habitat data in Latvia

In Latvia, the habitats are being identified according to the national interpretation manual of the Annex I habitat types (Auniņš (Ed.), 2013 and its latest amendments in 2016, published in the website of the Nature Conservation Agency, www.daba.gov.lv). The national interpretation manual has changed several times, because it was gradually improved, as the knowledge on the habitat types, their characteristics (sub-types, species, etc.) and their geographical distribution constantly increase. In Latvia, 61 habitat types listed in the Annex I of the Habitats Directive have been identified, out of these two are marine and 59 terrestrial and freshwater habitat types. The largest number of the habitat types are coastal, forest, grassland and mire habitats.

Habitat inventories are being done for various nature conservation and planning purposes (e.g. development of nature management plans, monitoring of Natura 2000 sites, inventories of semi-natural (biologically valuable) grasslands to implement agri-environmental scheme, different ecosystem restoration projects, e.g. under LIFE programme). There had been also habitat-specific inventories, e.g. inventories of semi-natural grasslands in 2000–2002, 2005–2009, and 2013; national scale and area-specific inventories of woodland key habitats in 1995–1996 and in the early 2000s. With very few exceptions, only protected habitat types (i.e. Annex I habitat types) are being mapped.

During the time period from 2017 to 2020, habitats are being inventoried and mapped within the so far largest national scale inventory that covers the entire territory of Latvia: the project "Preconditions for better biodiversity preservation and ecosystem protection in Latvia" (short name: Nature Census), No. 5.4.2.1/16/I/001). Since 2017, all surveyed habitat patches are attributed with a standard data field form which includes the basic information on the particular patch (habitat type according to the codes and names listed in the Habitats Directive, structures, functions and processes, impacts, characteristic species, protected species). The national scale inventory, both within and outside Natura 2000, is being carried out by certified habitat and species experts who are specially trained and have inter-calibrated their skills. Similarly, the habitat experts who contribute to nature management plans for protected nature areas, must be certified which means that certain qualifications and knowledge are required and must be improved on a regular basis.

All the habitat data and large amount of species data are stored in the National Biodiversity Information Data System OZOLS (http://ozols.daba.gov.lv/pub/), full information available for authenticated users including data downloading and updating options, whereas unauthenticated users can view the habitat and some other data online. Habitat data in GIS-formats are available also from the Nature Conservation Agency, the national public authority responsible for biodiversity data, upon request.

In 2019–2020, the habitat data in the National Biodiversity Information Data System OZOLS did not yet cover the entire territory of Latvia, as all inventory data are being double-checked before uploading into the database that is a time-consuming process. All the data from the Nature Census project will be available in the OZOLS data system around mid-2021. However, the data in the data system are constantly being improved, e.g. after local, detailed inventories, surveys for various purposes, as well as the situation is constantly changing (e.g. protected habitats are being restored, replaced by other land use types, etc.). Therefore, the habitat data layer is never "completed".

In 2019–2020, the quality of habitat data in Latvia was varying and uneven. The National Biodiversity Information Data System OZOLS contains both older information from earlier inventories, i.e. habitats mapped before 2017 (mostly without data field form information, only habitat code and sub-type, rarely some notes). In cases when the mapping has been done when developing nature management plans, more detailed, though descriptive, information can be found in the nature management plans (all available at the website of Nature Conservation Agency www.daba.gov.lv, in Latvian). Habitat patches mapped after 2017 mostly contain field data forms and cover areas outside Natura 2000.

Since numerous Natura 2000 sites or parts of them have been surveyed several times during the last decade and there had been slight differences in the national EU habitat interpretation, sometimes the data layers are overlapping with each other, i.e. certain habitat patches might be named with different codes or might have different patch configuration.

Considering the varying habitat data quality, at the current stage (2019–2020) before deeper analysis or further application the data user should critically revise the original data sources and, in the case of overlapping, use the newest data layers, which are of higher precision and contain more detailed information.

Vegetation studies and maps of plant communities may provide deeper insight into ecological processes in GDTEs. Vegetation maps and relevés with geospatial reference might help to identify the GDTE sites and their quality. However, up to now, there have been few studies on the vegetation and ecological processes of GDTEs, i.e. spring ecosystems, fens and other. The vegetation might be a useful indicator to identify GDTEs, however, currently the data and knowledge on the vegetation variation in Latvia and underlying drivers (groundwater dependent and other) are insufficient.

Several studies concerning vegetation diversity in groundwater dependent ecosystems (springs, fens, wetland forests) have been conducted in Latvia: forests – by Prieditis (1997a, 1997b), springs – by Pakalne (1994), Pakalne, Čakare (2001a, 2001b), Pakalne, Kalnina (2005), Pakalne et al. (2007). There is a private TURBOVEG-based Mire Vegetation Database of Latvia (GIVD ID EU-LV-002) that comprises more than 1900 relevés (in 2012, the database is being continuously being updated) described in mires (including GDTEs, but not only) in Latvia by L. Auniņa (Aunina, 2012). There are studies which focused on certain species groups, e.g. bryophytes and algae in spring ecosystems (e.g. Āboliņa, 1988; Pakalne et al., 2007; Medne, 2015). In many cases, the spring-related vascular plant species have been explored (data available in different published and unpublished sources, e.g. OZOLS data system). The studies on spring-related species do not analyze the relation with groundwater-related parameters and groundwater-dependency of these species. So far, no inventories focused on GDTE locations have been carried out in Latvia. For in-depth analysis, vegetation data can be extracted also from the vegetation monitoring data sets, i.e. if monitoring was carried out in areas that can be considered GDTEs – in most cases, such monitoring is/was conducted in nature conservation areas before and after restoration (grasslands, mires, rarely forests). These data are owned by the Nature Conservation Agency, scientific institutions or private owners.

Only one ecohydrological study carried out in Latvia is known. The authors have explored the groundwater-habitat-vegetation dependency in Slitere National Park (Elshehawi, 2019; Wołejko et al., 2019). The currently available published data can be used in identifying some GDTE areas, figuring out the diversity potential (vegetation variation) and, in a few cases, the data can be used in long-term landscape-scale monitoring of vegetation change in GDTEs.

4.4. Joint methodology

4.4.1. GDTE identification methodology for Estonia and Latvia

During the GroundEco project, GDTE identification was carried out both in Estonia and Latvia, but only in the Gauja-Koiva river basin. The Koiva basin in Estonia covers 1,309 km^2; the Gauja basin in Latvia is nearly 10 times larger with the total area of 13,039 km^2.

In identification of GDTEs, only the existing habitat and other available data were used. No special surveys (except for pilot areas, see Chapter 5) were carried out. Thus, the results may also reflect the data quality, though the impact of data deficiency on the result cannot be fully quantified. However, we assume that also in the future, the GDTE identification will be based on existing data sets, thus it will always depend on the habitat and other data quality and availability, unless special GDTE field inventories will be carried out.

In the GroundEco project, it was decided that the joint methodology for the identification of GDTEs in Estonia and Latvia will be based on the classification of habitats listed in Annex I of the Habitats Directive (hereinafter – Annex I habitat types). It is the path taken by most other EU countries (EC, 2014) and it is the only common classification that has been applied in both countries. The Annex I habitat types are all of Community interest and require protection according to the Habitats Directive, therefore they help to narrow down all potential groundwater dependent ecosystems to valuable groundwater dependent groundwater ecosystems, which should receive increased attention in terms of their status and well-being.

To identify the GDTE locations in Estonia and Latvia, from the list of all 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 and/or the national habitat interpretation. Each habitat type and its potential groundwater dependency was discussed by the experts involved in the GroundEco project. It was concluded that the habitat types that can be automatically considered GDTEs in Estonia and Latvia are:

- Humid dune slacks (2190);
- Fennoscandian mineral-rich springs and springfens (7160);
- Calcareous fens with *Cladium mariscus* and species of the *Caricion davallianae* (7210*[8]);
- Petrifying springs with tufa formation (*Cratoneurion*) (7220*);
- Alkaline fens (7230);
- Fennoscandian deciduous swamp forests (9080*).

Some habitat types are considered GDTEs only in exceptional cases either in both countries, or only in one of the countries. These are as follows:

- *Molini*a 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 5).

In Estonia and Latvia, in most cases the habitat type 6410 has developed in slightly drained fens that have been used for hay making and/or pasturing in the past, i.e. these are semi-natural habitats. Therefore, 6410 habitats are only considered GDTEs if they occur in larger GDTE habitat complexes, i.e. are part of a habitat complex which is predominantly fed by groundwater.

Habitat types 6430, 7110*, 7120 and 7140 are considered GDTEs under certain circumstances only in Estonia. This is due to different national interpretations of Annex I habitats in both countries and different

[8] Except for *Cladium mariscus* stands in lakes, as they cannot be considered terrestrial

degrees of impacts on groundwater. In Estonia, the habitat type 6430 includes fens (national vegetation site type 3.1.1.), paludified grasslands (2.4.1.), and floodplain grasslands (2.2.1.) according to the Estonian national interpretation (Paal, 2007). The first two habitat sub-types depend on groundwater input, therefore these sites of habitat 6430 are considered GDTEs, whereas sites that contain floodplain grasslands are not. Habitat types 7110*, 7120 and 7140 are considered GDTEs only in Northeastern Estonia in the oil shale mining region (FIGURE 2 in Chapter 3.2.1), and the dependence is only quantitative not qualitative. They are not critically dependent on groundwater under normal circumstances, but are considered GDTEs as a precautionary measure, as studies have shown (Kalm & Kohv, 2009; Marandi et al., 2013) that the groundwater drawdown caused by extensive subsurface mining could lower the water level both in transition mires or bogs. Both in Estonia and Latvia, the habitat type 91D0* includes not only transition mire and bog woodlands, which are not critically dependent on groundwater, but also coniferous fen woodlands, which are critically dependent on groundwater. Therefore, the sites where habitat 91D0* consists of fen woodlands should be considered GDTEs in both countries, whereas other subtypes of 91D0* – only in Northeastern Estonia, similarly as other habitats related with bogs and transition mires. In Estonia the distinction can be made if habitat inventories include data of national vegetation site types (fen forests – 4.1.1.). In Latvia, similar information is almost absent and currently it is not possible to make the distinction[9].

Other terrestrial habitat types related to waterlogged soils were not included in the list of groundwater dependent habitat types, because the experts involved in the GroundEco project assessed them to be more dependent on surface water or precipitation. These include the habitat type Boreal Baltic coastal meadows (1630*) that critically depends on seawater; Wet heaths (4010) critically dependent on precipitation; Northern Boreal alluvial meadows (6450), Alluvial forests (91E0*), Riparian mixed forests (91F0*) that critically depend on regular flooding.

In Estonia and Latvia, the selection of GDTEs is not limited to the Natura 2000 network of protected nature areas, but refers to the entire territory of both countries. In the identification process, all mapped Annex I habitat polygons in the official habitat datasets in both countries should be used. In Estonia, it includes both the official habitat dataset and the semi-natural communities' dataset but not the mire dataset (see Chapter 3.2.2.1.)

In order to prioritize GDTEs, to select the most important sites, to limit their number and subsequent financial burden related to monitoring, a minimum areal limit is fixed to 10 ha or 20 ha for the habitat polygons (the minimum area is habitat- and country-specific) (TABLE 5). The exceptions are very rare habitat types, which, in most cases, are naturally confined to small areas. For these the minimum area considered is 1 ha: habitats 2190, 7160, 7220*. The minimum areal limits were differentiated according to the actual cover and level of fragmentation of specific habitat types in both countries.

[9] The groundwater dependent 91D0* subtypes could be distinguished in the habitat inventory data since 2017, however, not in older inventory data. Thus, these criteria cannot be fully applied in Latvia yet

TABLE 5

Selected Annex I habitat types and additional criteria for selecting significant groundwater dependent terrestrial ecosystems in Estonia and Latvia

Habitat types listed in Annex I of the EU Habitats Directive (European Commission 1992)	**Additional criteria used (Latvia/Estonia)**
Considered as GDTEs	
Humid dune slacks (2190), Fennoscandian mineral-rich springs and springfens (7160), Petrifying springs with tufa formation (*Cratoneurion*) (7220*)	Single polygon with 1 ha area or smaller if part of a habitat complex with the total area of at least 1 ha.
Calcareous fens with *Cladium mariscus* and species of the *Caricion davallianae* (7210*)	Single polygon with 10 ha/20 ha area or smaller if part of a habitat complex with the total area of at least 10 ha/20 ha. *Cladium mariscus* stands in lakes are excluded.
Alkaline fens (7230)	Single polygon with 10 ha/20 ha area or smaller if part of a habitat complex with the total area of at least 10 ha/20 ha.
Fennoscandian deciduous swamp forests (9080*)	Single polygon with 20 ha area or smaller if part of a habitat complex with the total area of at least 20 ha.
Considered as GDTEs in exceptional cases	
Molina 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 the 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*), Degraded raised bogs still capable of natural regeneration (7120), Transition mires and quaking bogs (7140)	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 NE Estonia in oil shale mining region and only quantitative dependence), not considered GDTE in the rest of Estonia and in Latvia.
Bog woodlands (91D0*)	Transition mires and bog woodlands (only in NE Estonia in oil shale mining region and only quantitative dependence) - single polygon with 20 ha area or smaller if part of a habitat complex with the total area of at least 20 ha. Coniferous fen woodlands (included in 91D0*) (in both countries) - single polygon with 20 ha area or smaller if part of a habitat complex with the total area of at least 20 ha

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, 2011), and species that rely on groundwater-fed habitats listed in Annex II of the Habitats Directive in Latvia[10] (the list of species is given in TABLE 6). The presence of such species makes the site a significant GDTE, even if it does not meet the areal threshold.

Hereinafter the sites that comply with the above-mentioned criteria are called "significant GDTEs".

[10] The different criteria of GDTE-related species are based on different principles of listing the protected species in both countries. To avoid biased selection of species, the "GDTE species" were based only on existing protected species lists, though the actual number of species, including threatened species, that depend on GDTE habitats may be higher.

TABLE 6

GDTEs related species (additional criteria in identifying GDTE sites)

Important species relying on GDTEs in Estonia (Protected species of the categories I and II. Species selection according to Paal & Leibak (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, Coralorhiza 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	*Saussurea alpina ssp. esthonica, Ligularia sibirica, Liparis loeselii, Saxifraga hirculus, Vertigo geyeri, Vertigo genesii, Vertigo moulinsiana, Vertigo angustior*

Areas that have been highly damaged a long time ago and have a very low or no ecosystem recovery potential may be excluded from the GDTE list. 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. Again, the exceptions are rare habitat types that have very few locations and cover very small areas. Histograms of GDTE sites coverage are suggested to be made in order to assess the need for that criterion.

Sites consisting of multiple hydrologically or geomorphologically related patches (habitat complex) either from the same, or different habitat types may be merged into a single polygon (if the patches share a common border) or multipart (if the patches are separated) GDTE polygon with a single ID in the database. In that case the areal limit (10 ha/20 ha (Latvia/Estonia), or 1 ha in the case of rare GDTE types) applies for the whole merged polygon. The size of each individual patch should still be not less than 0.1 ha. Moreover, marginal fens and groundwater dependent woodlands around bogs or around bog woodlands can be approached as one site GDTE (polygon), but not the bog or bog woodland itself, as the latter are not considered GDTEs (unless in Northeastern Estonia). If the merged polygon consists of different habitat types, then these are described in the attribute table of the polygon.

In the process of testing the applicability of the GDTE identification methodology, several data insufficiencies were distinguished both in Estonia and Latvia that hinder a smooth identification process. When mapping Annex I habitats, the habitat patches are being identified on the basis of compliance of vegetation and abiotic conditions with the habitat description in the national Annex I habitat interpretation manuals. The area should meet certain criteria, e.g. presence of certain indicator species, minimum number of certain species, vegetation structure, processes. It means that the habitat patches depending on groundwater influence may be fragmented across landscape, e.g. spring mires may be fragmented by forest patches or many spring discharges can occur in river valley and ravine complexes, although feeding from the same aquifer. Thus, in perspective of GDTE identification, many habitat patches may belong to the same ecosystem and should be considered as a single GDTE. Additionally, the habitat maps may be with low precision or some GDTE habitat locations may be missed when carrying out inventories, or habitat types may be misinterpreted as non-GDTE habitat types (or vice versa). The habitat patches may overlap (mapped in different years with different quality of background information).

Therefore, the GroundEco project experts agreed that the selection of GDTE sites may not be based solely on habitat data, but also involve other types of data (e.g. relief, soils, peat deposits, forest types). Selection of GDTE sites is largely affected by data quality, i.e. precision of habitat data and other supporting data. In some cases, some important data layers might not be available or are outdated. For example, in 2019–2020 detailed relief data, based on LiDAR-data, were not available for the entire Gauja basin in Latvia, soil data is available almost only for agricultural lands and the maps are based on field inventories in the 1960s. In Estonia, relief and soil data are more readily available, as LiDAR-elevation data has been collected three times for the whole country and a digital soil map covering whole Estonia exists as well. The field inventories for the soil map were made between 1954–1990. Therefore, these layers support the habitat complex delineation process in Estonia well.

Topographic data help to identify geomorphologically related habitat polygons that should be approached as one GDTE. For example, all polygons of habitat 7220* that are inside one valley should be considered as a single GDTE (FIGURE 11). Organic soil and peat deposit data help to identify hydrologically related habitat polygons, for example various separated mire-habitat polygons that are situated on a single peat deposit should be considered as a single GDTE (FIGURE 12).

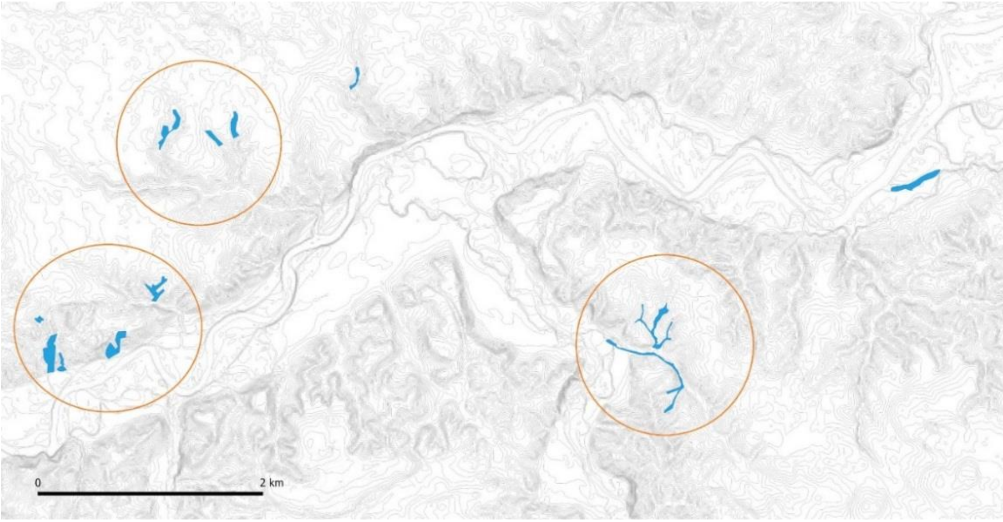


FIGURE 11 **An example of separate habitat polygons in a valley system and a suggestion how to combine them as a habitat complex for the identification of significant GDTEs**

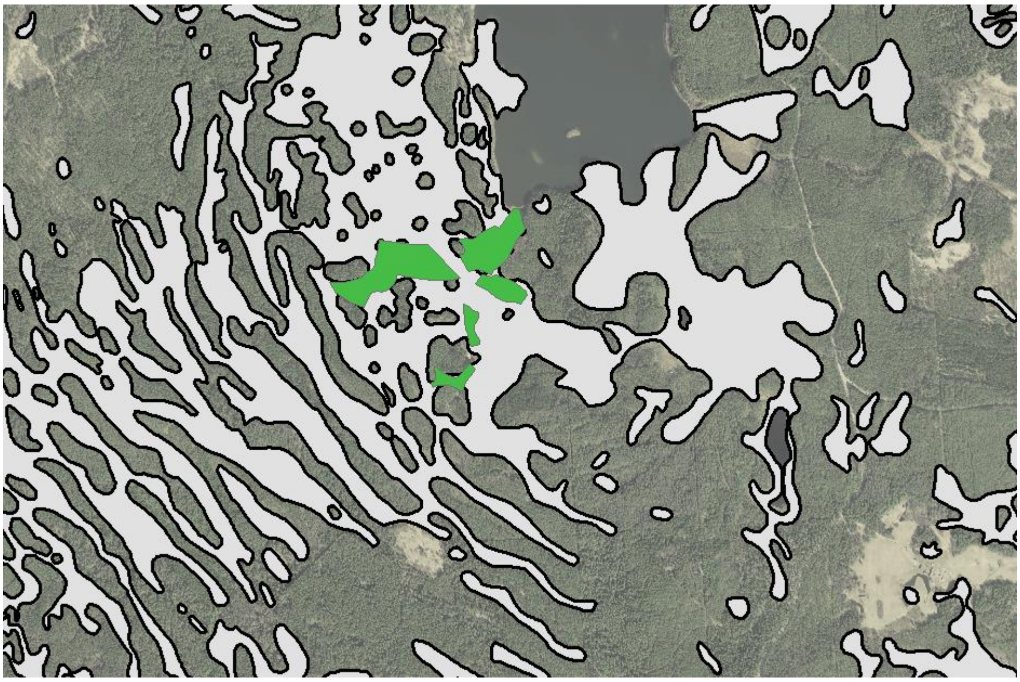


FIGURE 12 **An example of separate habitat polygons on a single organic soil (peat) deposit polygon; the habitat polygons should be considered as a habitat complex for the identification of significant GDTEs**

It means that **the GDTE site selection process cannot be entirely automated and cannot rely only on strictly defined criteria (e.g. habitat types and patch area), thus the expert judgement is still needed to take the final decision.** Also due to varying data availability in Estonia and Latvia the additional information that can be used to assist delineating habitat complexes will remain country-specific.

In addition to the varying availability of supportive data, the interpretation of Annex I habitat types in both countries varies slightly as well. Two main differences concerning habitat types considered GDTEs were identified:

a) Habitat type (Alkaline fens 7230) includes only base-rich (calcareous) fens in Latvia, but in Estonia it covers both rich (calcareous) and poor fens (acidic, oligotrophic). Poor fens do not have any protected habitat status in Latvia. Therefore, poor fens cannot be distinguished as GDTEs in Latvia from the Annex I habitat data, although they may be important in terms of ecosystems services and biodiversity conservation. However, by nature these are considered GDTEs also in Latvia; however, during development of this methodology (2019–2020) they could not be identified on the map due to lack of data.

b) In Latvia, the habitat type Hydrophilous tall herb fringe communities (6430) 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 both are considered GDTEs.

4.4.2. Results of the identification of significant GDTEs in Estonia and Latvia in Gauja-Koiva basin

According to the developed joint methodology, the delineation of significant GDTEs in the Koiva basin **in Estonia** resulted in 18 polygons in total covering an area of 570 ha (FIGURE 13). Their areas range from 1.9 ha to 92 ha and include six habitat types (TABLE 7). Eight sites include more than one habitat type. The most common GDTE in Koiva basin is the habitat Fenoscandian deciduous swamp woods (9080*), which is represented in 11 sites and also covers the largest area. The habitat type Hydrophilous tall herb fringe communities (6430) is represented in eight significant GDTEs, Alkaline fens (7230) – in seven, Mineral-rich springs and springfens (7160) – in two, and Bog woodland (91D0*) – in one significant GDTE. Twelve of the sites are fully inside Natura 2000 areas and 13 fully inside nature protection areas of national importance.

On four occasions the additional criterion of presence of mire-related species in the protection categories I and II were applied to polygons that otherwise would not have fulfilled the areal threshold. The GDTE-related protected plant species detected in potential GDTEs in the Koiva basin were *Malaxis monophyllos, Corallorhiza trifida, Dactylorhiza russowii, Liparis loeselii* and *Saxifraga hirculus*.

Ten significant GDTEs include habitat polygons whose conservation status has not been assessed and seven of the significant GDTEs include habitat polygons whose conservation status[11] is worse than "good" (B) (TABLE 7).

All significant GDTEs but one is presumably dependent on the Middle-Devonian groundwater body in the Koiva basin and one GDTE is presumably dependent on the Upper-Devonian groundwater body in the Koiva basin (TABLE 7). The dependencies on groundwater bodies were determined only based on spatial information. The uppermost groundwater body below the GDTE was considered to be the one that the GDTE depends on. Actual dependence could be verified using water level and chemistry data from both the groundwater body and the ecosystem. Before 2019 the verification would have been more important in

[11] Conservation status is a term used in the Habitats Directive for the assessment of the status of habitats. It combines three criteria – degree of conservation of structure, degree of conservation of functions, and restoration possibilities (Explanatory notes, n.d.). It can be provided at site level (field data forms, the Natura 2000 database) or nationally at country level in the Article 17 report under the Habitats Directive (mandatory for the EU Member States). Here, it applies to site level.

Estonia, because there were several independent Quaternary GWBs and most of the Quaternary aquifers were not considered as GWBs or parts of GWBs (see Chapter 4.2.1). Therefore, it would have been necessary to determine whether the water in the GDTE originates actually from a Quaternary GWB, a bedrock GWB below it or from a non-GWB Quaternary aquifer, as ecosystems depending on non-GWB aquifers are irrelevant in terms of the requirements posed by the WFD. Since 2019 almost all Quaternary aquifers in Estonia have been joined to the uppermost bedrock GWB and only four have remained as independent GWBs. Therefore, in most of Estonia (including the Koiva basin), using only spatial information for determining the GWB that the significant GDTE depends on, is reliable enough.

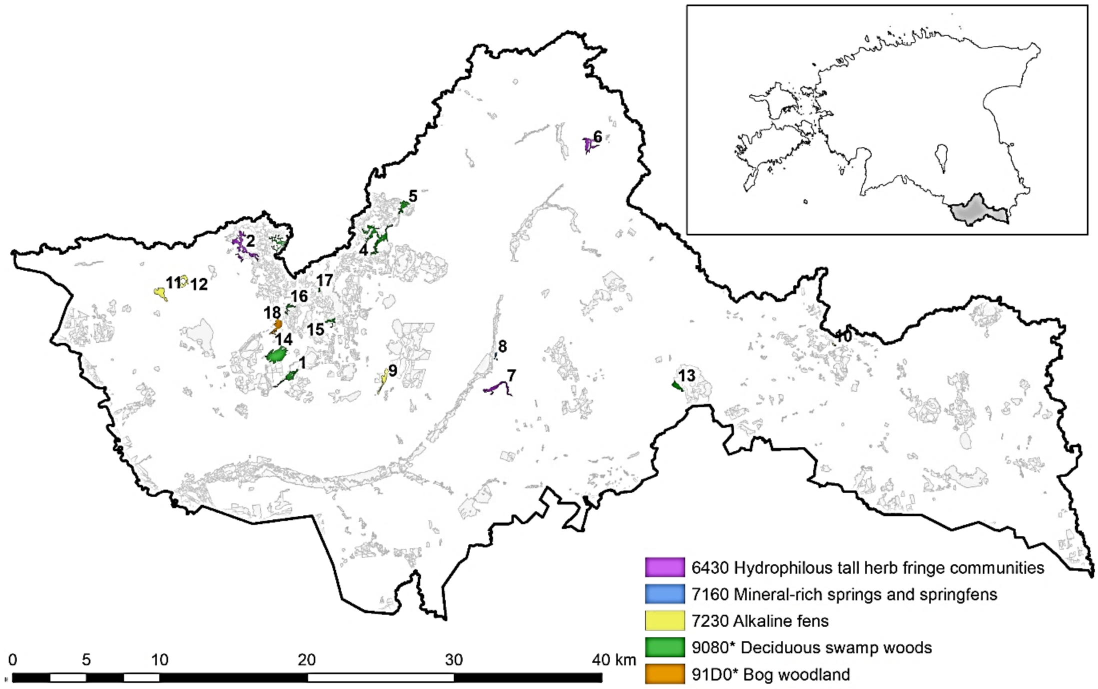


FIGURE 13 **Significant GDTEs identified in the Koiva river basin in Estonia. The colors of the polygons indicate the largest habitat type in the GDTE polygons (light grey color represents all Annex I habitat polygons in the Koiva river basin on November 20th, 2019). Numbers of the polygons overlap with the order numbers in TABLE 7**

TABLE 7

Attributes of the significant GDTEs identified in the Koiva river basin

No	Main habitat type	Secondary habitat types	Rare species	Natura 2000/ National PA	Conserv. status	Area (ha)	GWB
1	Fennoscandian deciduous swamp woods (9080*)	Hydrophilous tall herb fringe communities (6430)	0	Yes/Yes	A, B, C	32,6	D2
2	Hydrophilous tall herb fringe communities (6430)	Fennoscandian deciduous swamp woods (9080*), Alkaline fens (7230)	0	Yes/Yes	A, B, C, NA	68,4	D2
3	Fennoscandian deciduous swamp woods (9080*)	Hydrophilous tall herb fringe communities (6430)	0	Yes/Yes	NA, A	30,1	D2
4	Fennoscandian deciduous swamp woods (9080*)	Hydrophilous tall herb fringe communities (6430), Alkaline fens (7230)	0	Yes/Yes	C, NA	81,2	D2
5	9080* Fennoscandian deciduous swamp woods (9080*)	Hydrophilous tall herb fringe communities (6430), Alkaline fens (7230)	0	Yes/Yes	NA, B	24,9	D2
6	Hydrophilous tall herb fringe communities (6430)		0	No /Yes	NA	39,1	D2

No	Main habitat type	Secondary habitat types	Rare species	Natura 2000/ National PA	Conserv. status	Area (ha)	GWB
7	Hydrophilous tall herb fringe communities (6430)	Fennoscandian mineral-rich springs and springfens (7160)	0	No/No	B, A	35,0	D2
8	Fennoscandian mineral-rich springs and springfens (7160)		0	No/No	B, C	4,5	D2
9	Alkaline fens (7230)		0	No /No	C	25,1	D2
10	Alkaline fens (7230)		1	Yes/Yes	NA	1,9	D2
11	Alkaline fens (7230)		0	No/No	C	35,6	D2
12	Alkaline fens (7230)		0	No/No	NA	24,2	D2
13	Fennoscandian deciduous swamp woods (9080*)		0	Yes/Yes	A	20,9	D3
14	Fennoscandian deciduous swamp woods (9080*)	Hydrophilous tall herb fringe communities (6430)	0	Yes/Yes	C, NA, A	91,9	D2
15	Fennoscandian deciduous swamp woods (9080*)		1	Yes/Yes	B, NA	10,2	D2
16	Fennoscandian deciduous swamp woods (9080*)		1	Yes/Yes	NA	10,1	D2
17	Fennoscandian deciduous swamp woods (9080*)		1	Yes/Yes	A	2,7	D2
18	Bog woodland (91D0*)	Fennoscandian deciduous swamp woods (9080*)	0	Yes/Yes	A, B	31,7	D2

PA – protected area
Conserv. status – conservation status values according to the Annex I habitat status assessment
GWB – the groundwater body the ecosystem is dependent on (D2 – Middle–Devonian in the Koiva basin; D3 – Upper-Devonian in the Koiva basin)
NA – status not available
As the GDTEs may consist of several habitat polygons of the same or different types, there may be several conservation statuses values for a single GDTE

The GDTE delineation in the Gauja basin **in Latvia** resulted in 42 polygons (including multipart polygons) with the total area 162 ha (FIGURE 14). Significant GDTEs in Gauja basin include only two Annex I habitat types: Fennoscandian mineral-rich springs and springfens (7160) and Petrifying springs with tufa formation (*Cratoneurion*) (7220*).

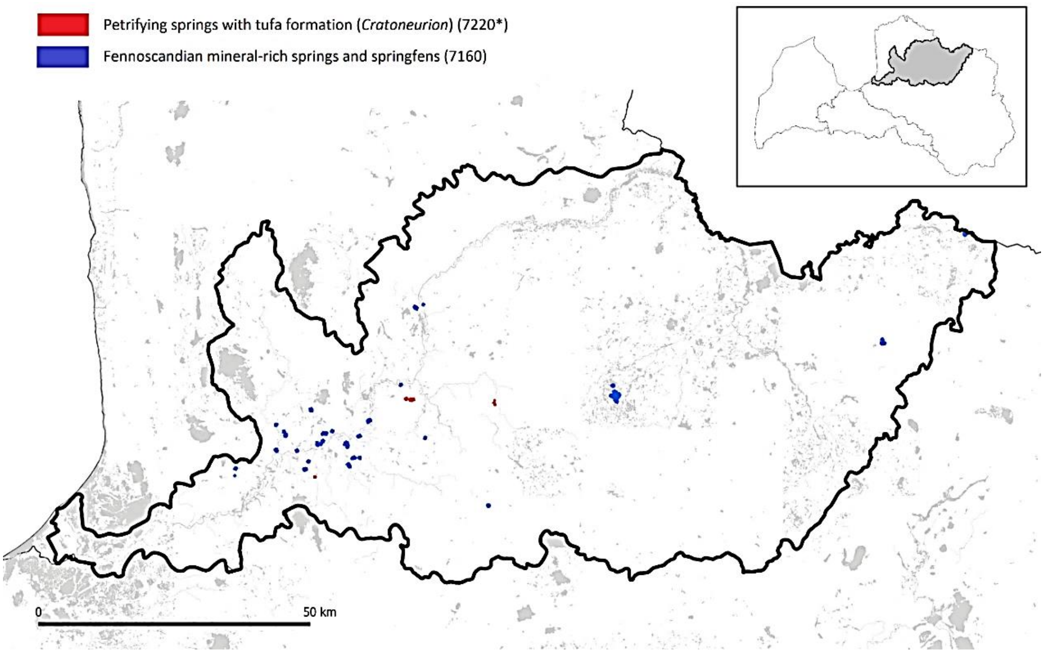


FIGURE 14 **Significant GDTEs identified in the Gauja river basin in Latvia. Light grey color represents all mapped Annex I habitat polygons in the Gauja river basin in November 2019.**

The areas of single patches or multipart polygons (considered as a single GDTE) range from 1 ha to 74.1 ha. In all cases using the existing habitat data (inventories until November 2019) the significant GDTEs occur in Natura 2000 sites (overlap with protected nature areas of national importance or micro-reserves), majority of them in Gauja National Park on the slopes of River Gauja Valley and side-valleys and ravines. The largest GDTE area was identified in Mežole Nature Reserve (habitat 7160). The other significant GDTEs occur in Veclaicene Protected Landscape Area, Raunas Staburags Nature Reserve, Zāgadu kalni micro-reserve and Bānūžu zelta avots micro-reserve. In most cases, a single GDTE consists of one of these habitat types, only in Raunas Staburags Nature Reserve both spring habitat types (7220*, 7160) form a significant GDTE. However, in some areas characteristics of both spring habitat types (7220*, 7160) are mixed and overlap (7220*, 7160), thus in the habitat inventories they are distinguished as one or another (e.g. Dāvida dzirnavu avoti, Bānūžu zelta avots).

According to the available data, in the Gauja basin in none of the cases presence species that rely on groundwater-fed habitats were found in GDTE habitat patches which were smaller than the minimum area (TABLE 5). The reason may be insufficient species data, especially concerning *Vertigo* spp.

Although the Koiva basin in Estonian side is almost 10 times smaller than the Gauja basin in Latvia, the total area of identified GDTEs is larger in Estonia (570 ha) than in Latvia (162 ha). Primarily, this may be explained by differences in the national interpretation of Annex I habitat types concerning the habitat type Hydrophilous tall herb fringe communities (6430), as in Estonia this includes poor fens that cover relatively large areas in the Koiva basin. In Latvia, such habitats are, most probably, present in the Gauja basin, but are not identified in the habitat inventories, because they are not considered as protected habitats. In the Koiva basin in Estonia, relatively large areas are covered by single patches of Fennoscandian deciduous swamp woods (9080*), a habitat type that is also present in the Gauja basin in Latvia, but the swamp woods are highly fragmented and thus continuous patches do not reach the minimum area. On the Estonian side several significant GDTEs were identified as Alkaline fens (7230), which are a very rare habitat type in the Gauja basin in Latvia and in none of the cases reached the minimum area. Similarly, in none of the cases Bog woodlands (91D0*) were not considered significant GDTEs in the Latvian side. In the Koiva basin, the spring habitats (7160, 7220*) are very rare and cover small areas, while on the Latvian side they are more common. Differences in both countries may be related also to land use, land use intensity and history (e.g. drainage intensity), degree of naturalness and proportion of protected nature areas.

Since the habitat data quality and quantity are constantly improving in both countries, the first attempt to apply the joint methodology in Latvia cannot be considered the final result. The GDTE maps should be revised if significant new habitat or other relevant data become available. In Latvia, the GDTE identification should be repeated after the country scale habitat inventory (the data will be available after mid-2021). After completing the inventory, the number of GDTEs and the total area of significant GDTEs may considerably increase.

4.4.3. Assessment schemes of groundwater dependent terrestrial ecosystems

4.4.3.1. Assessment schemes for the quantitative effect

After the identification of a GDTE in the landscape, and preferably verifying the dependence on the GWB using water level and water chemistry data, the next step is to assess whether the GWB could have a negative effect on the GDTE.

The assessment scheme for the negative quantitative effect of the GWB on the GDTE consists of five steps (FIGURE 15). The scheme was developed during the GroundEco project based on the preliminary Estonian assessment scheme (FIGURE 3) (Terasmaa et al., 2015) and the procedure suggested by the European Commission (EC, 2009).

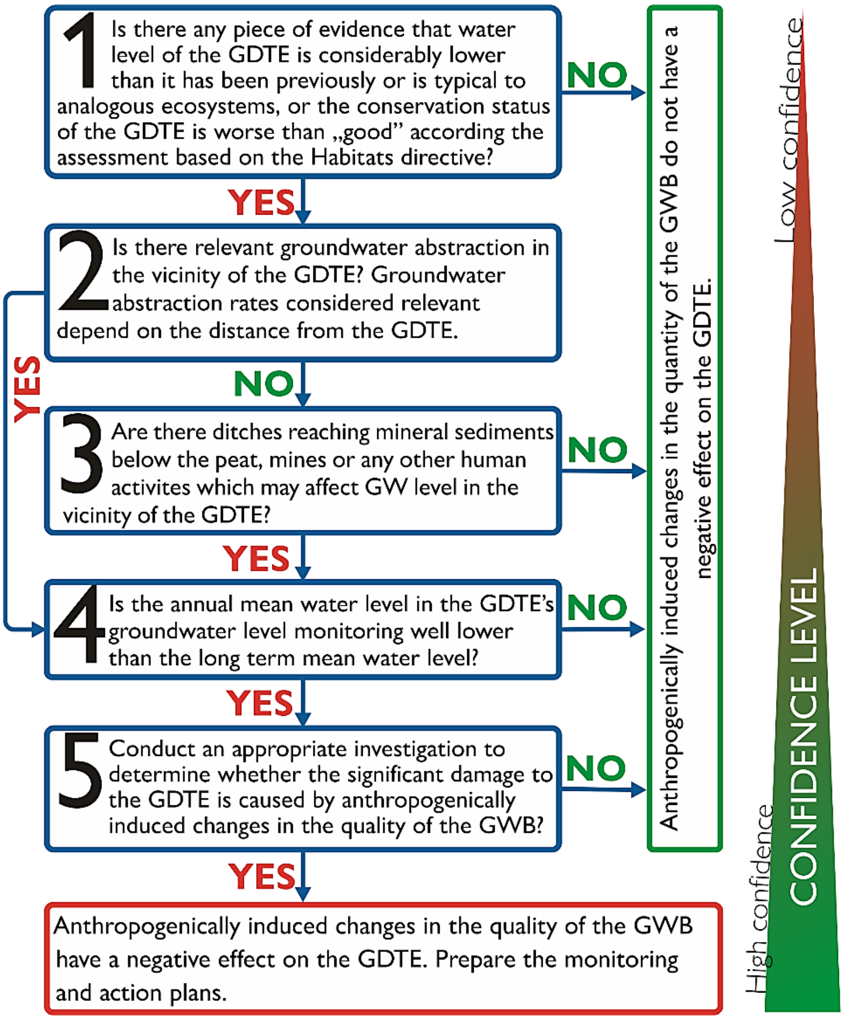


FIGURE 15 **Proposed scheme for assessment of significant damage to GDTE caused by quantitative pressures in GWB**

Combination of water level measurements in groundwater monitoring wells in GDTEs and their surroundings are typically used indicators for quantitative changes. For example, seasonal dynamics of mire water levels compared to short- and long-term average levels will indicate already occurring negative effects in GDTE (FIGURE 15, Step 1). Significant groundwater abstraction, dense abstraction well network or known human activities lowering the groundwater levels near GDTE can also result in a decrease of groundwater inflow to GDTE, thus deteriorating its condition (FIGURE 15, Step 2–3). Groundwater level drop in the monitoring well of GWB feeding the mire or the change of spring discharge providing the base flow for spring fen will indicate the first negative tendencies (FIGURE 15, Step 4). Still, appropriate measures may stop the negative influence, but timely actions may prevent potential damage to GDTE. The proposed assessment scheme (FIGURE 15) requires often limited long-term monitoring data (Kløve et al., 2011b) or missing water level targets for different GDTE types (Whiteman et al. 2010). Thus, the most common indicator is actual evidence of damage in GDTE (FIGURE 15, Step 1) observed as an ecological response to groundwater depletion (Rohde et al., 2017).

Commonly used groundwater quantitative management approaches in the GDTE management area are volumetric allocations, buffer/well exclusion zones, groundwater level triggers, and groundwater rate of decline triggers. Their advantages and disadvantages from a technical and implementation perspective are well described by (Noorduijn et al., 2019). Three groups of management approaches were identified:

Volumetric approach. Abstraction limits are often set as fraction of natural recharge for groundwater catchment or as historical long-term average abstraction rate, assuming that the system is already in a new steady state. In management plans the term "Environmental water allocation" (EWA) is sometimes included, explicitly reserving a share of groundwater for maintaining the GDE. Definition of EWA gives the environment the same legal status as other (human) water users. The approach requires reliable estimation of groundwater recharge. It can be useful for sustainable exploitation of groundwater resources in a catchment, but individual GDE sites, located close to groundwater abstraction sites can be permanently damaged.

Buffer zone approach foresees defining buffer zones around GDEs where groundwater abstraction is limited. It is one of the simplest measures. A number of analytical equations have been developed to model the buffer zone approach. The buffers zones will provide short term protection to GDE from groundwater abstraction; therefore, they should be used in combination with other approaches. It can be applied in combination with volume-based allocations – "soft-buffer zones" – where abstraction in close proximity to GDE can be limited by volume or season.

Groundwater response trigger values such as groundwater level or rate of groundwater head decline can be predefined indicating the need for initiating groundwater conservation measures. The problem with the trigger approach is that even in the event when trigger value is exceeded and groundwater abstraction is stopped completely, the groundwater table will continue to decline for some time. The trigger approach requires identification of adequate observation locations and threshold values and establishment of monitoring infrastructure.

In the assessment scheme we recommend a simple volume-based assessment approach. This approach should help to identify cases when groundwater abstraction can potentially have impact on GDTEs and in-depth investigation is needed. More sophisticated measures should be elaborated on a case-by-case basis in line with existing regulations.

Step 1. National monitoring and inventory programmes for terrestrial ecosystems in Estonia and Latvia do not include water level monitoring. Therefore, there is no data on their water level and its dynamics. Generally, the deterioration of the status of mire habitats is caused by a decrease in the mire water level. Therefore, the first step is to find out if there is any piece of evidence that the water level of the GDTE is considerably lower than it has been previously. For that historical maps, pictures, oral descriptions or any other means of information can be used. Comparison with analogous ecosystems known to be in good condition is also one possible solution for determining the potential water level decrease and deterioration of the ecosystem. If the given ecosystem already has conservation status determined and it is less than good (B) according to the assessment of habitats listed in the Habitats Directive, this GDTE also moves to step 2. If the habitat status is at least good and there is no evidence about water level lowering, the GDTE will be not assessed further.

Step 2. It is proposed that simple as possible assumptions are used to initially assess if groundwater abstraction can have impacts on GDTEs (FIGURE 15, Step 2). One possibility is to estimate the area needed for the groundwater recharge to compensate for groundwater abstraction.

The decline of groundwater head within an aquifer due to water abstraction is triggering enhanced inflow of the water from aquifers above and below and recharge from the surface. The rate of this additional groundwater inflow is controlled by local hydrogeological and hydrological conditions; therefore, it can vary significantly from place to place. Provided that groundwater abstraction is sustainable, in a certain distance from the abstraction site enhanced water flow to the aquifer (recharge) will compensate for the water abstraction. In a simplified case it can be assumed that there is a circular area around the abstraction site that is affected by it. Groundwater abstraction rates considered relevant depend on the distance from the GDTE according to the following equation:

$$x = \sqrt{\frac{Q_{yr}}{\pi \times R_{yr}}} ,$$

where x is the distance from the water abstraction or group of wells (m), Q_{yr} is abstraction from the GWB ($m^3\ yr^{-1}$), R_{yr} is average recharge rate ($m^3\ m^{-2}\ yr^{-1}$). It is recommended to use local value for R_{yr}. If the information is not available use the following value: $R_{yr} = 0.07\ m^3\ m^{-2}\ yr^{-1}$. If the GDTE is closer to the abstraction site than the value of the x, groundwater abstraction is considered relevant to the GDTE. More complex methods such as a combination of most appropriate tools or local-scale mathematical models should be used to conduct investigations when there is evidence for the possible threat to GDTE (FIGURE 15, Step 5).

Step 3. If there is no relevant GW abstraction in the vicinity, also other means of human activities may lower the GW level and have a negative impact on the GDTE. Deep ditches reaching the mineral sediments below the peat and mines (both open-pit and underground) could be the reasons. Appropriate investigation (using maps, databases, *in-situ* observations) of the presence of such stressors has to be carried out.

Step 4. The next step is to assess if the annual average groundwater level in the aquifer feeding or supporting the GDTE is lower than its long-term average water level. A groundwater level drop in the recharge area of the GDTE will likely cause a drop in the amount of water reaching the ecosystem. A drop in the groundwater level downstream of the GDTE, could cause an increase in the amount of water seeping out from the ecosystem. If the ecosystem does not directly feed from groundwater, but its water level is kept high because of the pressure from the underlying aquifer, a drop in groundwater level could also cause a drop in the ecosystem's water level. If water level data is not available, a preliminary three-point monitoring network has to be set up (see Chapter 5 for best practices). This data helps to evaluate whether the water level drop in the ecosystem could be caused by groundwater abstraction or other human activities in the vicinity of the GDTE that have lowered the groundwater level.

Step 5. If the negative effect on the GDTE by the GWB cannot be ruled out in the first four steps of the assessment, a more thorough appropriate investigation has to be conducted. A thorough study (see Chapter 5 for best practices) includes a detailed water level network in and around GDTE, and collection of the hydrogeological and meteorological data, to determine the functional connections between the GDTE and the GWB.

After that a monitoring and action plan should be prepared and applied in order to survey possible further changes in the ecosystem's status and to find out means for improving it. That step is especially important for GDTEs situated in protected or Natura 2000 areas.

The proposed quantitative assessment scheme (FIGURE 15) provides more concrete steps, than the initially proposed scheme by the European Commission (2009). At the same time, it leaves more flexibility by also considering indirect data (such as analysis of any evidence in FIGURE 15, Step 1 and 3) and assigning the confidence levels. The reason for that is reporting deadlines set by WFD (European Commission 2000) and the high probability that many necessary data will remain missing as gathering it takes a long time.

Unfavorable status of GDTEs is often caused by surface drainage. Due to a typically smaller area of the spring fens in comparison with minerotrophic fens, the area affected by the surface drainage is usually smaller in the spring fens, and the drainage caused water deficit might be more intensively mitigated by the constant inflow of groundwater. As highlighted by Kilroy et al. (2005), the delineation of catchment areas for GDTEs is an essential step to carry out an appropriate assessment of quantitative pressures.

UKTAG (2004) emphasizes that many GDTEs could be damaged by a variety of pressures such as afforestation and land development which are not directly related to groundwater supply, and identification of main pressures requires detailed assessments. Thus, other potential causes must be ruled out at first. The negative quantitative effect from GWB is expected if there have not been any changes in the surrounding drainage network and other surface water bodies, but the water level in the mire has dropped and discharge from the mire has decreased. It is important to consider also meteorological conditions which may have a considerable effect on the water regime in the mire. When the linkage degree between GDTEs and GWB is not well understood, then GWB can be classified as of good status but may remain at risk. These cases may be prioritized for further investigation (European Commission, 2011).

4.4.3.2. Assessment schemes for the qualitative effect

The chemical status of GWB is determined based on established TVs by each Member State required by Groundwater Directive (European Commission, 2006). The criteria for establishment and application of TVs are set in the guidance document (European Commission 2009), and it is based on the outputs of European Union research project "BRIDGE" (Müller et al., 2006). Derivation of groundwater TVs for GDTEs is a complex task and requires a good knowledge of ecological needs and responses to various pollutants of each GDTE type. Also the information about the influencing area expressed as GDTE catchment area and domination pressures, as well as a good understanding of the main physical, chemical and biological processes occurring in the pathway from groundwater formation in GWB to groundwater inflow into GDTE is needed (Hinsby et al. 2008; Kilroy et al. 2005; Kløve et al. 2011a; Rohde et al. 2017).

Many EU Member States struggle with the derivation of TVs for GDTE and have set limits for nitrates mostly. In terms of phosphates, the derivation process is ongoing in the United Kingdom (European Commission, 2015; UKTAG, 2012). GDTEs are often not considered when establishing national groundwater monitoring networks, and lack of long-term integrated monitoring programmes combining the needs for both groundwater and nature protection has led to poor understanding of GWB and GDTE interactions (Kløve et al. 2011b). Thus, Estonia and Latvia have not established TVs for GWBs considering GDTEs.

Similarly, to the quantitative assessment scheme (FIGURE 15), a qualitative or chemical assessment scheme (FIGURE 16), applied to terrestrial ecosystems identified as being groundwater dependent, was developed during the GroundEco project. It consists of five steps as well and was developed based on the preliminary Estonian assessment scheme (FIGURE 4) (Terasmaa et al., 2015) and the procedure suggested by the European Commission (EC, 2009). The following analysis should be carried out only if there is any evidence of GDTE not having a good status (FIGURE 16, Step 1). Considering the fact that groundwater TVs for GDTEs are missing in many countries (Rohde et al. 2017) and their establishment is a long-term process which includes the negotiation between various experts, managing authorities and decision-makers, the proposed second step (FIGURE 16, Step 2) suggests usage of indirect data (such as fertilization amounts, location of polluted sites and land cover data). Such analysis will indicate if there are any significant human-induced chemical pressures, their type, and will point out the relevant parameters to be monitored and analyzed. After that the national monitoring should provide necessary data for comparison with the established TVs (FIGURE 16, Step 5). Again, there are certain proposed strategies on how to set GWB chemical status in case the groundwater monitoring results exceed the derived TVs (European Commission 2009). Still the chosen approach is a matter of each Member State.

Step 1. National monitoring and inventory programmes for terrestrial ecosystems in Estonia and Latvia do not include water chemistry monitoring. Therefore, there is no data about water chemistry and decisions have to be made based on indirect evidence - if the GDTE is in considerably worse conditions than it has been previously or is typical for analogous ecosystems because of the water quality of the GWB. For that historical maps, pictures, oral descriptions or any other means of information can be used. If the given ecosystem already has conservation status determined and it is less than good (B) according to the assessment of habitats listed in the Habitats Directive, this GDTE also moves to step 2. If the habitat status is at least good and there is no other evidence about changes in ecosystem, the GDTE will be not assessed further.

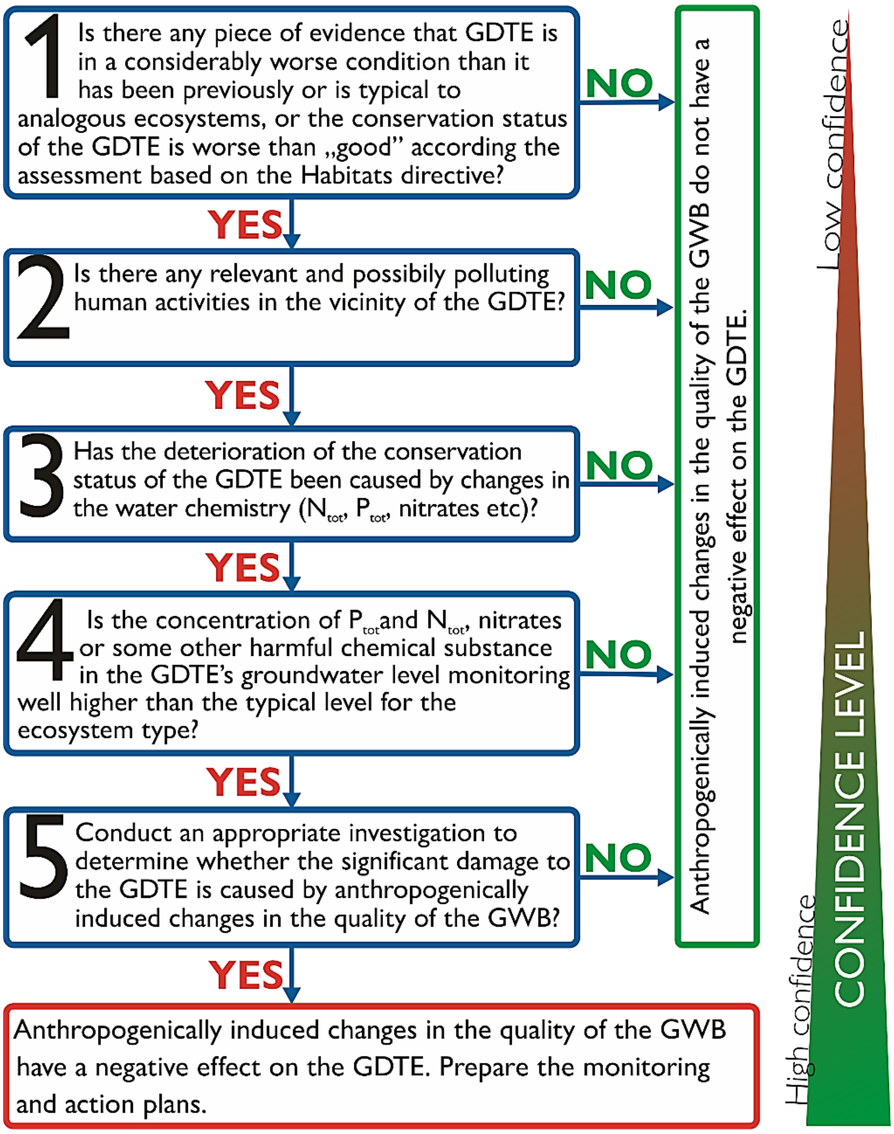


FIGURE 16 **Proposed scheme for assessment of significant damage to GDTE caused by qualitative (chemical) pressures on GWB**

Step 2. This step evaluates whether there are any relevant and possibly polluting human activities (e.g. agriculture - fields, cattle farms; wastewater treatment systems; industrial complexes) in the vicinity of the GDTE which may affect the given ecosystem through the GWB. Appropriate investigation (using maps, databases, *in-situ* observations) has to be carried out.

Step 3. This step assesses whether the deterioration of the status of the ecosystem could be caused by changes in the water chemistry (increased concentrations of Ptot, N_{tot}, nitrates or other relevant substances). As groundwater threshold values for GDTEs are missing and their establishment is a long-term process, the usage of indirect data (such as fertilization amounts, location of polluted sites and land cover data) is suggested. If the given ecosystem's worse than good status is explainable by terrestrial effects (drainage, peat extraction, alkaline pollution etc.) and not by impact from the GWB, the GDTE will be not assessed further.

Step 4. After that the groundwater quality in the ecosystem and ecosystem's catchment area has to be determined. If the water chemistry data is not available, a preliminary monitoring network has to be set up (see chapter 5 for best practices). These data help to evaluate whether the worse than good status of the ecosystem could be caused by poor water quality of the groundwater. If the concentration of P_{tot}, N_{tot}, nitrates or other relevant substance in the groundwater monitoring well of the GDTE is higher than the level typical for the specific ecosystem type, then the GDTE moves to the next step.

Step 5. If the negative effect on the GDTE by the GWB cannot be ruled out in the first four steps of the assessment, a more thorough appropriate investigation has to be conducted. A thorough study (see Chapter 5 for best practices) includes a detailed water sampling network in and around the GDTE to determine the functional connections between the GDTE and the GWB.

After that a monitoring and action plan should be prepared and applied in order to survey possible further changes in the ecosystem's status and to find out means for improving it. That step is especially important for GDTEs situated in protected or Natura 2000 areas.

The proposed qualitative assessment scheme (FIGURE 16) differs slightly from the one proposed by the European Commission (2009) again providing more flexibility to conduct preliminary analysis using indirect data to identify if the damage to GDTE potentially could be a result of chemical pressures on GWB supporting the GDTE (FIGURE 16, Step 2). In the case of missing knowledge, the GWB is considered to be still in good status but with low confidence, thus allowing to prioritize such cases for further investigations. Only then the use of appropriate groundwater TVs, and consequently, derivation of them is suggested (FIGURE 16, Step 4). The reason is the same as for the quantitative assessment scheme (FIGURE 15) that the Member States must report results in a certain timeline (EC, 2000).

4.4.4. Monitoring of GDTEs in Estonia and Latvia

Assessing the effects of GWBs on groundwater dependent ecosystems in Estonia and Latvia is complicated because monitoring networks often have not been established to determine interactions between groundwater and surface water. Often the parameters monitored from surface water or terrestrial ecosystems are relevant only for these ecosystems. Parameters monitored from GWBs are relevant only for groundwater and monitoring points are located mostly in places that are not representative for GDTEs.

In Latvia, potentially threatened GDTEs and other mires occur in some areas in the surroundings of quarries where groundwater table is lowered (e.g. surroundings of Lielie Kangari Nature Reserve, surroundings of Ape, surroundings of Popes zāļu purvs Nature Reserve). In cases when it may threaten significant protected habitats and/or Natura 2000 sites, the mining companies are obligated to carry out groundwater monitoring and, in most cases, also vegetation monitoring. As the monitoring should be established prior to starting the mining activity, this approach could potentially provide useful data for estimating the impacts of GDTEs and consequently GWB, as well as the adaptive measures should be taken if the ecosystem condition is worsening. However, in 2019–2020 no cases were known with available data and/or data series that represent data from several years. According to the national legislation, the data are coordinated and administered by several institutions, but there are no clear mechanisms how to act in cases when the groundwater abstraction or groundwater table lowering cause unfavorable impacts on ecosystems. The data are not available for public use.

On a few occasions' similar groundwater and mire-water monitoring obligations have been put to mining companies also in Estonia. Currently such monitoring is performed in two protected bogs and adjacent areas (Muraka bog and Selisoo bog) in Northeastern Estonia close to underground oil-shale mines. Vegetation monitoring is performed as well. The data is made publicly available once a year. It is established in the mining permits that if the monitoring reveals "undesirable environmental effects" to the protected areas, then the Environmental Ministry has the right to change the mining permits.

In assessing the quantitative effect of groundwater supply on GDTEs (FIGURE 15), the first step is evaluating whether the mean or annual water table of the GDTE is considerably lower than it has been before or in analogous ecosystems, or the status of biota indicates water table decrease. It means that if no special monitoring programme for GDTEs is developed at national level (currently does not exist neither in Estonia nor in Latvia), the impacts caused by the drop of water table might be first detected within habitat monitoring programmes, i.e. in case of Estonia and Latvia – monitoring of Annex I habitats. In both countries, such national monitoring programmes exist. The same applies for the assessment of the qualitative effect of groundwater on GDTEs. In the first step of the assessment scheme the ecological status of the ecosystem is the main indicator of potential negative influence from the GWB. As no separate evaluation system exists and most likely will not be developed neither in Estonia nor Latvia, then the conservation status of habitats obtained in the framework of habitat monitoring programmes provides the necessary information.

In Estonia, habitat monitoring is a part of the national environmental monitoring programme and belongs to the biodiversity and landscape monitoring sub-programme. Only Annex I habitats are being monitored. In March 2019 a new national environmental monitoring programme was approved and therefore also the structure of habitat monitoring was reworked. Previously "Monitoring of endangered (Natura 2000) plant communities" included separate surveys and methodologies for a) alvars and heathlands (6280*, 4030); b) alluvial and paludified meadows (6450, 7230); c) forest habitats (2180, 9010, 9020, 9050, 9060, 9070, 9080*, 9180, 91D0*, 91E0 and 91F0); d) dry meadows (6530*, 6210*, 6270*); e) coastal meadows (1630); f) mires (7110*, 7120, 7140, 7160, 7210*, 7230). Since 2019 monitored habitats have been divided into five groups with specific methodologies: forests, grasslands, wetlands, coastal habitats and marine habitats.

Under each habitat group the following Annex I habitat types are considered:

- forest habitats: 2180, 9010, 9020, 9050, 9060, 9070, 9080, 9180, 91D0*, 91E0 and 91F0;
- grassland habitats: 6280*, 6530*, 1630, 4030, 6210*, 6270*, 6450 and 7230;
- coastal habitats: 1210, 1220, 1230, 1310, 1640, 2110, 2120, 2130*, 2140*, 2190 and 2320;
- wetlands: 7110*, 7120, 7140, 7160, 7210*, 7230;
- marine habitats – the division and specific methodology has not yet been outlined.

Therefore, out of six habitat types considered as GDTEs (TABLE 5), only four (excluding Humid dune slacks (2190*) and Petrifying springs with tufa formation (7220*)) have been monitored in Estonia. Habitat types 7160, 7210*, 7230 have been monitored as mires and habitat type 9080* as forests. Of the six habitat types considered GDTEs in exceptional cases, only four (excluding *Molinia* meadows… (6410) and Hydrophilous tall herb fringe communities (6430)) have been monitored in Estonia. Habitat types 7110*, 7120 and 7140 have been monitored as mires and habitat type 91D0* as forests.

Neither under the previous nor the current version of the monitoring programme, the sites chosen for monitoring were/are not restricted to Natura 2000 sites (i.e. sites of special community interest or special areas of conservation), but were/are chosen from the whole list of the mapped habitat polygons. However, the quality of mapping varies among different regions in Estonia. In some areas, exhaustive inventories have been carried out and, in some areas, mostly known habitats in protected areas have been mapped. Therefore, because the original sample favors protected areas, then the monitoring sample is not entirely representative of unprotected sites.

Mire habitats were monitored between 2006–2016 in rotation in a 6-year cycle. During the first cycle (2006–2011) ca. 120 mires were monitored in total. Most of them were revisited during the second cycle. The mires were not chosen randomly, but it was attempted to distribute the sites proportionally over the country. During the first years separate programmes "monitoring of bogs" and "monitoring of fens" existed. 10 bogs and 10 fens were visited every year. Later also transition mires and spring fens were included. Therefore, habitat types 7110* and 7230 dominated over other monitored mire habitat types. The methodology of mire monitoring consisted of two parts: a) qualitative estimations of the status, structure, functions of mire habitats and various human-caused impacts, and b) vegetation analyses on 2 x 2 m (divided into four 1 x 1 m subplots) permanent plots. On each monitored mire, the permanent vegetation plots were analyzed and the qualitative assessment was made around the plot for each existing habitat type. Also, parts of the habitat in different conditions or with varying character were assessed.

Since 2017, that kind of monitoring has been suspended and the aim of the new methodology is to collect information from more areas but with a longer time step. In addition to national monitoring, a full scale mire inventory (all areas larger than 0.5 ha, but not mire-forests) was performed in Estonia by the Estonian Fund of Nature in 2008–2011. Among other parameters, it included determination and evaluation of Annex I habitat types in the mires. As of 2020, the results of that inventory are not yet fully included in the "natura_elupaik" dataset in the EELIS database, which houses the official Annex I habitat data in Estonia (see Chapter 4.3.1.). There are plans to incorporate that data to the EELIS database, though. The next similar inventory is planned 15 years after the previous one and that is supposed to provide all the data necessary to assess the status of Annex I mire habitats. As mires are naturally a very slowly changing habitat, that interval is considered sufficient for detecting meaningful changes in their status.

Forest habitats are monitored on the site around a randomly placed point and not in permanent monitoring areas (methodology available here, in Estonian). The evaluated criteria are assessed in a circle with a 20–40 m radius. Monitoring sites for the period 2013–2018 were randomly selected from the full list of all forest habitat polygons, taking into consideration the areal proportion of specific habitat types in Estonia and setting the minimum number of monitoring sites for a specific habitat type to 75. Every year a selection of these sites was visited. For the new monitoring period 2019–2024 a new random selection was made, therefore each monitoring site is visited only once. The main purpose of forest habitat monitoring is to collect enough data from all forest habitat types in order to allow general statistical analysis of the status of these habitat types in Estonia. As monitoring of the status of specific habitat polygons is not its main purpose, it does not result in automatically updated status of the polygons in the EELIS database.

Information about the status of specific Annex I habitat polygons is more often derived from habitat inventories that are ordered by the Environmental Board and usually cover all habitat types of some protected areas. These inventories do not have a regular timestep and therefore all elements of the status assessment of some habitat polygons may not have been assessed at all yet, but some habitat polygons have been inventoried several times already. Additionally, data collected during monitoring is used for planning inventories and remapping of polygons, when they indicate a significant change (see Chapter 4.3.1.).

Vascular plants that are listed under protection categories I and II are also monitored under the biodiversity and landscape monitoring sub-programme (methodology available here, in Estonian). Assessed parameters include population size, population condition, community structure, threats and pressures etc. Monitoring sites are randomly selected from all known localities of specific species. During a six year monitoring cycle 12–30 localities (depending on the total number of known localities) of each species are visited. There are no permanent monitoring sites.

All monitoring data and reports from the national monitoring programme (except for species under protection categories I and II) is made publicly available through the Environmental Monitoring Information System (KESE). KESE does not yet include data from the habitat inventories ordered by the Environmental Board, but these data will also be included in KESE in the future.

As a conclusion, data on the status of Annex I habitats has been collected in Estonia with various means and will also be collected with various means in the future, but the coverage of status data has and probably will be non-uniform.

In Latvia, the national habitat monitoring programme and methodology for Natura 2000 is a sub-programme of the national environmental monitoring programme. The methodology (available here, in

Latvian) was developed in 2007 (methodology improved several times in the period from 2008 to 2013)[12]. It is aimed at identifying the changes in the conservation of all habitat types listed in Annex I of the Habitats Directive within the Natura 2000 network of protected nature areas. However, the monitoring is not aimed at certain areas, rather than habitat types. The first step in establishing the monitoring network is mapping of all protected habitats within Natura 2000 sites. The mapping is not completed yet, though most Natura 2000 sites (>95%) were mapped until 2012.

In the methodology, the habitat monitoring is based on two approaches: (1) field surveys (randomly selected transects aimed at covering all the habitat types listed in the Habitats Directive at national level, not the quality of certain habitat types in certain Natura 2000 sites) with filling in special field data forms, and (2) monitoring the changes in habitat patch areas and configuration using ortophotomaps and satellite images. However, since the habitat mapping has not been completed yet (in 2019–2020), and the national interpretation and monitoring methodology for EU importance habitats have been changed several times, a stable monitoring scheme does not exist. The transects established in 2010–2012 have been surveyed only once, thus long-term data of dynamics are not available. It is planned that the habitat monitoring system will be critically revised after completion of national-scale mapping within the Nature Census project (from 2022).

In Latvia, only two transects (GDTE habitat types 7160 and 9080*) are located in the Gauja basin, which is insufficient to cover this vast area. The current monitoring data are not representative neither for the Gauja basin, nor the particular habitat types at national or Gauja basin level.

The habitat monitoring programme under Natura 2000 habitat and plant monitoring sub-programme in Latvia should be revised and updated. Since the Nature Census project will provide new, up-to-date habitat data, it will allow better planning of monitoring and selecting representative areas to be monitored. Need for GDTE assessment is one of the aspects that should be taken into account when revising and improving the national Natura 2000 monitoring programme.

Additionally, GDTE related plant and invertebrate species listed in Annex II of the EU Habitats Directive are monitored under the same Natura 2000 monitoring programme with special methodologies that include estimation of population size, habitat condition, threats, etc. For very rare species, all locations are visited on a regular basis, whereas for more frequent species a certain number of sites are visited at least once in a 6-year period. Monitoring methodologies are available at the website of here (in Latvian). The species monitoring results might provide applicable information for GDTE quality assessment.

[12] http://biodiv.daba.gov.lv/fol302307/fol634754/natura-2000-teritoriju-monitoringa-metodikas-2007.-gada-redakcija, http://biodiv.daba.gov.lv/fol302307/fol634754/natura-2000-teritoriju-monitoringa-metodikas-2013.-gada-redakcija-aktualizetas/biotopi

5. Methodology validation and testing of best tools and parameters

5.1. Matsi spring fen pilot site in Estonia

5.1.1. Study site description

The Matsi spring fen was chosen as the pilot study site in the Estonian side of the transboundary Gauja-Koiva river basin. It is located in SE Estonia, Võru County, Rõuge Parish, Matsi village. The spring fen is situated in a sapping valley in the left slope of the valley of the Mustjõgi River, at an approximate elevation of 66 m asl (FIGURE 17). The valley has been incised in the plateau of sandstone of the Gauja formation. The Mustjõgi River (catchment area 1820 km2) is the largest tributary of the transboundary Gauja River basin (basin area 9800 km^2) in Estonia (FIGURE 17).

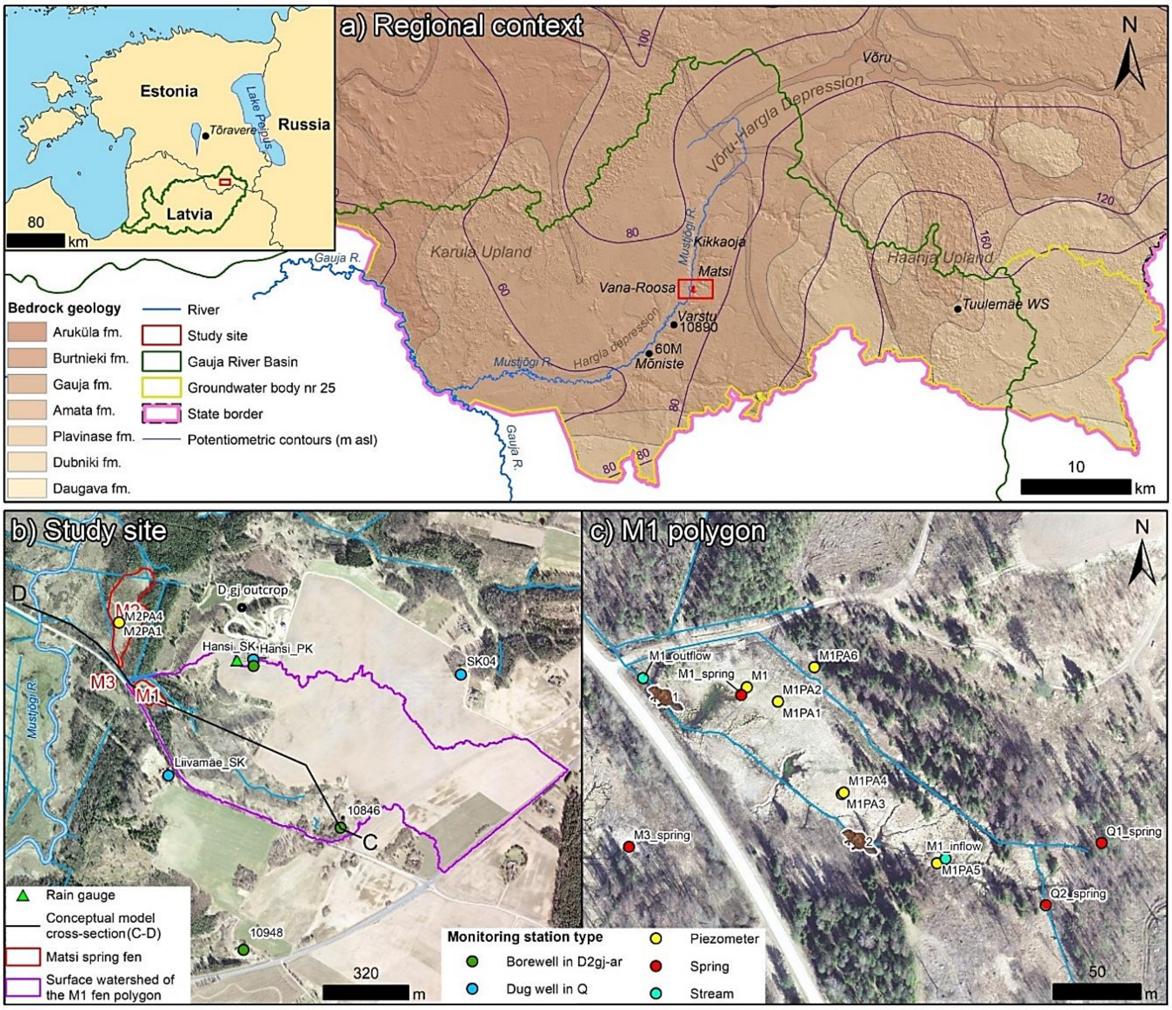


FIGURE 17 **The regional geology of the study site (a), map of the study site (b) and map of the M1 fen polygon (c); beavers in (c) indicate the location of the dams 1 and 2 (orthophoto by Estonian Land Boars, 2020)**

The fen consists of three polygons (M1, M2 and M3) (areas 0.7, 3.4, and 0.4 ha respectively) separated by road dams and ditches (FIGURE 17). The fen has been previously inventoried by Ilomets et al. (2012) as an important groundwater dependent terrestrial ecosystem with a very representative botanical community belonging to the Fennoscandian mineral-rich springs and spring fens (code 7160 Habitats Directive Annex I habitat type). There are tufa deposits present in the spring fen according to Lõokene (1968). It has been hypothesized that the Matsi spring fen is dependent on the Middle Devonian groundwater body in the Gauja-Koiva River Basin (Terasmaa et al., 2015; Marandi et al., 2019).

The hydrogeology of the study area is generally determined by the Middle Devonian (D_2) and the overlying Quaternary (Q) aquifer systems which constitute the Middle Devonian groundwater body in the Gauja-Koiva River basin (Marandi et al., 2019). As of 2019, the general status of the groundwater body has been assessed as good according to Marandi et al. (2019).

The Quaternary glacial, glaciolacustrine and glaciofluvial deposits host the Q aquifer system in the study area. Therefore, the lateral and vertical variation of the water-bearing properties of the Q aquifers is significant. Generally, the layers and lenses of more conductive (sand and gravel) glaciolacustrine and glaciofluvial sediments form the uppermost shallow Q unconfined or confined aquifer (if between till layers) that receives dominantly meteoric recharge. The shallow Q aquifer seemed to be perched on the underlying glacial till layer functioning as a weak aquitard (Kummisaar, 1967; Vallner et al., 1968; Jalast, 1976; Põldvere et al., 2016). The groundwater of the shallow Q aquifer is usually abstracted through the shallow dug wells (up to 6 m deep) of small households.

The Middle Devonian aquifer system (D2) is hosted by the sand- and siltstones with interlayers and lenses of clay of the Gauja, Burtnieki and Aruküla stages. Clayey interlayers are frequent in this aquifer system, likely forming several confined water bearing intervals of local distribution, which have not been delineated in detail. Thus, the aquifer system is generally regarded as the Gauja-Aruküla aquifer (D2gj-ar) (Verte, 1965; Perens & Vallner, 1997; Põldvere et al., 2016). The D2gj-ar aquifer is regionally recharged from the overlying Q and/or Snetnaya Gora-Amata aquitard of the Upper Devonian (D3) aquifer system in the Haanja Upland (Vallner et al., 1968; Perens et al., 2012; Põldvere et al., 2016; Marandi et al., 2019). The D2gj-ar aquifer discharges in the slopes of the Hargla/Mustjõgi valley (Viiding, 1959; Põldvere et al., 2016; Marandi et al., 2019). According to the borehole 60M in the Mõniste village, the D2gj-ar aquifer could be up to 188 m thick in total in the vicinity of the study area. The prevalent groundwater type in the D2gj-ar aquifer is Ca-HCO3, with the approximate TDS range of 200-600 mg/l (Perens et al., 2012; Marandi et al., 2019).

5.1.2. Methods

5.1.2.1. Geological and botanical survey

A shallow sediment coring survey with a Russian-type peat corer and spiral auger was carried out in the Matsi spring fen M1 and M2 polygons. In addition, a ground penetrating radar (GPR) survey was carried out in both polygons. The GPR profiles were tied with the shallow sediment coring results.

Borehole geophysical survey was conducted in one bore well 10846 opening the D2gj-ar aquifer. Survey provides a single continuous log of different physical-chemical parameters in a bore well, to aid in determining rock types and layer boundaries. Measurement methods included natural gamma ray (detector 125mm x 17.5mm Cs (Tl)), caliper (borehole diameter recorded by three mechanically coupled arms in contact with the borehole wall), water temperature and electrical conductivity measurements. All equipment used is manufactured by Robertson Geo.

Natural gamma ray measurements are based on scintillation gamma detectors. The detectors measure the natural gamma radiation released from potassium and the decay products of uranium and thorium in the surrounding rocks.

The vegetation of both M1 (n = 30) and M2 (n = 18) polygons was analyzed by 50 x 50 cm plots in mid-July 2019. In parallel to recording the coverage of all vascular plant and moss species, the mean water table in relation to the ground surface was measured from PVC tubes (d = 2.5 cm) installed to the plots. The patchy M1 polygon was divided roughly to subcommunities, and sampled by establishing at least three plots into each subcommunity. M2 as a more uniform area was sampled along two crossing transects, still aiming to cover the diversity of the area as much as possible. GPS coordinates of each plot were recorded as well.

5.1.2.2. Monitoring network and hydrological survey

To verify and characterize the interaction of the Matsi spring fen and the groundwater body, a comprehensive hydrodynamic and hydrochemical monitoring study was planned and carried out. The Matsi M1 fen polygon as the most representative polygon was chosen as the central object of study.

In total 22 monitoring stations were established (see FIGURE 17 and TABLE 8), including 9 shallow piezometers, 3 dug wells in the shallow Q aquifer, 4 borewells in D_2*gj-ar* aquifer, 3 springs and 3 stream stations. The selection and construction of the stations was generally based on the principle of a transect along the likely preferential flow path of the M1 spring fen polygon from the local recharge area to the discharge point of the polygon. During the study, additional monitoring stations (dug wells in the Q aquifer and springs etc.) were spontaneously included (also in the neighboring fen polygons M2 and M3) to better identify and delineate the potential end-members.

In the core of the M1 spring fen, two piezometer nests (M1PA2/M1PA1 and M1PA4/M1PA3 respectively) were constructed (for location see FIGURE 17), each consisting of one PVC piezometer in peat (screen depth <1.5 m) and one steel piezometer in below peat stratum (screen depth >1.5 m). Two steel piezometers (M1PA6 and M1PA5 respectively) were constructed at the edges of the M1 fen polygon, at the bottom of the N and S slope of the sapping valley respectively. In some cases, hybrid steel piezometers (M1 and M2PA1), screening both the peat and the below peat stratum, were installed (for location see FIGURE 17).

Twelve monitoring stations were equipped either with water level (HOBO OnSet U20L-01 and Van Essen TD-Diver 10 m), temperature (HOBO OnSet TidbiT v2) or electrical conductivity data logger (HOBO OnSet U24-001) (see TABLE 8). HOBO OnSet RG3-M Automatic rain gauge was installed next to the Hansi_sk monitoring station (FIGURE 17) during a period of above-freezing air temperatures (period). In addition, precipitation and air temperature data of the Tuulemäe weather station (WS) of the Estonian Weather Service was also used (Estonian Weather Service, 2020). The discharge of the M1 polygon was measured during each sampling campaign in the M1_outflow monitoring station with SonTek FlowTracker2 ADV.

TABLE 8

The list of monitoring stations in the Matsi spring fen study site (for locations see FIGURE 17)

#	Station	Datum elevation (m asl)	Depth (m)	Mean ± SD water level (m asl)	Station type	Group	Datalogger*
1	10846	93,43	110,00	72,31±0,09	Borewell in D2gj-ar	D2gj-ar	WL/T/
2	Hansi_pk**	90,00	65,00	70,68±0,07	Borewell in D2gj-ar	D2gj-ar	
3	10890	85,25	123,00	70,73	Borewell in D2gj-ar	D2gj-ar	
4	10948	90,64	76,00	70,72	Borewell in D2gj-ar	D2gj-ar	
5	Liivamae_sk	88,34	3,55	86,46±0,32	Dug well in Q	Q	
6	Hansi_sk	90,26	5,57	88,64±0,39	Dug well in Q	Q	WL/T
7	SK04	95,08	2,50	93,98±40,31	Dug well in Q	Q	
8	Q1 spring	69,00			Spring	Q	
9	Q2 spring	68,00			Spring	Q	
10	MI.	65,18	1,81	65,33±0,05	Piezometer (hybrid)	M1	WL/T
11	MI_spring	65,30			Spring	M1	T
12	M1PA1	65,54	2,87	65,7±0,09	Piezometer (below peat)	M1	WL/T
13	M1PA2	65,55	0,77	65,52±0,02	Piezometer (peat)	M1	WL/T
14	M1PA3	66,25	2,51	66,42±0,02	Piezometer (below peat)	M1	WL/T
15	M1PA4	66,27	1,13	66,28±0,01	Piezometer (peat)	M1	WL/T
16	M1PA5	67,77	2,03	67,21±0,13	Piezometer (fen edge)	M1	WL/T
17	M1PA6	67,31	2,55	66,50±0,02	Piezometer (fen edge)	M1	WL/T
18	MI outflow	65,30			Stream	M1	WL/T/SEC
19	MI Inflow	67,00			Stream	M1	
20	M2PA1	64,40	1,94	64,41±0,02	Piezometer (hybrid)	M2	WL/T
21	M2PA4	64,39	1,60	63,92±0,02	Piezometer (peat)	M2	
22	M3_spring	64,50			Spring	M3	

*WL - water level; T - temperature; SEC - specific conductance; **Water treated for iron removal

5.1.2.3. Water sampling campaigns and analysis

A total of 86 water samples (n=86) were collected during six sampling campaigns in the period of April 2019-March 2020 (TABLE 9).

TABLE 9

Description of sampling campaigns

Sampling campaign	Date	Collected samples
S1	April 2019	11
S2	May 2019	14
S3	July 2019	14
S4	September 2019	16
S5	November 2019	15
S6	March 2020	16
Total		86

Water samples were abstracted either by using a Solinst Peristaltic Pump Model 410 in case of shallow piezometers, dug wells and streams, or a Grundfos MP1 submersible pump in case of borewells. The in situ physico-chemical parameters (pH (accuracy ±0.2 units), temperature (accuracy ±0.2˚C), specific conductance (SEC) (accuracy ±0.001 m/Scm), dissolved oxygen (DO) (accuracy ±0.6), oxidation reduction potential (ORP) (accuracy ±20 mV)) were determined using a YSI Professional Plus Multi-Parameter water quality meter with an in-line flow cell. Carbonate hardness and alkalinity (as CaCO3) were determined in situ by volumetric titration using VISOCOLOR® HE Carbonate Hardness C 20 kit (accuracy±0.2 mmol/L H+).

The water samples were filtered in situ by using an in-line 47 mm filter assembly with a 0.45 µm RC (regenerated cellulose) filter bypassing the flow cell (except in the case of the Ptot samples). Each water sample was divided into six individual MilliQ and acid-washed Nalgene HDPE bottles to be shipped to the laboratories of the Tallinn University or University of Latvia. Following parameters were determined in the laboratories of the Tallinn University: major ions (Na^+; NH_4^-; K^+; Mg^{2+}; Ca^{2+}; F^-; Cl^-; Br^-; NO_3^-; SO_4^{2-}); nutrients (DOC; Ntot; Ptot); and in the laboratories of the University of Latvia: trace elements (Al; Ba; Fe; Mn; Sr; SiO_2) and stable isotopes (δ18O and δ2H).

Water samples for cations, anions, trace elements and stable isotopes (δ18O and δ2H) were collected in 15 ml HDPE bottles. The cation and trace metal samples were acidified down to pH <2 with concentrated HNO_3. The samples for cations and anions were analyzed in the laboratories of the School of Natural Sciences and Health at Tallinn University by using high-pressure liquid chromatography with SHIMADZU® RID-10A or CDD-10A conductivity detectors with Shodex IC YS-50 and Shodex IC SI-50 4E columns.

Trace element and stable isotope (δ18O and δ2H) analysis were carried out in the laboratories of University of Latvia, Faculty of Geography and Earth Sciences by using a Thermo Scientific Inc. iCAP7000 ICP-OES (inductively coupled plasma-optical emission spectrometry) at Department of Environmental, Sciences Environmentally quality laboratory and Picarro Isotopic Water Analyzer L2130-I (Cavity Ring-Down Spectroscopy) respectively at Department of Geology, Environmental Dating Laboratory. Isotope ratios are expressed in standard δ- notation relative to Vienna Standard Mean Ocean Water (V-SMOW). For creating the local meteoric water line (LMWL), the isotopic signature of precipitation from Tartu (Tõravere) (2013–2018), Riga (2016-2018) and Ramata (2016-2017) of the IAEA/WMO GNIP database were used (IAEA/WMO, 2020).

The water samples for dissolved organic carbon (DOC) and total nitrogen (Ntot) were collected in 60 ml HDPE bottles and acidified down to pH <2 with HCl. The water samples for total phosphorus (Ptot) were collected in 175 ml HDPE bottles and acidified down to pH <2 with H_2SO_4. The samples for DOC, DIC, Ntot and Ptot were analyzed in the laboratory of the Institute of Ecology at Tallinn University by using Analytik Jena multi N/C 3100 TOC/TNb analyzer, and in the case of Ptot, by using standard methods (Murphy & Riley, 1962; SFS 3025, 1986; SFS 3026, 1986; Method 365.3, 1978) and a Shimadzu UV-1800 spectrophotometer.

Ionic charge-balance error (CBE), saturation indices of calcite and dolomite (SIcalcite and SIdolomite respectively) and the partial pressure values of CO_2 (logPCO2) were calculated using the Geochemist Workbench 14® software.

5.1.3. Results and discussion

The following chapters characterize the Matsi spring fen study area while focusing primarily on the M1 polygon. Some data from the M2 and M3 fen polygons are also provided for comparison.

5.1.3.1. Vascular plant and bryophyte communities in the spring fen

M1 and M2 fen polygons are flat with minor microtopographic variation and aspect. Water table is generally close to the ground surface (TABLE 10), except in the wooded parts of the M2 polygon. The M1 polygon is open and devoid of tree cover; M2 is open in the central lower part, with a transition to spring fen forest in the higher part. The communities of both polygons are rich in species and generally characteristic to spring fens, which is supported by high water tables even during summer (TABLE 10). The most abundant vascular plants are *Menyanthes trifoliata* and several sedge species (e.g. *Carex rostrata, C. lasiocarpa, C. acuta*). *Rumex acetosa, Galium uliginosum* and *Myosotis scorpioides* can be frequently found as well.

TABLE 10

The number of vascular plant and bryophyte species, their cover (percent) and water table (cm) within the vegetation plots in Matsi spring fen

	M1 (n = 30)	M2 (n = 18)	Total (n = 48)
Vascular plant species	53	40	70
Bryophyte species	10	11	13
Total spp	63	51	83
Vascular plant cover	42	50	44
Bryophyte cover	34	79	51
Total cover	76	129	95
Mean water table	-3.5	-8.8	-5.5
Min water table	-18	-30	-30
Max water table	+17	0	+17

Five vegetation clusters were distinguished in the M1 polygon and two in the M2 polygon (FIGURE 18). The water level in the M1 polygon was higher than in M2 (TABLE 10). The plant communities in the northern and central parts of the M1 polygon were similar to the open part of the M2 fen polygon. *Menyanthes trifoliata* and typical spring fen bryophytes (e.g. *Plagiomnium elipticum*) dominated in both. *Saxifraga hirculus* – a rare and protected (II category) spring fen specific plant species – was present in that part of the fen. In the southern and eastern part of the M1 polygon, but also in its northern margin, the plant communities were not characteristic of pristine spring fens. In the south-eastern end of M1 the full domination of *Carex acuta* prevents the formation of moss layer and the overall plant diversity is low. In other parts of M1, *Carex paniculata* tussocks have rarely left much space to other species, and moss layer is absent. These issues could be due to high water level in the fen, but also from the unfavorable nutrient loading in water for the natural spring fen communities.

The M2 polygon vegetation was rather uniform, and thus subclustered to open (lower) and wooded (higher) parts. Wooded cluster also included the transitional areas from the open to wooded area. Both sub-clusters of M2 are clearly dominated by *Menyanthes trifoliata* with a frequent co-domination of *Epipactis palustris*, a protected species (III category). Additionally, *Galium uliginosum, Carex rostrata, Festuca rubra, Poa palustris* and *Agrostis stolonifera* can be frequently found from both sub-clusters of M2. Whilst *Lychnis flos-cuculi* is frequent in open clusters, *Equisetum fluviatile, Pedicularis palustris* and *Cirsium palustre* are more abundant in wooded or transitional areas, where the ground surface elevation is higher and water table is lower. The Moss layer was almost continuous and consisted mostly of typical spring fen species, e.g.

Plagiomnium ellipticum, Paludella squarrosa and *Calliergonella cuspidata*. The biggest disturbance in the M2 polygon seemed to be drainage surrounding the polygon that had initiated the expansion of the tree cover at the edges. The ditches are old but still functioning. The higher areas at the northern part of the fen are indeed still open at the historical maps (Estonian Land Board, 2020).

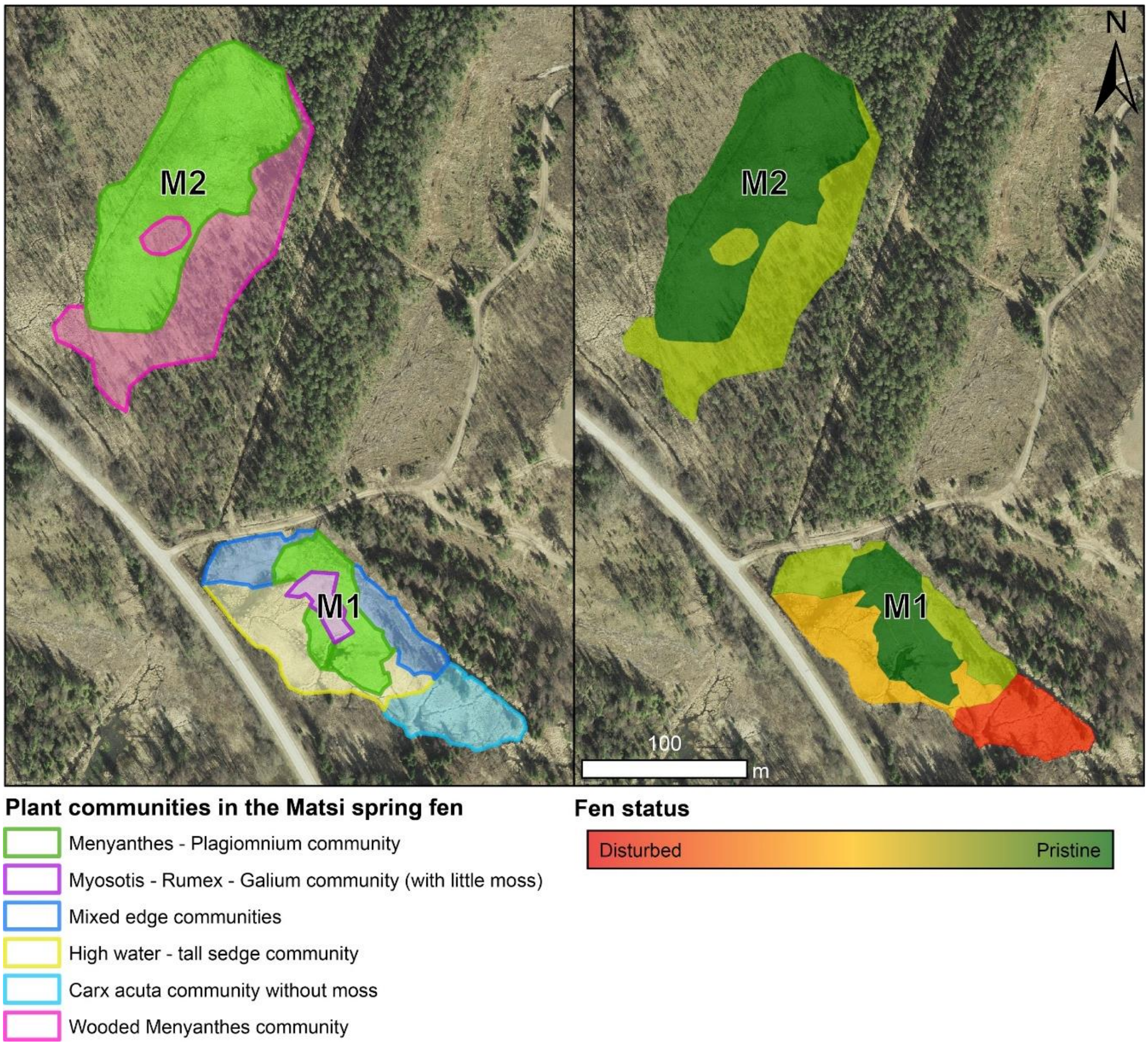


FIGURE 18 **Plant community clusters (a) and fen ecosystem status (b) in the Matsi spring fen polygons M1 and M2**

According to the inventory of peatland habitats in Estonia (Paal & Leibak, 2011) **the conservation value of Matsi spring fen was good.** The evaluation of the inventory is, however, supported only partly by our data on the community composition, plant cover and the water table of the spring fens.

5.1.3.2. General hydrodynamics

Water level and temperature variations were automatically monitored in eleven monitoring stations in the Matsi spring fen study area from November 2018 to March 2020 (FIGURE 19). The hydraulic head variations in M1 and M2PA1 monitoring stations were monitored from September 2018. During the study period the mean annual sum of precipitation in Estonia varied from 508 mm in 2018 to 671 mm in 2019 respectively Estonian Weather Service (2020). Compared to the national long-term norm of precipitation (672 mm), 2018 was one of the driest years since 1964 according to Estonian Weather Service (2020). The dryness of 2018 was also reflected in the modest spring flood of 2019 (FIGURE 19). In contrast to the near-normal annual rainfall sum in 2019, the year was abnormally warm (2019 annual mean 7.6 °C vs. long-term mean 6 °C), so there was almost no significant snow cover in the winter and most of the precipitation fell as rain. The autumn flood of 2019 expanded into the spring of 2020.

The hydraulic heads in the D_2gj-ar borewells generally fluctuated within 30 cm throughout the study period (TABLE 8). The initial hydraulic head time series observed in the borewell 10846 representing the D_2gj-ar aquifer, referred to the aquifer being confined. (FIGURE 19). The elevation of the potentiometric surface of the D_2gj-ar aquifer in the Matsi spring fen study was estimated to range from 74 to 62 m asl, with a east-west directional slope towards the valley of the Mustjõgi River. The shallow Q aquifer represented by the dug wells, was found to be perched in the plateau surrounding the sapping valley. The water table surface of the Q aquifer ranged from 84 to 92 m asl. As previously stated in the study area chapter, the shallow Q aquifer was expected to lie on a glacial till layer. As seen in the case of the Hansi_sk in FIGURE 19, the amplitude of water table fluctuations in the Q aquifer could reach approximately 1,5 m.

The hydraulic heads in the M1 and M2 fen polygon core piezometers generally fluctuated within 10 cm throughout the study period. Head fluctuations were somewhat bigger in the fen edge piezometers M1PA5 and M1PA6, where the range of fluctuation was about 15-20 cm. In the case of the two M1 fen polygon piezometer nests, the hydraulic heads in the below peat stratum piezometers (M1PA1 and M1PA3) were above the ground surface. Moreover, the heads in the below peat stratum were also higher compared to the heads in the peat piezometers (M1PA2 and M1PA4). This was likely due to artesian flow conditions below the fen that could be associated with the underlying confined D_2gj-ar aquifer.

The water level in the M1 fen polygon was at times regulated by beaver dams as seen in FIGURE 19. Two larger beaver dams can be identified in the M1 area (see FIGURE 17). In most cases, the removal of the dams by the landowner caused the water levels to drop abruptly. Only the dam 2 in the upstream part of the fen between M1PA5 and M1PA3/M1PA4 remained intact until the end of the monitoring period (FIGURE 19). During the study period, the water level in the M1_outflow fluctuated within 70 cm. The damming seemed to have little effect on the SEC of the M1 fen polygon outflow at M1_outflow (mean ± SD = 480±47 µS/cm) (FIGURE 19).

The discharge in the M1_outflow was relatively stable throughout the study period (mean ± SD =13,8±2,3 l/s), reaching up to 18 l/s during high flow conditions in March 2020 (S6) (FIGURE 19). The minimum discharge (11,7 l/s), being only 15% lower than the mean, was observed during low flow conditions in July 2019 (S3). The beaver dams seemed to have little or no long-term effect on the outflow discharge of the M1 fen polygon. The stability of discharge values also indicated that the spring fen is likely to be fed by an artesian aquifer.

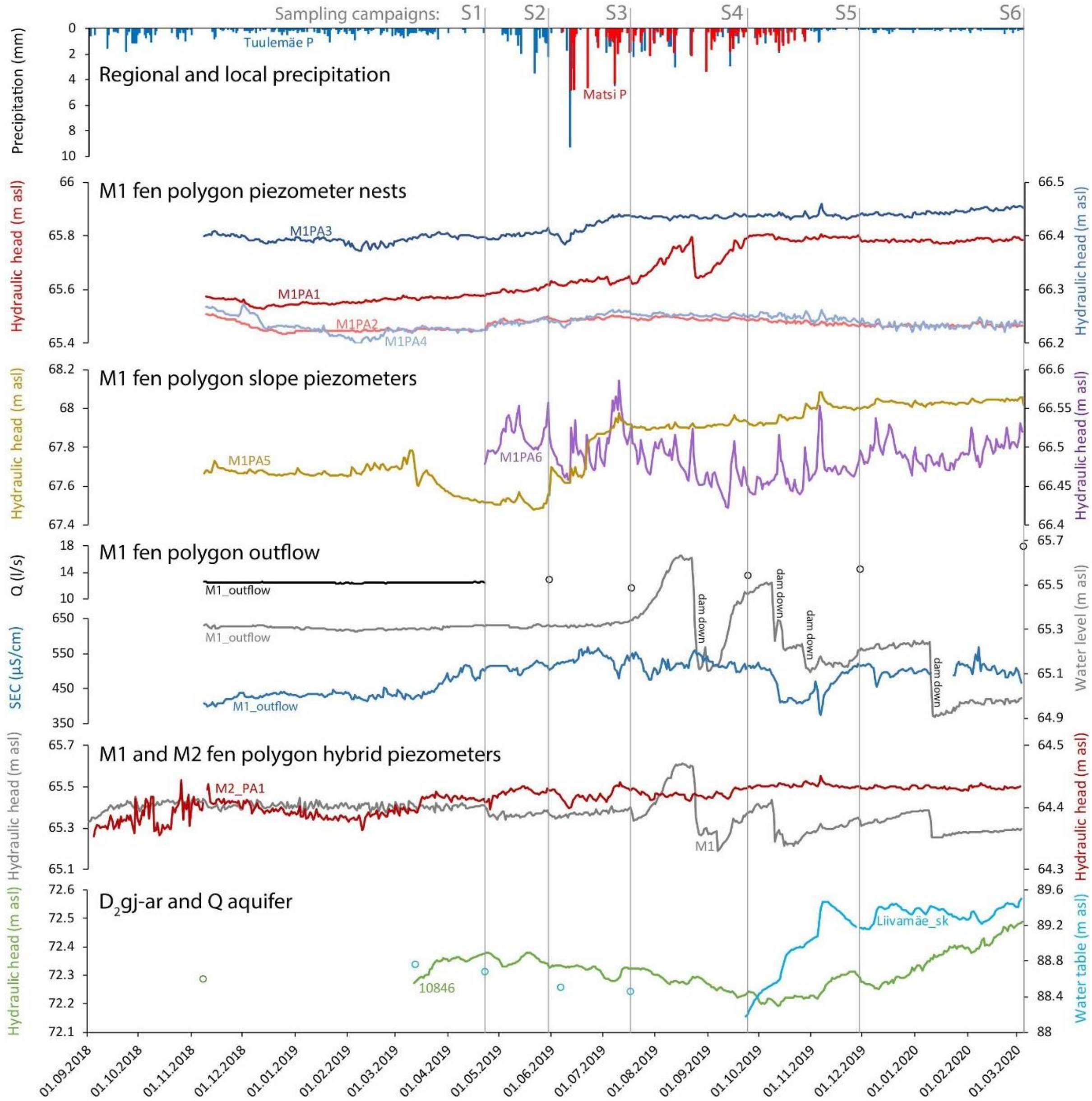


FIGURE 19 **Daily time series of regional (Tuulemäe WS) and local (Matsi) precipitation, hydraulic head, water table and water level fluctuations in the selected monitoring stations in the Matsi spring fen study area (for location see FIGURE 17)**

5.1.3.3. Hydrochemical characteristics

The charge balance error (CBE) threshold value applied for the water samples was ±10%. An exception was made for two water samples with CBE values of -11.1 and -11.2% respectively. The mean ± SD CBE of the assessed water samples was -0,6±3,9%.

The major anion composition of the studied waters was dominated by the HCO_3^-, followed by (in the decreasing order of abundance) Cl^- > SO_4^{2-} > NO_3^- > Br^- > F^- (FIGURE 20, FIGURE 21 and FIGURE 22). Among cations, Ca^{2+} was the most abundant, followed by (in the decreasing order of abundance) Mg^{2+} > Na^+ > K^+ > NH_4^+ (FIGURE 20 and FIGURE 22). The abundance of other reported minor components occurred in the order of SiO_2 > Fe > Ba > Mn > Sr > Al (FIGURE 21). Dominant water type in the study area was Mg-HCO3 to Ca-Mg-CO3 (FIGURE 22).

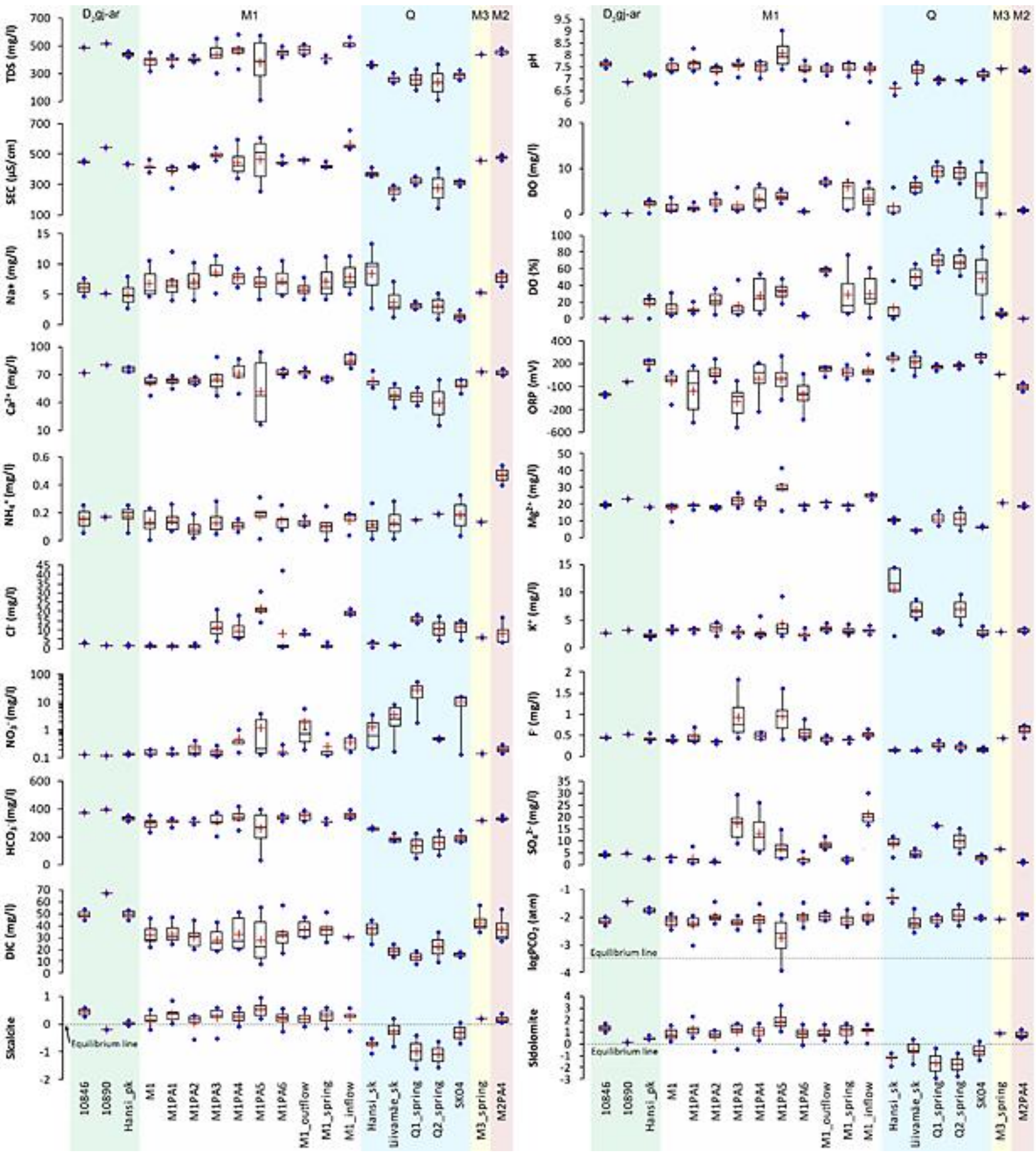


FIGURE 20 **Boxplots of physicochemical parameters, major ions and saturation indices of carbonate minerals in the studied water samples by monitoring stations (see TABLE 8)**

The mean ± SD TDS concentration of the studied waters was 402±95 mg/l, and the respective range 105-577 mg/l (FIGURE 20). The respective mean ± SD value of SEC was 421±91 µS/cm and the range 145-659 µS/cm (FIGURE 20). The highest mean values of TDS and SEC were observed in the deeper D_2gj-ar borewells (10890 and 10846) and the upstream monitoring stations of the M1 fen (M1_inflow, M1PA5, M1PA4 and M1PA3) (FIGURE 20). The lowest TDS values were generally observed in the Q dug wells (Hansi_sk, Liivamae_sk and SK04) and in the Q1_ and Q2_springs but occasionally also in M1PA5 located in the margin of the M1 polygon (FIGURE 20). The TDS and SEC values (Figure 3 and 5) of the monitoring stations in M2 (M2PA4) and M3 (M3_spring) fen polygons roughly fell in line with those of the D_2gj-ar group. As a generalization, the stations with higher TDS had higher minor component (Fe, Ba, Mn, Sr) concentrations, and on the contrary, the ones with lower TDS had higher macronutrient concentrations (DOC, N_{tot}, P_{tot}, K^+, NO_3^-).

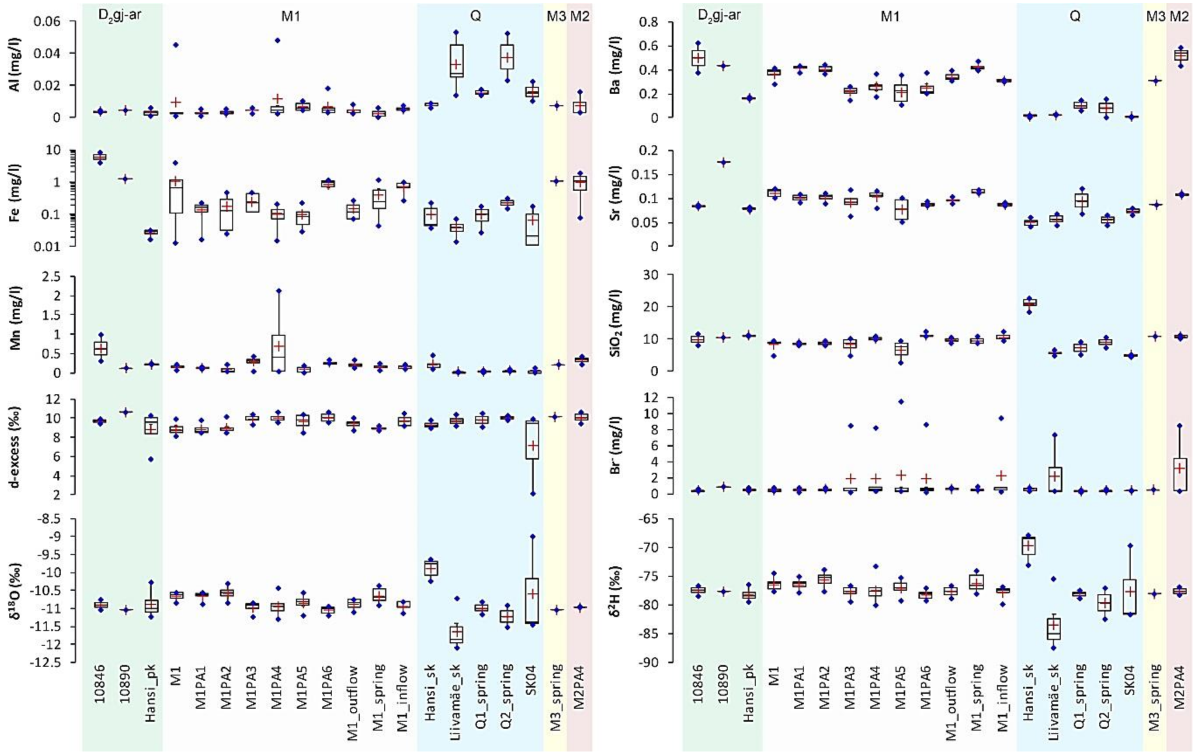


FIGURE 21 **Boxplots of minor ion concentrations, stable isotopes signatures in the studied water samples by monitoring stations (see TABLE 8)**

The mean ± SD of pH of the studied water was 7.4±0.4, and the range from 6.3 to 9 (FIGURE 20). Lower values were observed in the Hansi_sk, a dug well screening the shallow Q aquifer, and in the Q1_ and Q2_spring (TABLE 8 and FIGURE 20). Based on Knutsson (1994), the acidification of shallow groundwater can occur due to numerous natural factors, namely due to the acidity of precipitation, soil respiration, the dissociation of humic acids and the occasional lack of carbonate minerals in the spatially variable, primarily terrigenous, Quaternary aquifer material. However, the acidification can rise also from the use of fertilizers (Jolley & Pierre, 1977; Knutsson, 1994) which is likely on the agricultural lands on the plateau.

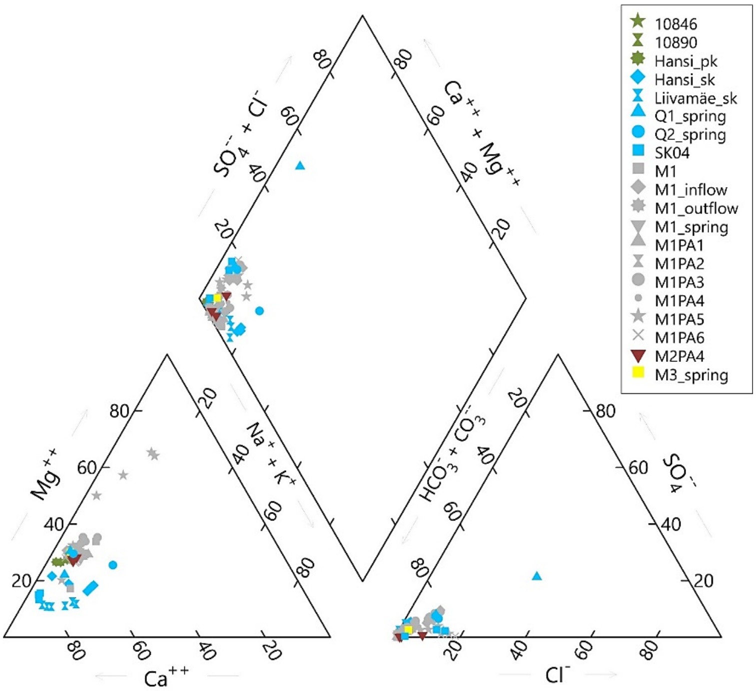


FIGURE 22 **Piper plot showing the major ion composition of the studied water samples**

The dissolved oxygen (DO) concentration of the studied waters was generally moderate to low (mean ± SD = 3,3±3,4 ppm) (FIGURE 20). Conditions were predominantly oxic in the Q aquifer dug wells, Q1_spring, Q2_spring and in surface streams; and occasionally or constantly suboxic to anoxic in the D_2gj-ar boreholes, M1 and M2 fen polygon piezometers.

The sampled waters were predominantly of the Ca-HCO_3 type, whereas the water in M1PA5 (for location see FIGURE 17), a shallow piezometer under the slope of the sapping valley, was of the Mg-HCO_3 type (FIGURE 22). Despite the fact that siliciclastic material prevailing in the terrigenous aquifer matrix, carbonate minerals were the major contributor to the hydrochemistry of the waters in the study site. The occurrence of carbonate cement, primarily dolomite and in less cases calcite, is common in the Middle Devonian siliciclastic rocks (Kleesment, 1995; Kleesment & Mark-Kurik, 1997; Kleesment & Shogenova, 2005; Kleesment et al., 2012). Moreover, carbonate material can also be abundant in Quaternary glacigenic sediments (Raukas, 1978; Räägel, 1997; Rattas & Kalm, 2000, Rattas et al., 2014).

5.1.3.4. Nutrient dynamics

The primary macronutrients (DOC, N_{tot} and P_{tot}) observed in the monitoring stations are presented in FIGURE 20 and FIGURE 23. The mean macronutrient values of the monitoring station groups (TABLE 8) are shown in FIGURE 23. The mean ± SD concentration of DOC of all of the studied water samples was 3,3±3,9 mg/l, with the respective value of 1,8±2,6 mg/l in the M1 fen polygon group (FIGURE 23). The highest and the lowest mean ± SD DOC concentrations (8,4±2,9 and 0,03±0,07 mg/l) were observed in the Q and D_2*gj-ar* groups respectively. Over time, the mean and median values of DOC in the M1 polygon lowered gradually (FIGURE 24).

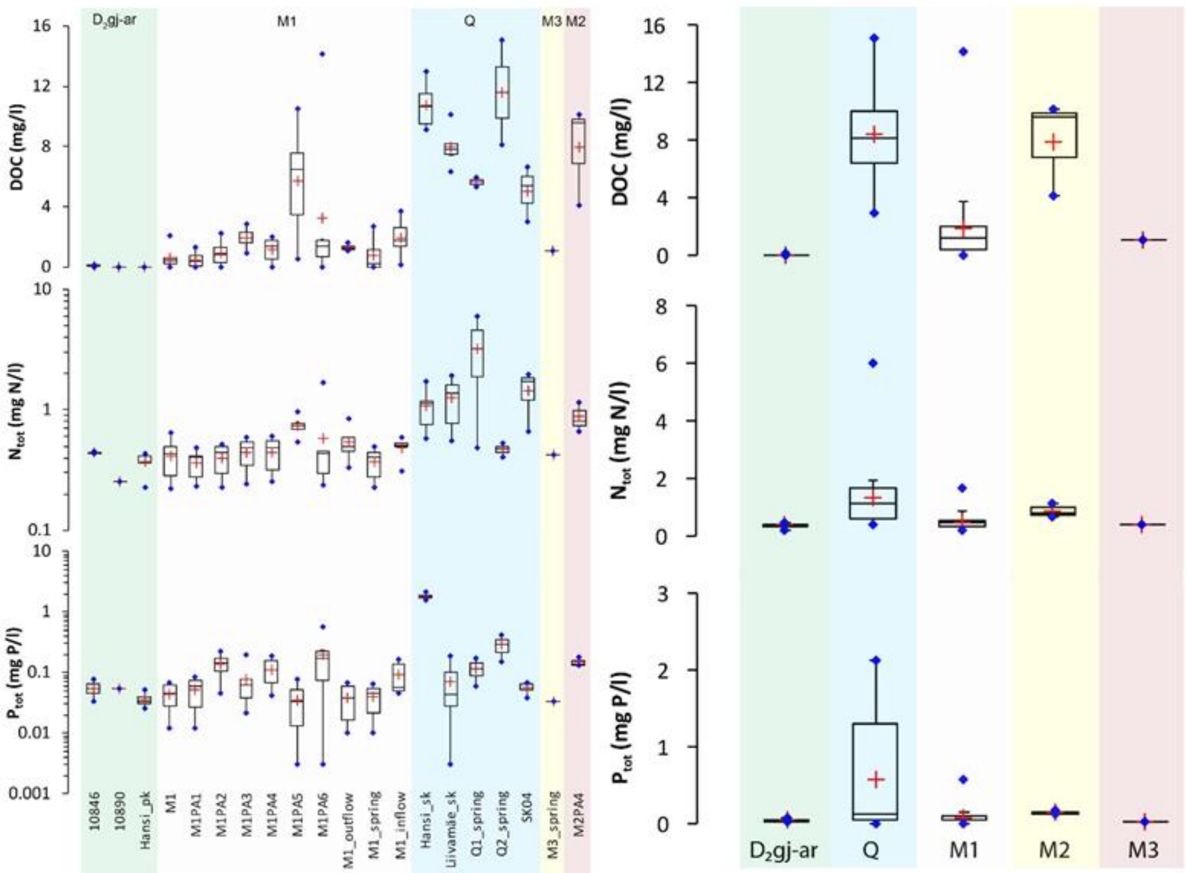


FIGURE 23 **Boxplots of assessed macronutrients (on the left) and their mean concentrations (on the right) in the studied water samples by monitoring stations (see TABLE 8)**

The mean ± SD concentrations of N_{tot}, P_{tot} of all of the studied water samples (n=68) were 0,67±0,71 mg N/l and 0.19±0.42 mg P/l respectively. The highest and the lowest mean ± SD concentrations for both N_{tot} and P_{tot} were observed in the Q (1,4±1,3 mg N/l and 0,59±0,8 mg P/l) and D_2*gj-ar* (0.3±0.1 mg N/l and 0,04±0,02 mg P/l) groups respectively (FIGURE 23). The respective mean ± SD N_{tot} and P_{tot} values for the M1 fen polygon group (0,5±0,2 mg N/l and 0,08±0,09 mg P/l respectively) fell in-between the latter two. Within the M1 fen polygon, the highest N_{tot} and P_{tot} mean values were generally observed in the slope piezometers (M1PA5 and M1PA6), and/or the M1_inflow and the neighboring downstream monitoring stations (FIGURE 23).

According to thresholds set for Estonian fluvial water bodies the M1 fen polygon was in a very good trophic status regarding mean N_{tot} values, and from very good to very bad status regarding mean P_{tot} values (FIGURE 24). In three cases, the mean N:P ratios of M1 polygon were below the 16:1 ratio defined by Redfield (1958), and only in one case (in September 2019 (S4)) the median ratio was above the 16:1 line, indicating prevailing nitrogen limitation and phosphorus excess (FIGURE 24). According to Reddy & DeLaune (2008), the general cause for nitrogen limitation in wetlands is usually increased upland loading of phosphorus, commonly due to the use of fertilizers.

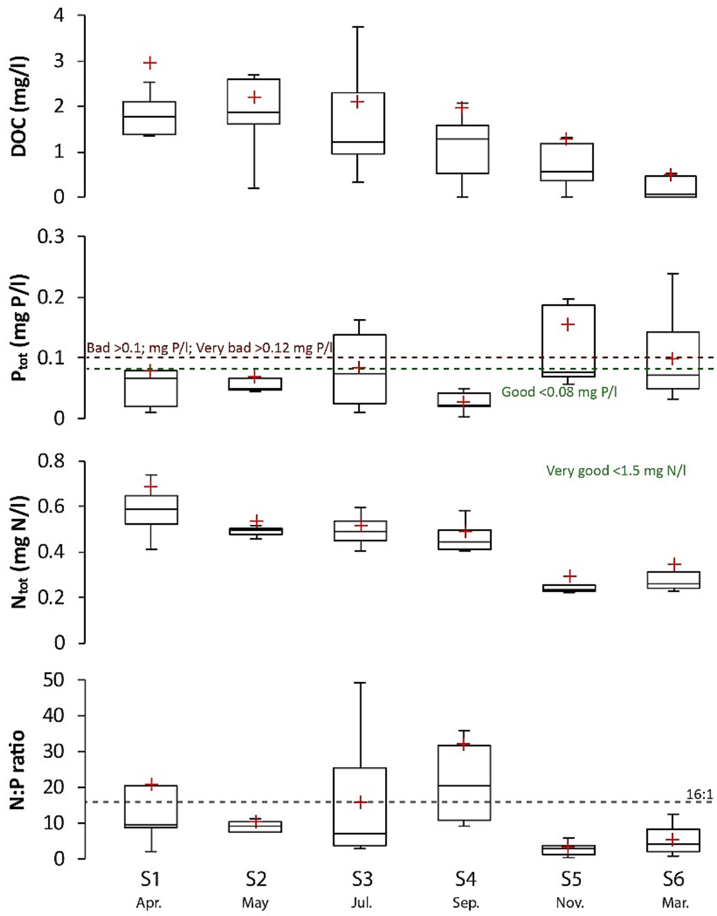


FIGURE 24 **Boxplots of mean concentrations of DOC, N_{tot}, P_{tot} and the N:P ratio in the M1 fen polygon, grouped by sampling campaigns S1-S6 (see TABLE 9). Dashed lines indicate the ecological status thresholds for Estonian fluvial water bodies.**

As a generalization, the highest and the lowest mean values of the assessed macronutrients were observed in the Q and D_2gj-ar group respectively. Similar trends as in the case of DOC, N_{tot} and P_{tot} could be also seen when assessing the K^+ and Cl^- values amongst the station groups. Based on the above, a general hypothesis can be proposed that the Q group was the primary source of macronutrients in the Matsi spring fen study site. The latter is evident due to intensive agricultural activities carried out on the plateau surrounding the sapping valley. To a lesser extent, for example regarding the concentrations of Cl^-, the impact from road salting could also be the case. The highest nutrient concentrations within the M1 fen polygon group were generally observed in the slope piezometers (M1PA5 and M1PA6), and in the monitoring stations capturing the upstream inflow from the sapping valley and the upland (M1_inflow, Q1_spring and Q2_spring). The patterns of the macronutrient dynamics also agree with the specificities of the distribution and status of plant communities in the M1 fen polygon. Communities that are characteristic to nutrient rich environments rather than pristine spring fens, were usually found in NE-E part (near the slopes and inflow) of the M1 polygon, but also at the SW margin, along the open water section channeling the upstream inflow to the M1_outflow (FIGURE 18).

5.1.3.5. Water stable isotopes

The mean ± SD of $\delta^{18}O$ and δ^2H values of the sampled waters in the Matsi spring fen study were -10,8±0,5‰ and -77,2±3,3‰ respectively (FIGURE 21, FIGURE 25), falling close to the mean $\delta^{18}O$ and δ^2H values for precipitation of the Tõravere, Riga and Ramata GNIP stations (-10,5‰ and -75,6‰ respectively). For better overview, the monitoring stations were divided into initial groups based on presumed stratigraphy (as seen in TABLE 8).

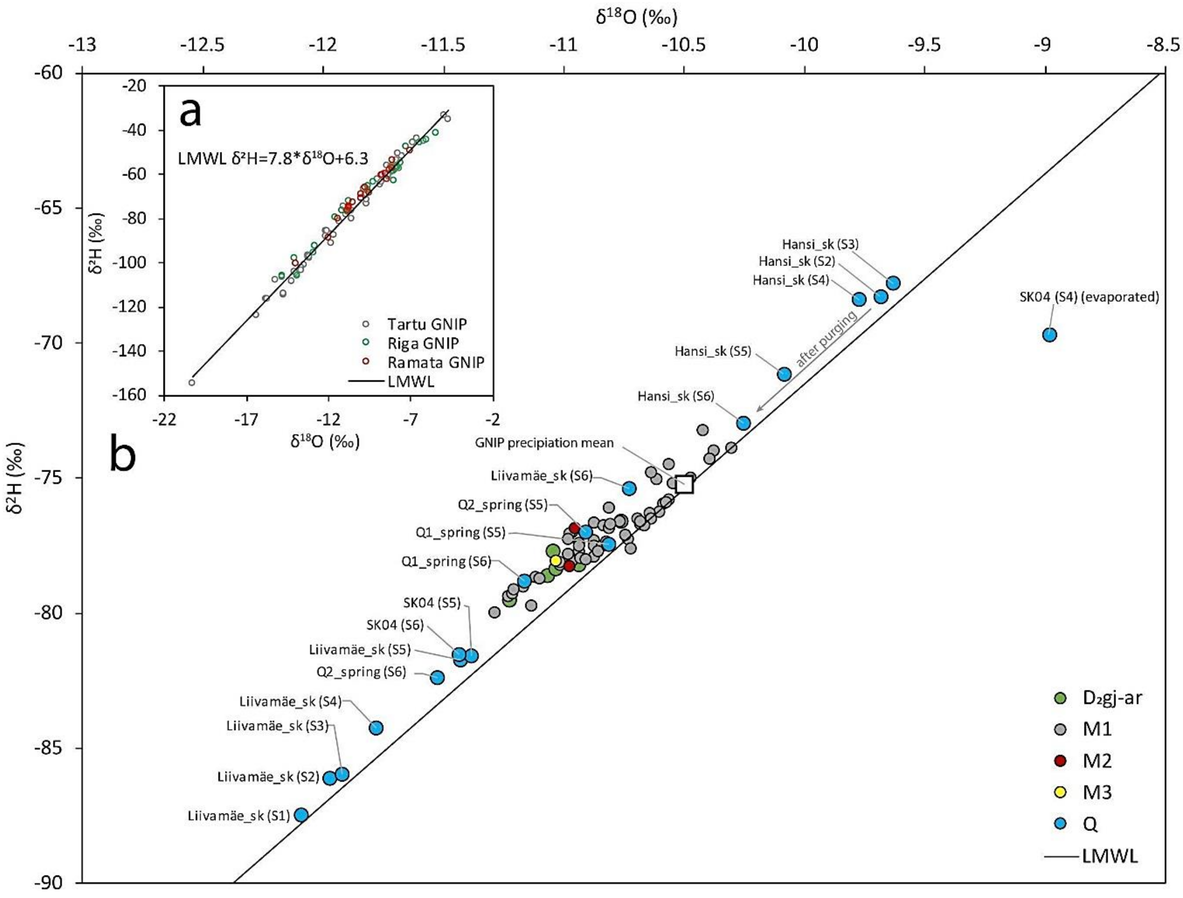


FIGURE 25 The isotopic composition ($\delta^{18}O$ vs. δ^2H) of the precipitation and the LMWL based on the data of the IAEA/WMO GNIP stations (a). The isotopic composition ($\delta^{18}O$ vs. δ^2H) of the studied waters in the Matsi spring fen study site (b).

The studied waters generally plotted along the LMWL (defined by $\delta^2H=7,8*\delta^{18}O+6,3$; mean) compiled from the $\delta^{18}O$ and δ^2H precipitation data of the IAEA/WMO GNIP (IAEA/WMO, 2020) Tartu, Riga and Ramata stations, indicating their meteoric origin in general. The latter was also supported by the majority of deuterium excess (Dansgaard, 1964) values (shown in FIGURE 21) falling in the range of 8.1-10.7‰, thus agreeing with regional value ranges for shallow groundwater estimated by Kortelainen & Karhu, (2004) and

Raidla et al. (2016). In two cases deuterium excess values lower than 8‰ were observed. In some Q group monitoring stations, significant deviation from the LMWL was observed, most likely due to evaporation of the shallow Q groundwater.

5.1.3.6. Hydrogeochemical conceptualization of the study site

Clustering approach

To define and characterize the major end-members involved in the hydrochemical system of the Matsi spring fen study site, a combined multivariate statistical analysis approach, including agglomerative hierarchical cluster analysis, discriminant analysis and principal component analysis, was conducted on the hydrochemical data. A more detailed overview of the combined statistical analysis methodology is planned to be published in a research article. The main emphasis here was placed on the M1 fen polygon. In total 86 observations with 28 variables (pH; SEC; ORP; Na^+; K^+; Mg^{2+}; Ca^{2+}; F^-; Cl^-; Br^-, SO_4^{2-}; HCO^{3-}; DOC; DIC; N_{tot}; P_{tot}; Al; Ba; Fe; Mn; SiO_2; Sr; TDS; SIcalcite; SIdolomite; $logPCO_2$; $\delta^{18}O$; δ^2H) were assessed in the further multivariate statistical analysis.

In the agglomerative hierarchical cluster (AHC) analysis, the assessed monitoring stations were clustered into three clusters, based on automatic truncation: Q, A and B clusters respectively (FIGURE 26). The Q cluster included all the monitoring stations that were initially grouped into the Q group (TABLE 8), The A cluster included the M1 group stations located in the downstream half of the M1 fen polygon and the D_2*gj-ar* group stations. The B cluster included the monitoring stations in the upstream half of the M1 fen polygon.

The AHC-derived three cluster distribution (FIGURE 26) was validated by carrying out discriminant analysis (DA). The discriminant model classified 98% of the assessed observations into the correct AHC-derived clusters (FIGURE 26). This means that the AHC-derived cluster distribution could be assumed to be reliable according to the DA.

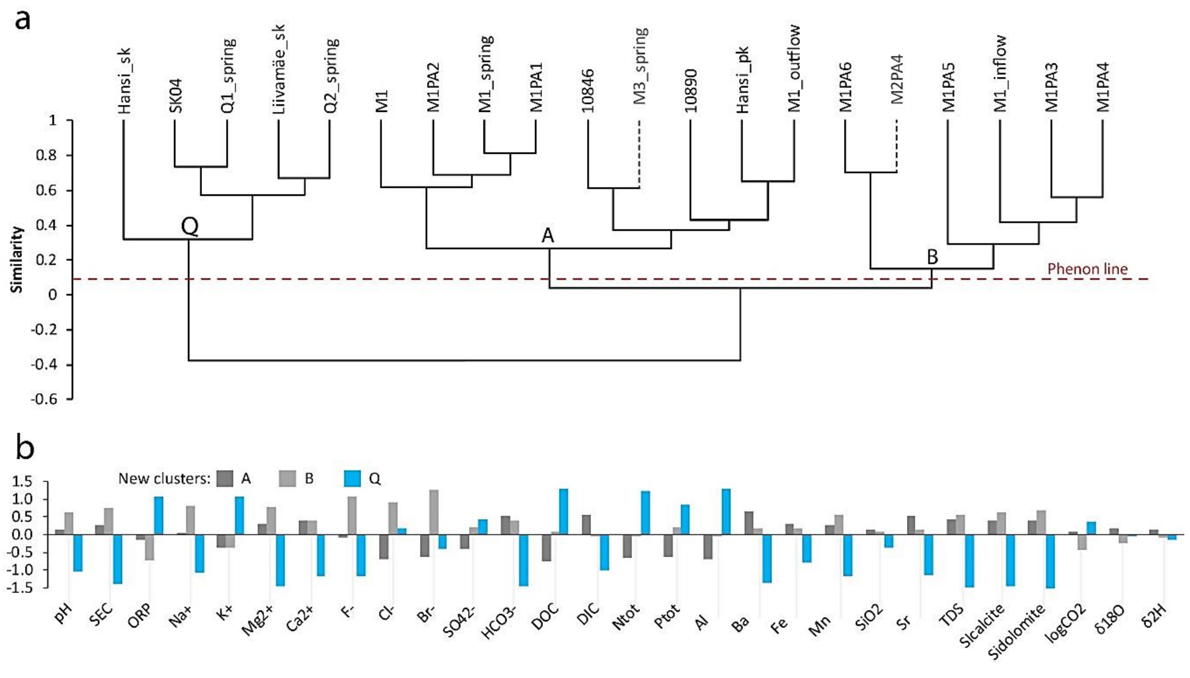


FIGURE 26 **Dendrogram of the Matsi spring fen study site monitoring stations (n=20) clustered by the mean values of 28 physicochemical parameters obtained from agglomerative hierarchical cluster analysis (a). The red phenon line represents the clusters delineated by automatic truncation. The M3_spring and M2PA4 are also presented for comparison. Class centroids for the cluster distribution (b).**

The Q cluster observations were characterized by higher ORP and $logPCO_2$ values (oxic conditions), high macronutrient loading (DOC, N_{tot}, P_{tot} and K^+), but also higher Al concentrations. In nature Al is known to originate primarily from gibbsite dissolution, and its mobility is related to soil acidity and organic matter abundance. Like many other trace metals, Al forms organometallic complexes with organic matter enhancing its mobility, and thus could be abundant in shallow groundwater (Budd et al., 1981). By all means, the Q cluster observations were strongly affected by the agricultural activities on the plateau.

The A and B cluster observations generally featured higher mineralization (higher TDS), were rather anoxic, more saturated with respect to carbonate minerals and featured higher concentrations of ions associated with longer residence time in the aquifer (Mg^{2+}, Ba and Sr), when compared to the Q cluster. Furthermore, the A and B clusters allowed the M1 fen polygon to be divided into two (FIGURE 26). This was also evident based on the initial assessment of macronutrient dynamics in the previous chapters. In addition to the higher macronutrient concentrations, the upstream B cluster observations also featured elevated Mg^{2+}, halogen (F^-, Cl^- and Br^-) and SO_4^{2-} concentrations when compared to the downstream A cluster observations. The origin of elevated concentrations of halogens (F^-, Cl^- and Br^-), Mg^{2+} and SO_4^{2-} could be explained by the agricultural activities on the plateau (Adriano and Doner, 2015; Härdter et al., 2005). The lower presence of Mg^{2+}, halogens and SO_4^{2-} in the Q cluster, when compared to the B cluster, could be justified by the fact that the dug wells primarily forming the Q cluster, were mostly located upstream of the arable lands that were surrounding the sapping valley.

Moreover, it is likely that the arable lands also had a hydro-melioration system, which in turn would enhance the drainage of soil water/surficial groundwater to the sapping valley. Thus, in addition to the typical groundwater of the Q aquifer (Q cluster), the sapping valley and consequently the upstream part of the M1 spring fen polygon would exclusively receive anything applied to the arable fields, most likely in a form of dilute interflow component as occasionally observed in M1PA5 and Q1_spring.

The A cluster combined the observations of the downstream stations of the M1 fen polygon with the D_2gj-ar observations (FIGURE 26). The A cluster also included the observations of the M1_outflow monitoring station, which represented a mix of all the inputs to the M1 fen polygon. This suggests that the mean water quality of the M1 fen polygon as a whole (as represented by the M1_outflow) seems to be more affected by the D_2gj-ar aquifer groundwater.

End-member mixing analysis

To verify and quantify the presumed mixing of three hydrochemical end-members (D_2*gj-ar* aquifer, Q aquifer and dilute interflow) and the evolution of the final water quality in the M1 spring fen polygon outflow (M1_outflow), the end-member mixing analysis (EMMA), as proposed by Christophersen and Hooper (1992) was performed. Based on the list of physico-chemical parameters that were previously included in the AHC and DA, six conservative or semi-conservative tracers (Ba, Sr, F^-, $\delta^{18}O$, SiO_2, Al) characteristic for the presumed end-members were selected for further analysis.

The results of EMMA suggest that on average the D_2*gj-ar* aquifer contributed 67±19% of the water to the mixture in the M1_outflow (FIGURE 27). The respective mean ± SD values for the Q aquifer and interflow component were 15±12% and 18±18%. The latter two end-members could be generally addressed together as the baseflow and the interflow components of the shallow Q aquifer. The proportional contribution of the D_2*gj-ar* end-member was the greatest during the low-flow period in July and September (S3-S4) and the lowest in the case of the high flow period in March (S6). Although D_2*gj-ar* is the dominant contributor to the M1 fen polygon, the Q aquifer, when the baseflow and interflow components were combined, may provide up to 59% of the water to the polygon.

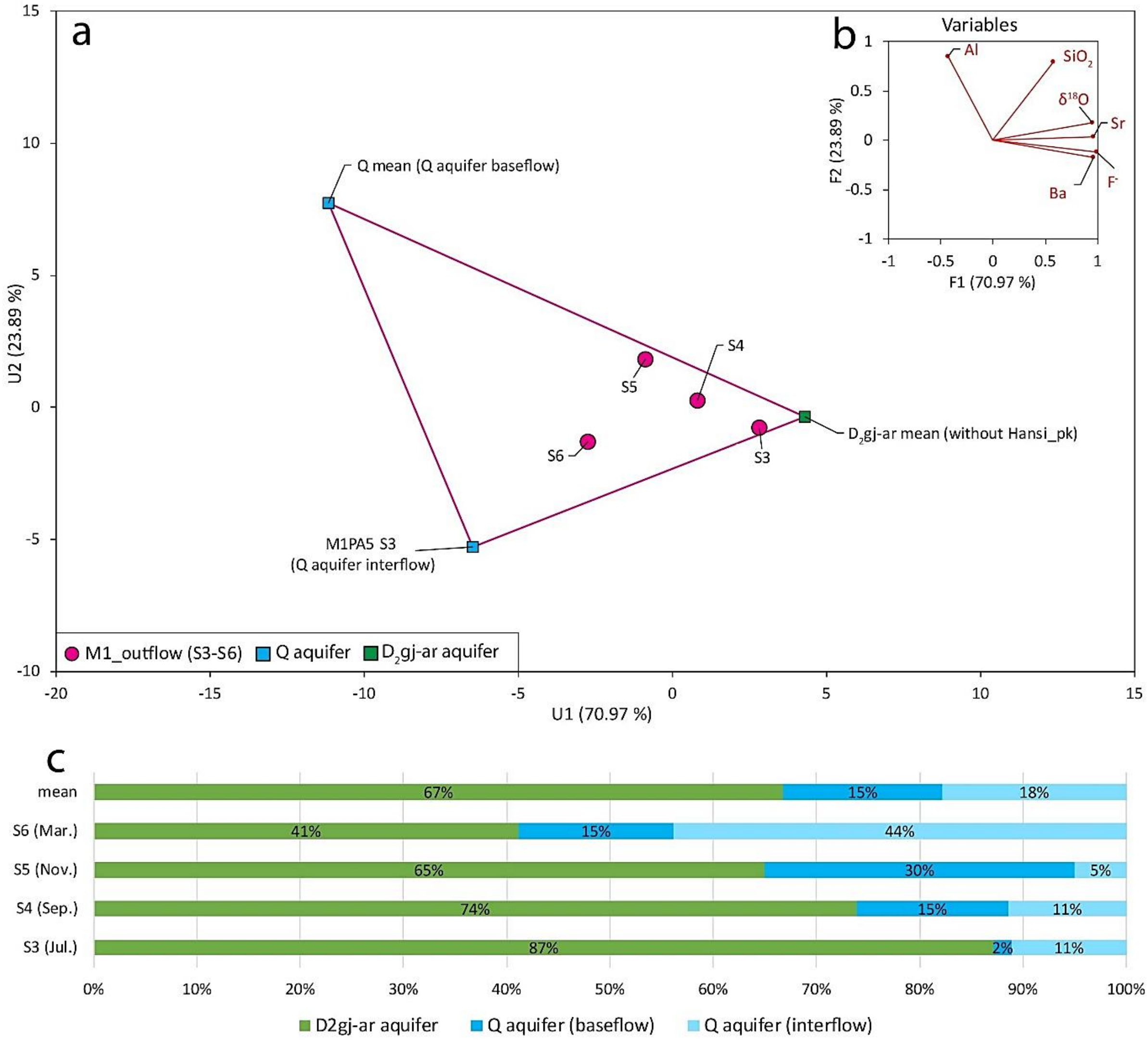


FIGURE 27 **The M1_outflow mixtures and potential end-members projected in the U- and EM-spaces respectively (a). The U-space is defined by the two principal components U1 and U2. The final mixing model is defined with pink lines. Factor loadings for the selected variables (b). Contribution proportions of end-members in the M1_outflow mixture during sampling campaigns S3-S6 (c).**

5.1.3.7. Conceptual model of the study site

The botanical, hydrogeological and hydrochemical knowledge obtained in the framework of this study was combined in the conceptual hydrogeological model of the Matsi spring fen (FIGURE 28). The conceptual model aims to characterize the hydrogeology of the Matsi spring fen on the example of the M1 fen polygon.

The combined interpretation of hydrodynamics, thermal/isotopic signatures and hydrochemistry allowed to confirm the dependence of the Matsi spring fen on the Middle Devonian groundwater body in the Gauja-Koiva River basin. In the Matsi spring fen study site, this groundwater body included the shallow Q aquifer and the underlying D_2*gj-ar* aquifer. The shallow Q aquifer, providing a cumulative 33% of recharge to the M1 fen polygon on average, was the primary source of macronutrients and pollutants due to intensive agricultural activities on the plateau. The D_2*gj-ar* aquifer, on the other hand, was found to be providing the majority (67±19%) of the recharge to the M1 fen polygon. The D_2*gj-ar* aquifer was also found to be the primary contributor to the M2 and M3 fen polygons.

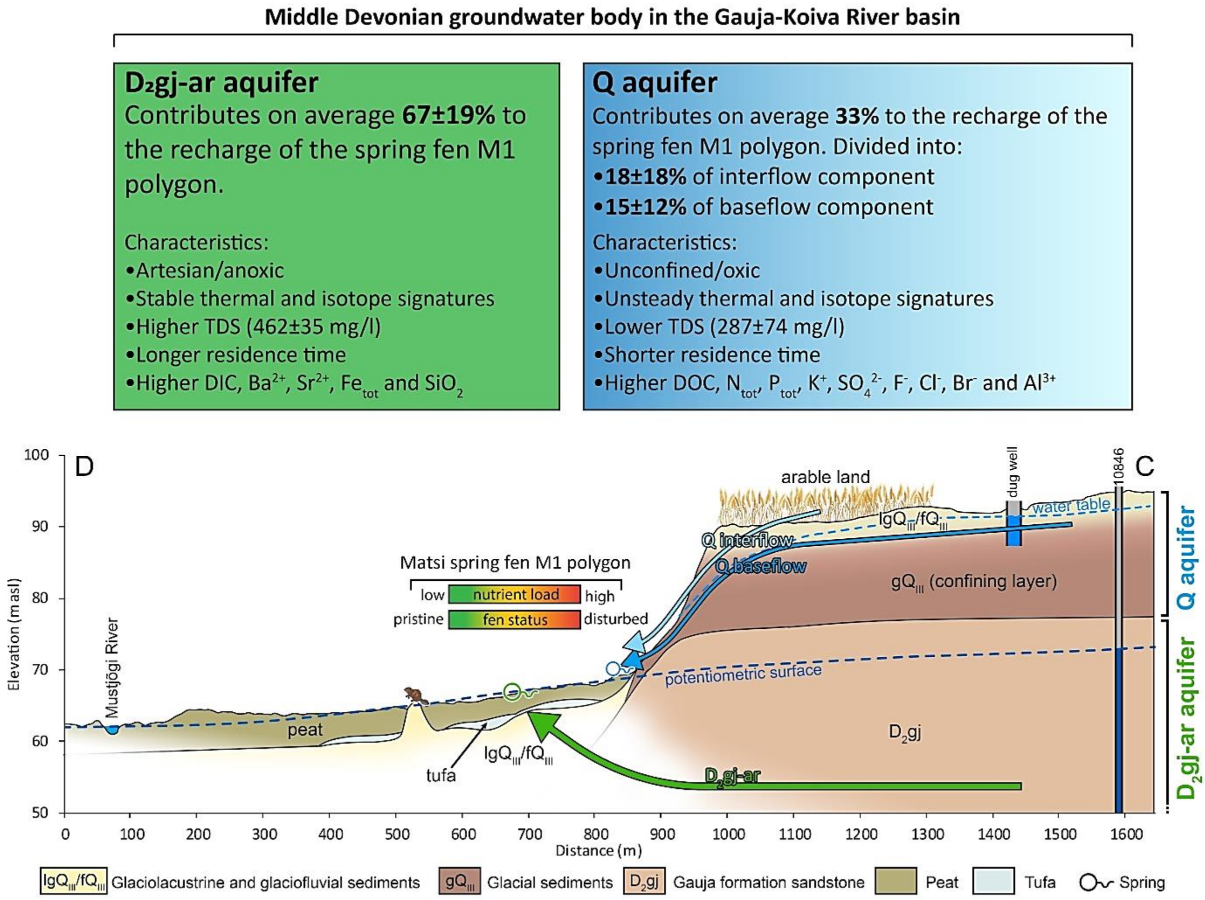


FIGURE 28 **The conceptual hydrogeological model of the Matsi spring fen study site. For cross-section location see FIGURE 17.**

The groundwater of the D_2gj-ar aquifer on average featured higher mineralization when compared to the Q aquifer, likely due to longer residence time but also slightly different mineral composition of the aquifer material. The D_2gj-ar groundwater dilutes the nutrient load originating from the Q aquifer to some extent as it flows downstream in the M1 fen polygon. However, the occasional high mean values of P_{tot} and N:P ratio (FIGURE 24) suggest that the dilution of the upland pollution in the M1 fen polygon was not always sufficient. Consequently, the dominance of D_2gj-ar aquifer discharge in the northern and central part of the M1 polygon has favored the formation of characteristic spring fen vegetation. This part of the fen is valuable in its current state. To the contrast, the nutrient load from the Q aquifer in combination with high water level has altered the plant communities at the upstream part of the M1 fen polygon. The movement of water might have its impact on the vegetation as well.

During the short period of this study, the impact of beaver activity to the M1 fen polygon status was found to be mainly quantitative, and short-term. Due to the dominant artesian discharge of the D_2*gj-ar* aquifer, the quantitative effect of damming was mainly limited to the period of construction and destruction of the dams. Due to the shortness and the relatively low resolution of our time series, the only significant physico-chemical effects rising from the beaver activity were mainly limited to changes in thermal characteristics of the surface water in ponds. In some cases, also minor alterations in local flow directions could be observed. The beaver damming has likely contributed to the distribution of high-water-dwelling plant species in the SE part of the M1 polygon.

5.1.3.8. Assessment of Matsi spring fen according to the schemes of quantitative and qualitative effect

The collected dataset allows to evaluate whether the Middle-Devonian GWB in the Gauja-Koiva river basin has caused significant damage to the Matsi spring fen as a GDTE.

Quantitative effect

Step 1 - *Is there any piece of evidence that water level in the GDTE is considerably lower than it has been previously or is typical to analogous ecosystems, or the conservation status of the GDTE is worse than "good" according to the assessment based on the Habitats Directive?*

There is no evidence that the water level in the Matsi spring fen has dropped or is too low for that type of ecosystem to function properly. Also, the overall conservation status of the fen is good, though the vegetation communities in the marginal areas of the fen indicate some deterioration.

Verdict - the Middle-Devonian GWB in the Gauja-Koiva river basin **has not caused significant quantitative damage** to the Matsi spring fen as a GDTE.

Qualitative effect

Step 1 - *Is there any piece of evidence that the GDTE is in a considerably worse condition than it has been previously or is typical to analogous ecosystems, or the conservation status of the GDTE is worse than "good" according to the assessment based on the Habitats Directive?*

The overall conservation status of the fen is good, and though some of the vegetation communities in the marginal areas of the fen are atypical, it cannot be said that the status of the fen is considerably worse than is typical to spring fens in general. Therefore, in a standard assessment of the GWB status, the verdict would be that the Middle-Devonian GWB in the Gauja-Koiva river basin **has not caused significant qualitative damage** to the Matsi spring fen as a GDTE.

<u>But for the sake of evaluating the assessment scheme, the Matsi spring fen will here be considered as failing to be eliminated in the first step.</u>

Step 2 - *Is there any relevant and possibly polluting human activities in the vicinity of the GDTE?*

Yes, there is a fertilized field a few hundred meters upstream of the fen. Fertilizers applied to fields may enrich shallow aquifers with nutrients.

Step 3 - *Has the deterioration of the conservation status of the GDTE been caused by changes in the water chemistry (N_{tot}, P_{tot}, nitrates, etc.)?*

If yes, then not primarily. The atypical vegetation communities in the marginal areas of the fen have most probably developed because of a high-water level. That supports the domination of *Carex acuta* and prevents the formation of a moss layer typical to spring fens. Therefore, the primary cause for the deterioration of the conservation status of parts of the fen are the beaver dams and not unsuitable water chemistry.

Verdict - the Middle-Devonian GWB in the Gauja-Koiva river basin **has not caused significant qualitative damage** to the Matsi spring fen as a GDTE.

5.2. Kazu leja valley pilot site in Latvia

5.2.1. Introduction

The Latvian pilot area Kazu leja (57°19`58, 25°20`28) is located in Gauja National Park in the surroundings of Cēsis and Priekuļi. The approximate area of the pilot area is 112 ha. Kazu leja is a complex site with plentiful evidence of groundwater discharge. Due to the presence of fens and petrifying springs it appears to be a complex groundwater dependent ecosystem.

Kazu leja is a spectacular subglacial valley, up to 42 m deep. Numerous springs are found on the slopes of Kazu leja, particularly in Lībāni-Jaunzemji side-ravine. During Holocene, extensive freshwater tufa deposits precipitated out of spring waters on the slopes of the Kazu leja (Timšs, 1944). Nowadays the tufa deposits are exhausted except for one block at the mouth of the Lībāni-Jaunzemji ravine. There, water from the main group of springs cascades over up to 6 m high slope of the former tufa mining pit. In warm summer mornings, occasional whitening events (FIGURE 29) in ponds where spring water collects have been observed by local inhabitants, indicating precipitating of carbonates in the water column. Also, along the flow pass of the springs carbonate precipitation can be observed. The water from springs is collected in the River Triečupīte on the valley floor and leaves Kazu leja through a culvert in the railway embankment.

FIGURE 29 **Seven Spring' Brook near Kalna Jaunzemji homestead: strong whitening of the spring water is evident likely due to formation of calcium carbonate crystals in water column (photo: Anda Magone, July 25, 2019)**

The Latvian GroundEco project team (University of Latvia, Nature Conservation Agency) investigated hydrogeological and ecological conditions in Kazu leja in 2018–2019 exploring the interactions between groundwater and terrestrial ecosystems and the applicability of the joint Estonian-Latvian methodology developed by the GroundEco expert team (see Chapter 4.4.). The following questions were set:

1. Which elements (aquifers) of the groundwater system feed the GDTE in Kazu leja?
2. What are the dominant water sources in Kazu leja and what is their proportion?
3. What types of GDTEs are found in Kazu leja? What is their distribution, origin, quality?
4. Are there any anthropogenic pressures to groundwater quantity or quality that might threaten the GDTE quality in Kazu leja?
5. What are the lessons learned from the Kazu leja pilot study concerning the applicability of the joint Estonian-Latvian GDTE methodology?

To answer these questions, the available data were reviewed and extensive field work carried out. Four groundwater and surface water automatic level loggers were installed. Six campaigns of ground- and surface water sampling were undertaken, monitoring major ion, trace element and nutrient concentration and water stable isotope ratios. A field inventory was carried out to identify and map the spring-fed habitats, to describe and classify the vegetation of spring flushes and fens, and to find out the factors affecting the habitat variation and quality.

5.2.1.1. Description of Kazu leja

The area of interest to be examined in Kazu leja pilot site were loosely defined as stretching to the east from Cēsis–Valmiera railway embankment up to the opening of the valley to the River Vaive valley near Priekuļi–Jāņmuiža local road. Kazu leja valley crests mark the N and S boundaries, including the Lībāni-Jaunzemji side ravine where the largest of the springs are located. In addition, potential surface and subsurface catchment area marked by the surrounding river valleys (Gauja, Rauna, Vaive) was identified. Finally, the nearby settlements – Cēsis town and Priekuļi and Jāņmuiža villages, that have centralized municipal water abstraction sites - were included in broader surrounding that should be considered.

5.2.1.2. Protection regimes in Kazu leja

Kazu leja is located in Gauja National Park. Most of the area of pilot site lies in the nature reserve zone of the national park (Law of Gauja National Park). Some marginal areas of the pilot site occur in the landscape protection zone and neutral zone of the national park. The core area of the pilot site largely overlaps with the area of geological and geomorphological nature monument (Cabinet Regulation No.175 (14.04.2001), "Regulation on Protected Geological and Geomorphological Nature Monuments") (FIGURE 30). The restricted and permitted activities in the Gauja National Park and in each of its zones are defined in the Cabinet Regulation No.317 (02.05.2012), "Individual Regulation on Protection and Use of Gauja National Park" and the aforementioned Cabinet Regulation No.175 (14.04.2001).

All types of protection regimes in Kazu leja (FIGURE 30) are of national importance, whereas the entire territory of Gauja National Park is included in the network of protected nature areas of European importance (Natura 2000).

The national regulations that apply to the area of Kazu leja within the borders of the nature reserve and geological and geomorphological monument include a prohibition of mining, installing hydrotechnical constructions (e.g. dams), establishment and new drainage systems, reconstruction and renovation of existing drainage systems (excluding special cases aimed at restoring habitats and species by receiving a permit from the Nature Conservation Agency), building of underground buildings. Extraction of groundwater is permitted only for personal use.

In the landscape protection zone adjacent to the Kazu leja, it is not allowed to establish new mining areas, as it is a valuable landscape defined in the spatial plan of the local municipality. However, the existing mining operations can be extended, though this may require an assessment of environmental impact, as defined by the national regulations concerning environmental impact assessment and assessment of the impact on Natura 2000. In the neutral zone, no particular restrictions of land use are applied, other than those applied anywhere in Latvia, e.g. in agriculture, mining and restrictions defined in the spatial plan of the local municipality.

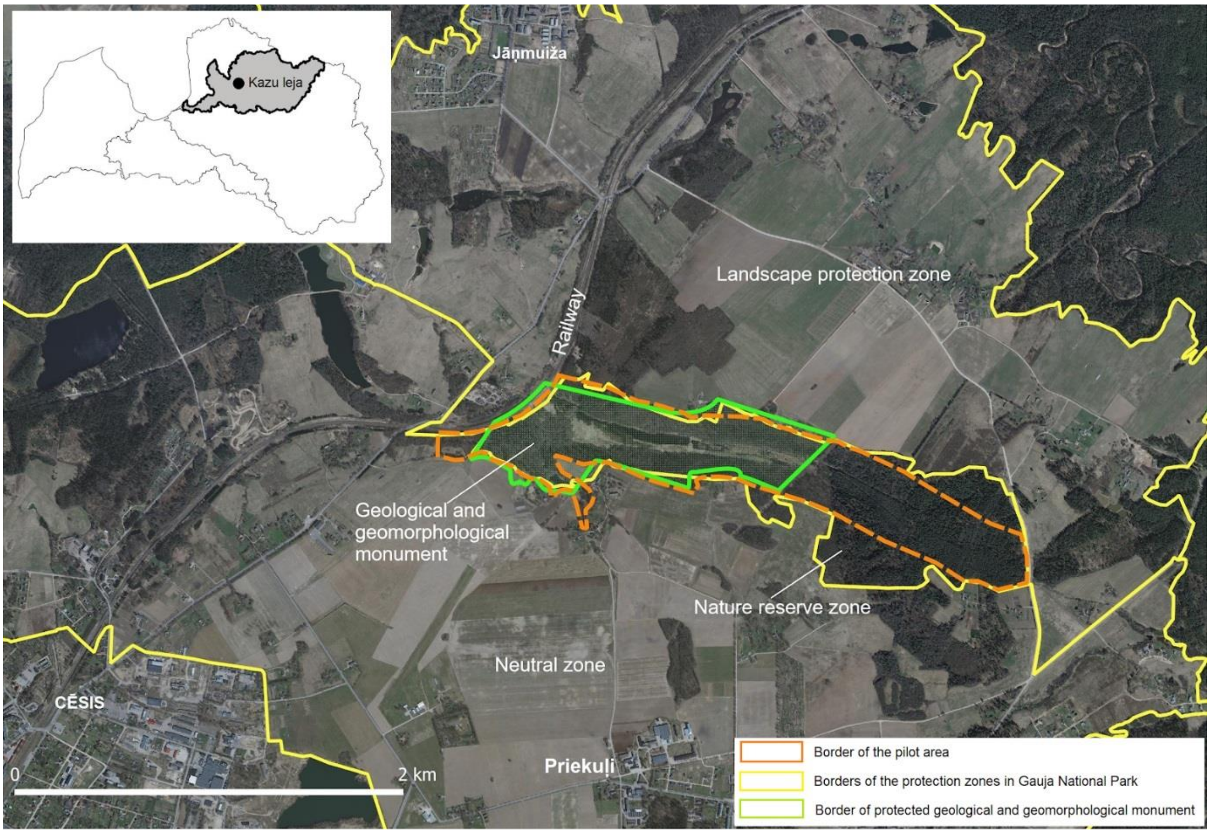


FIGURE 30 **The protection regimes in Kazu leja**
(Orthophoto: © Latvian Geospatial Information Agency, 2016-2018)

5.2.1.3. Geomorphology

Kazu leja is 3.6 km long, 0.3–0.8 km wide and 35–42 m deep valley-like depression oriented in EW direction. It connects the modern valleys of River Vaive and River Gauja (Āboltiņš, 1998). Outside of territory of interest NW from the railway embankment, the valley is less pronounced (Bušuleja valley) (Āboltiņš, 1998) that is eventually merging with the valley of River Gauja. Kazu leja has a distinct U-shaped cross section with steep slopes and flat bottom. The elevation of the surrounding undulating plain is 105-117 m asl, the valley floor is 65-72 m asl (FIGURE 31). The depth of the valley is 40-45 m. The valley topography partly reflects the paleo-incision system incised into Upper and Middle Devonian sediments.

According to Krievāns (2015) and Juškevičs (2000), this paleo-incision system was formed in proglacial conditions of the Fenoscandian Ice sheet deglaciation. The preset topography is a result of paleo-relief transformation. On the slopes of the valley within the area of interest up to three terraces of erosional origin can be traced at about 72, 66 and 62 m asl (Krievāns, 2015). The valley slopes were steep reaching the critical angle of repose in places. The largest artificial modifications in the morphology are associated with excavation of tufa deposits on the slope of the Kazu leja and along Lībānu-Jaunzemju ravine.

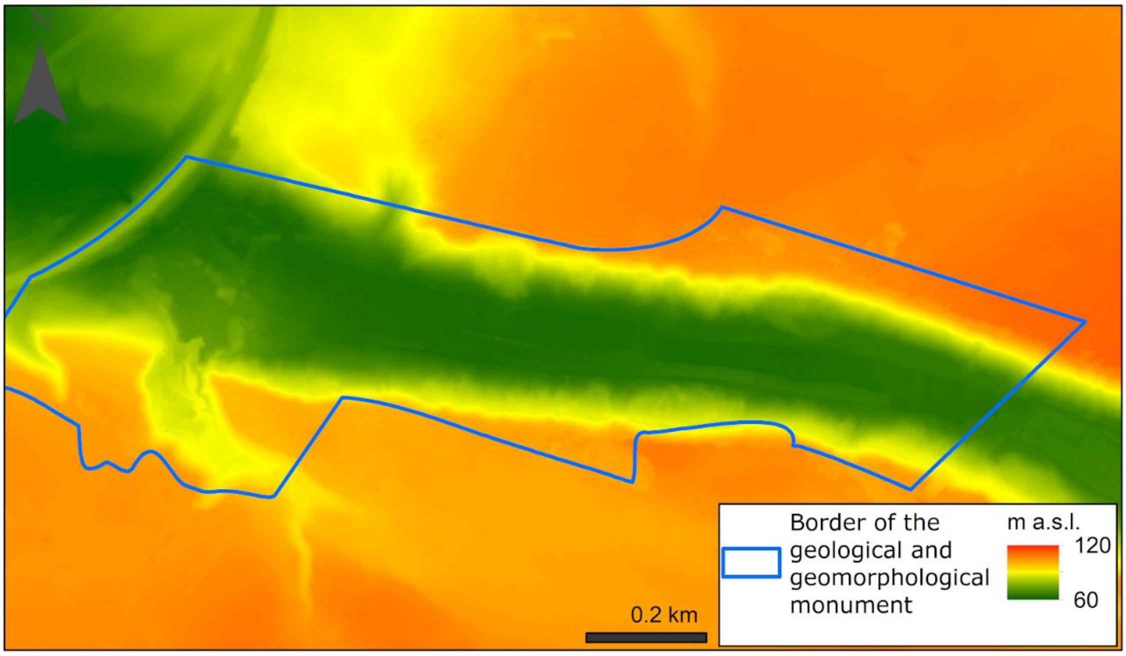


FIGURE 31 **Topography of Kazu leja (data from LĢIA laser altimetry (LIDAR, April 2, 2019))**

5.2.1.4. Hydrology

The small River Triečupīte drains the Kazu leja site. Formally, it can be considered that the culvert under the Cēsis-Valmiera railway line embankment is the confluence of all the surface water draining from the Kazu leja. At the base of the main valley there are several cascading ponds. These ponds appear to be a result of peat extraction in the mid-20th century. The water table and extent of the ponds are largely regulated by beavers.

There is a group of seven springs, here dubbed the main group of springs, discharging in Lībāni-Jaunzemji side-ravine (FIGURE 32). A ditch collecting water from tile drainage of the cultivated fields on the plato adjacent to Kazu leja discharge into the same Lībāni-Jaunzemji side-ravine. However, noticeable water flow in this ditch has been observed only immediately after significant precipitation events. Before entering Kazu leja, the spring water runs through two ponds that are partly regulated by beavers. After that the spring water runs over 6 m high artificial freshwater tufa outcrop forming a cascading waterfall. The waterfall is often called Septiņu avotu ūdenskritums (Seven Springs' Waterfall), and the stream is sometimes referred to as Septiņu avotu strauts (Seven Springs' Brook). During high-water periods when the amount of water is supplemented by snowmelt and excessive rainfall, another temporary waterfall is formed next to Seven Springs' Waterfall. During periods of low water flow the Seven Springs' Brook enters River Triečupīte just before the railway culvert. During periods of high-water part of its flow was diverted to the NE entering the Triečupīte near another small culvert of a dirt track.

Other springs discharge on the slopes of the main valley. There is a relatively large spring in the right (northern) slope of the valley and a zone of many small springs on the left (southern) slope. The water of springs on the slopes of the main valley often infiltrates into the slope scree deposits. It is hypothesized that it re-emerges on the land surface in the valley floor again as discrete springs or diffuse seepage.

There is a derelict tile drainage system at the base of the Kazu leja valley (FIGURE 32).

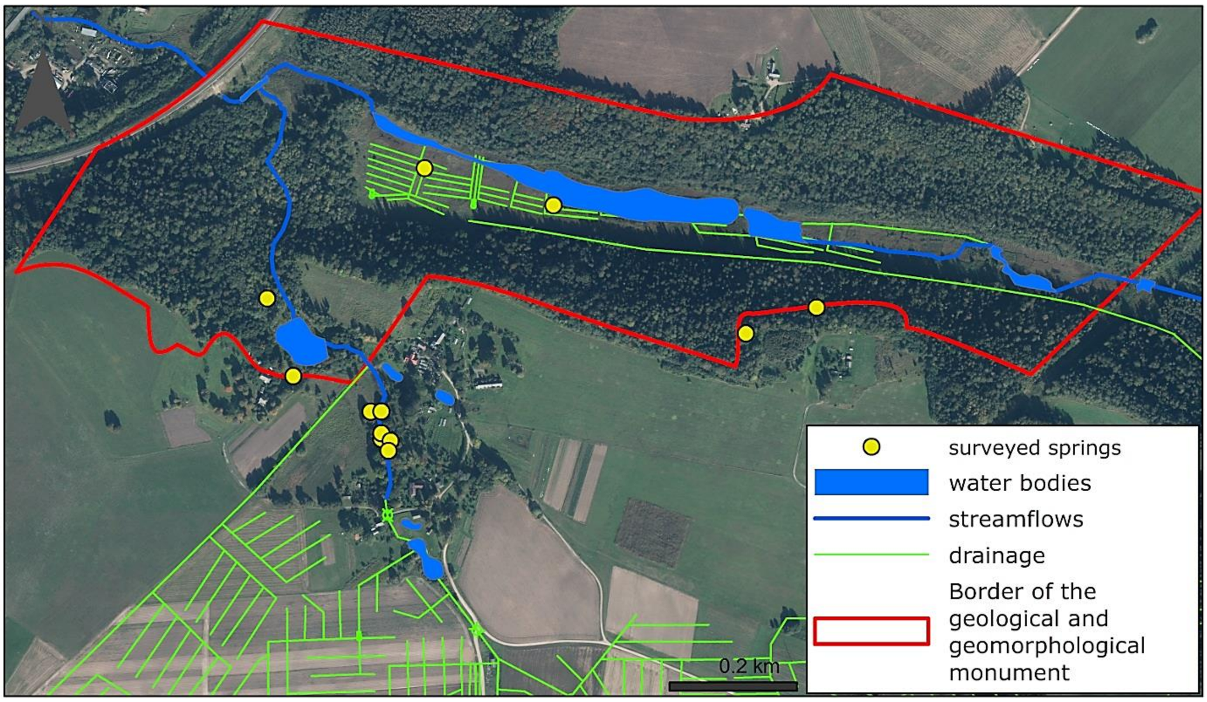


FIGURE 32 **Schematic map of the hydrological network, derelict tile and ditch drainage system in Kazu leja**

5.2.1.5. Geology

The surrounding terrain of Kazu leja site is undulating plain covered with up to 10 m thick till (diamicton, loam) deposits. Alongside the slopes and within the valley peat and alluvial sediments have accumulated (Āboltiņš, 1995). The bedrock surface in the valley vicinity varies about 110-115 m asl and along the course of the paleo-incision that largely coincide with modern terrain reaches down to about -30 m asl (FIGURE 34). It is incised into dolomites of upper Devonian Pļaviņu formation as well as underlying sandstones, siltstones and clays of middle Devonian Amata, Gauja and Burtnieks formations (FIGURE 33). The deepest part of the incised valley is believed to be filled with glacial sediments (Bendrupe, Arharova, 1981). The valley fill is recovered in borehole No.18641 (Well Database maintained by LEGMC, Takčidi, 1999) some 2 km NW from the W-end of the Kazu leja to a depth of 32 m bsl. It is filed by layering of till (diamicton, loam) and sand-gravel deposits.

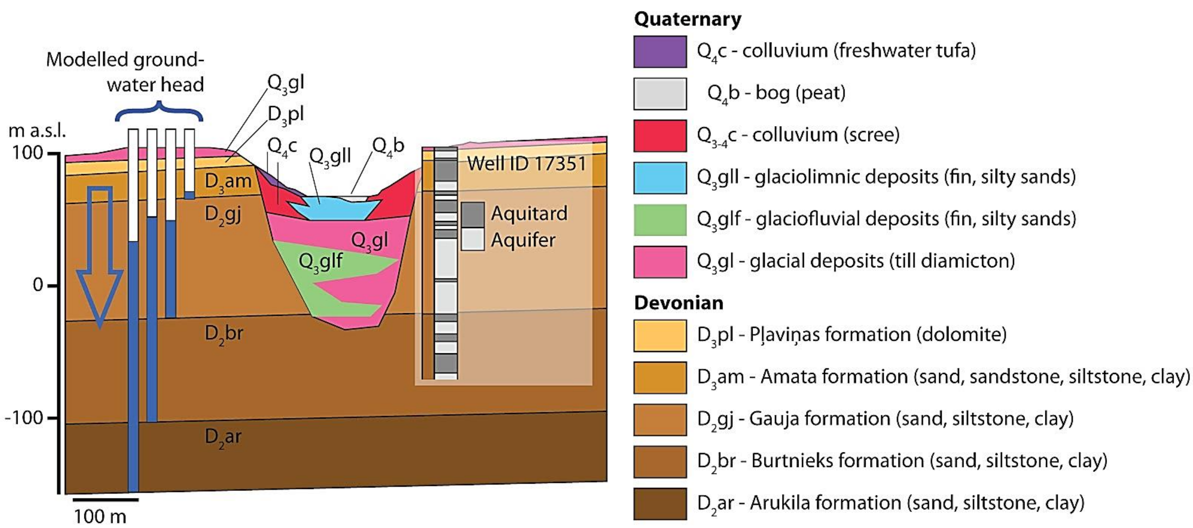


FIGURE 33 **Conceptual geological core-section across Kazu leja, S-N direction, exposing buried valley filled up with quaternary deposits**

The dolomites of Upper Devonian Pļaviņas Fm form an 8 to 13 m thick patch with increased clay content at the bottom few meters. Dolomites overlay up to 20 m thick upper Devonian Amata formation of mostly fine grained (clay, silt) sediments with sandy interbeds. It is underlain by layering of clays and fine-grained sandstone of Middle Devonian Gauja formation. Deeper there are sandstones and clays of Burtnieks formation. The interface between Gauja and Burtnieks formation is at elevation between -16 and -34 m asl.

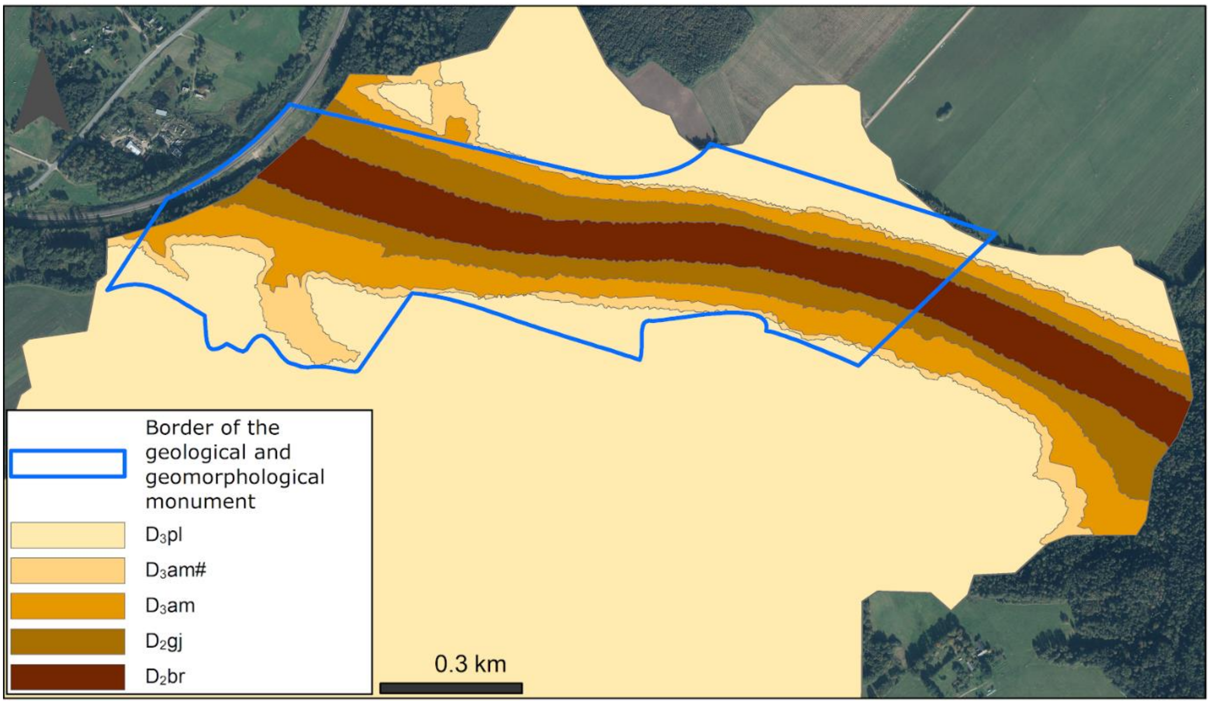


FIGURE 34 **Geological map of the Triečupīte watershed. D_3pl – Upper Devonian Pļaviņas Fm, $D_3am\#$ –- upper impermeable part of Upper Devonian Amata Fm, D_3am – Upper Devonian Amata Fm, D_2gj – middle Devonian Gauja Fm, D_2br – middle Devonian Burtnieki Fm**

During the Holocene extensive freshwater tufa deposits precipitated out of spring water on the Kazu leja slopes (Timšs, 1944). The estimated initial volume of tufa in Lībāni-Jaunzemji deposit was 230,000 m^3 (Rozenšteins, Lancmanis, 1924). Other smaller deposits were found on the slope of the Kazu leja ca. 1 km to the east from Lībāni (Melnalksnis et al., 1955). These all were extracted by the second part of the 20th century, except for one block in the mouth of the Lībāni-Jaunzemji ravine. Besides this, there were historical reports of sinkholes in the plato surrounding Kazu leja. This evidence suggests that the dolomite aquifer feeding the springs must be strongly karstified, forming the source area of precipitated tufa.

5.2.1.6. Hydrogeology

Formally Kazu Leja is located within the margin of the groundwater body D6 of Pļaviņas-Amula Upper Devonian multi aquifer system. At this location the D6 groundwater body is overlaying A8 groundwater body of Arukila-Amata Middle-Upper Devonian multi aquifer system (Borozdins, 2018). More specifically, above the first regional aquitard of Middle Devonian Narva formation three regional aquifers separated with continuous rather impermeable deposits – Middle Devonian Arukila-Burtnieki, Middle Devonian Gauja – Upper Devonian Amata and Upper Devonian Pļaviņas aquifer. Piezometric heads of these aquifers vary between 50 to 70 m asl (FIGURE 33). Considering the depth of the buried valley and its lithological composition, connection of these aquifers is possible, as evidenced by similar levels of piezometric heads, which typically are more distinct (Spalviņš, 2016).

Springs discharging on the slopes of Kazu leja are draining karstified and fissured dolomite aquifer of upper Devonian Pļaviņas formation. It was speculated that groundwater might discharge from sand interlayers in underlying clastic Devonian Amata and Gauja formations as well emerging at the surface on the lower part of the valley slopes or its bottom.

Likely, groundwater sources discharging in the Kazu leja were identified from geological (Popovs et al., 2015) and hydrogeological (Spalviņš, 2016) models.

5.2.1.7. Land cover and short land use history

The largest proportion of Kazu leja (nearly the entire area of slopes, part of the valley floor) is covered by forest, both mixed (linden *Tilia cordata*, elm *Ulmus* spp., maple *Acer platanoides*, spruce *Picea abies*, pine *Pinus sylvestris*, birches *Betula* spp., hazel *Corylus avellana*, etc.) and coniferous (spruce) forest. Most forests have developed naturally and have a relatively high naturalness degree, though a large proportion of them have been selectively used for logging in the past. On the left slope in the western part of the valley, the forest is secondary and has developed in the former tufa mining area – the ground is largely modified by human activity in the past. Tufa was extracted from the end of the 19th century or the very beginning of the 20th century to the 1960s. The mining activity significantly changed the landscape (relief, soils, vegetation cover). Locally, also dolomite was extracted for the needs of the surrounding homesteads.

The valley floor is covered by a spring-fed peatland. In the eastern end of the valley, there has been a transition mire or a raised bog, nowadays completely overgrown with coniferous forest, most probably planted. In the 1930s and 1940s, this area was used for peat cutting. The peat was used as fuel in the nearby agricultural school in Priekuļi and in the nearby Cēsis town. In the rest of the valley, the peatland can be classified as a fen. During the period from the 1930s to 1960s, it was used for peat cutting, though the extent is not known – most probably it affected the entire area of the valley floor. Nowadays, the valley floor is covered by a fen that has recovered after cessation of peat cutting, shallow ditches (excavated during the peat extraction period), and a mesic grassland. The grassland was drained and most probably used for hay making, nowadays it is abandoned, ruralized and gradually overgrows. The valley bottom is drained – most likely, the open ditches were established in the peat cutting time, whereas the tile drainage was established later to use the area for agriculture. Currently, the drainage system is no longer functioning, the wetland is recovering.

The plains surrounding Kazu leja are dominated by cultivated arable land. There are numerous homesteads near the valley, including Inkuļi village (a group of homesteads) at the southern edge of the Kazu leja valley. Few kilometers from Kazu leja, there is the Cēsis town, Priekuļi and Jāņmuiža villages.

5.2.2. Identification of ecosystems and impacts using vegetation as a proxy

5.2.2.1. Methods

The field work included inventory and mapping of terrestrial wetland habitats in the entire area of Kazu leja and classification of plant communities recorded in the spring-fed habitats. In this case, the term "terrestrial" includes spring flushes, fens, banks of small, shallow brooks and small waterfalls formed by the spring brooks; the ponds were excluded. The habitat and vegetation inventories were carried out in summer 2019.

The habitat types were determined according to the vegetation uniformity and abiotic conditions, their relevance to the Latvian national interpretation (Auniņš (ed.), 2013) of the habitats listed in the EU Habitats Directive's Annex I (Annex I habitat types) was assessed. The habitat conservation status was assessed during the field survey by filling in a standard data form (national monitoring methodology). That was done also during the pilot study in Kazu leja (field form filled in for each habitat patch). The data form contains information on the location, structures and functions, impacts, species composition. The wetland habitat types not listed as protected habitats were mapped using descriptive names created for the particular study.

The vegetation was described in 50x50 cm squares, located in transects that were placed in spring-fed habitat patches (flushes, banks of spring brooks, fen, grassland) all across Kazu leja, each spring-fed patch was represented by at least three sample plots (relevés). Vegetation of very small spring flushes and flushes without vegetation (e.g. pebbles without vegetation) was not sampled. In total 125 sample plots were described. In each of them, all vascular plant and bryophyte species were identified, the cover (%) of each species visually estimated. Some other parameters were collected: water pH (measured in the field by manual pH-meter *Hanna HI 938127–HI 98128*. In each sample plot, direct impact of flowing water (yes/no), moisture conditions (dry, mesic, wet), shading (open, partly shaded, shaded), presence of coarse woody debris (yes/no) were recorded and later converted into numerical values. The vascular plant species were identified in the field, while bryophytes were either identified in the field, or collected for identification in the laboratory. The collected samples were identified by bryologists Dr. Ligita Liepiņa and Dr. Līga Strazdiņa (University of Latvia).

The vegetation data were stored in the *Turboveg* database and exported for further analysis by *Detrended Canonical Analysis (DCA)* (*PC-ord 5* program package) using the default settings (downweighing rare species, rescaling axis, rescaling threshold 0.6, number of segments 26); the vegetation data were used as the main matrix, the environmental variables – in the second matrix.

After initial cluster analysis, solely the spring-related relevés were extracted for further analysis, i.e. 99 relevés. The rest of relevés did not contain spring-specific vegetation, the vegetation was composed mainly of mesophytic plant species, therefore they were not acknowledged as part of ecosystems directly affected by groundwater.

To explore the spring ecosystems and understand the species diversity, also a snail inventory was carried out (by Dr. Voldemārs Spuņģis, University of Latvia). For the field inventory, basically the national methodology for snail monitoring in Natura 2000 sites was applied. However, due to site specifics the methodology was slightly modified. Since the areas of spring flushes are small, it was not always possible to establish as 100 m long transects, as required by the methodology; the spring flushes are heterogenous and it is nearly impossible to find larger homogenous patches; and the sampling of upper soil layer (turfs) is rather destructive for the habitat. Therefore, according to the methodological modification, the length of transects in Kazu leja was 10 m, the soil samples were taken into three parallel transects with 50 cm distance from each other. The soil samples were taken in each 1 m distance using biocenometre with size 20x20 cm (except for the brooks – in that case, the sample was taken in the nearest possible location). The samples were dried in the laboratory, all taxa identified.

In total six sampling plots were selected, in each 30 samples taken. The sample plots were selected so that they represent typical spring-fed habitats in Kazu leja: (1) spring flushes with active tufa precipitation on steep forested slope; (2) fens with *Carex rostrata*, *Epilobium palustre*, *Equisetum fluviatile* on the valley floor; (3) spring flushes with tufa formation on gentle slope, abundant in shrubs, herbs and mosses. For each site type, the data forms were filled in (vegetation, management, etc.).

5.2.2.2. Results and discussion

5.2.2.2.1. Habitats

In total five terrestrial wetland habitat types were identified in Kazu leja (FIGURE 35). Only one Annex I habitat type – **Petrifying springs with tufa formation (7220*)** – was found in Kazu leja, in total 2.3 ha. The habitat types occur in several locations in the pilot area, covering patches of different size (from small patches covering few square meters to a relatively large complex with mixed slope forest with spring flushes, nearly 2 ha). On the slope forest with many spring flushes (FIGURE 35), each spring discharge was not distinguished as a separate patch, but merged into a single polygon.

According to the Latvian national interpretation manual for identifying the Annex I habitat types (Auniņš (Ed.) 2013 and later versions published in www.daba.gov.lv), also habitats of secondary origin, i.e. human-created tufa outcrops and some of the former tufa extractions areas may be distinguished as 7220* if the characteristic species are present. In Latvia, the relevance to 7220* is not distinguished by active tufa formation process and direct groundwater impact, but by presence of tufa deposits and presence of characteristic species. It means that part of the former tufa extraction area near Lībāni and Jaunzemji (abandoned in the 1960s), including the Lībāni-Jaunzemji tufa outcrop, may be classified as habitat 7220*. In this case, the dry tufa outcrop, the Seven Brooks’ Waterfall (FIGURE 35), a smaller waterfall downstream and the lower part of the slope in the former mining area with active spring discharges and tufa precipitation were classified as the 7220* habitat, as the vegetation of 7220* habitat has recovered. The spring water is alkaline, pH varies averagely from 7,3 to 8,1.

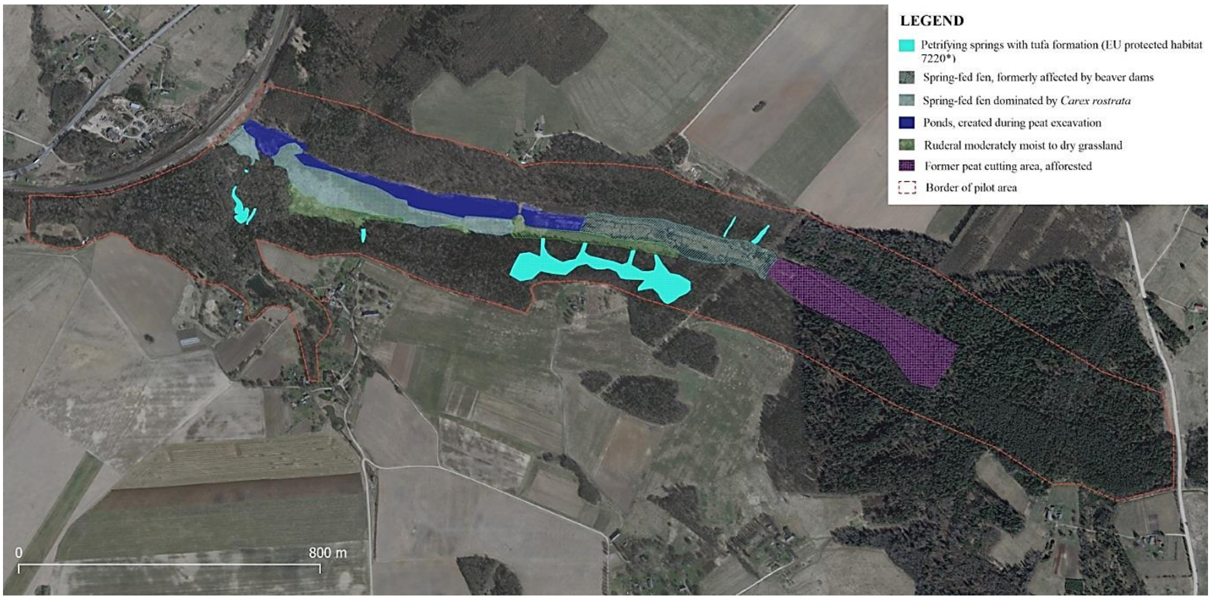


FIGURE 35 **Wetland habitat types in Kazu leja (mapped in 2019)**

On the valley bottom, fen vegetation on peat deposits with a high-water table was found. It did not coincide with any of the Annex I habitat types in the Latvian national interpretation (e.g. 7230, 7220*, 7160), therefore it was mapped as two “new” habitat types: “Spring-fed fen dominated by *Carex rostrata*” and “Spring-fed fen, formerly affected by beaver activity” (FIGURE 35). The difference between both “new” habitat types is defined by differences in vegetation composition, as explained below, most probably caused by beaver activity.

Both fen habitats are of secondary origin – the fen has developed in the former peat cutting area, used in the period from 1930s to 1960s. Part of the fen is transformed into ponds.

In the western part of the valley, its lowest terrestrial part on peat deposits was covered by *Carex rostrata*-dominated vegetation (FIGURE 38). In the central part of the valley, the same type of fen (FIGURE 39) has been inundated by beaver during the last decades (as shown in the earlier orthophoto images) and recently the water level has dropped down, though still fluctuating. The vegetation is polydominant, without dominance of certain species, except for *Phalaris arundinacea*, dominating in part of the area.

Both fens are fed by the spring water flowing down the valley slopes and discharging in the fen. The pH value of water in the fen is averagely 7,5-8,0. At high water level, the valley fens may be partly flooded. Several diffuse spring discharges are found in the fens.

FIGURE 36 **One of the tufa-forming spring flushes on the forested slope of the valley**

FIGURE 37 **The upper part of the Seven Springs' Waterfall, a habitat of secondary origin, the slope of the former tufa extraction area. Nowadays the waterfall is classified as 7220* habitat, as it hosts plant species typical for petrifying springs and has active tufa formation.**

FIGURE 38 **A fen dominated by *Carex rostrata* in the western part of the valley**

FIGURE 39 **A fen, previously flooded by beaver, in the central part of the valley**

In the eastern end of the valley, there is a forested peatland (5,2 ha) that similarly to the rest of the valley floor was used for peat extraction. There are overgrown old peat cutting trenches, some of them overgrown with *Sphagnum* spp. and *Eriophorum vaginatum*, however, most area has overgrown with immature forest, most probably planted in the 1970s. The area is drained by ditches that surround the former mire. The peatland type and its condition before peat cutting is not known, but most probably that had been a small raised bog or transition mire, most probably overgrown with trees, as in old maps it is depicted as forest. Nowadays, fragments of peat-forming vegetation were found only in some peat cutting trenches, therefore the area was considered as highly degraded as peatland without active peat formation, transformed into drained forest.

The rest of valley bottom is covered by mesic, drained grasslands on mineral soils (2,2 ha), most probably cultivated in the soviet time and nowadays dominated by ruderal tall herbs (mainly *Cirsium arvense*) and *Phalaris arundinacea*, and shallow ponds (3,2 ha) that were dug for peat extraction. Small proportion of the area is overgrown with the invasive alien species *Heracleum sosnowskyi* that has also invaded the electric power-line on the eastern part of the valley and spreads uphill on the valley slopes in the forest.

Since the aim of the pilot study was terrestrial ecosystems, the pond vegetation was not studied.

5.2.2.2.2. Compliance of the habitats with GDTEs

According to the GDTE identification methodology (see Chapter 4.4), agreed among the Estonian and Latvian experts within the GroundEco project, all patches of **the habitat Petrifying springs with tufa formation (7220*)** are unambiguously classified as GDTEs. As the spring habitats are fed by the same aquifer, the spring habitat patches may be merged as a single multipart polygon and considered as a single GDTE complex. As only petrifying springs with active tufa formation may be considered GDTEs, the 7220* habitat patch with inactive (dry) Lībāni-Jaunzemji tufa outcrop must be excluded from the GDTE area (FIGURE 40).

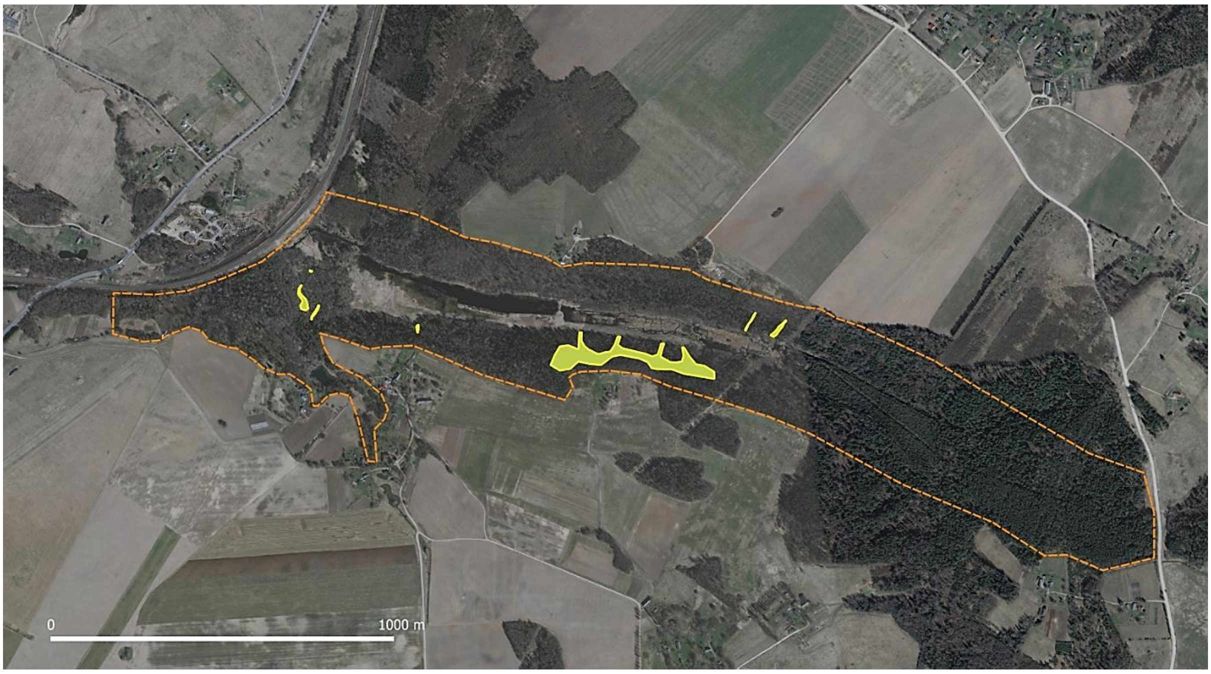


FIGURE 40 **Groundwater dependent ecosystems in Kazu leja, according to the methodology agreed among the GroundEco project experts (only 7220* habitat)**

According to the national Annex I habitat interpretation in Latvia, **the fen on the valley floor** does not comply with any of the Annex I habitat types due to atypical vegetation and lack of characteristic species, therefore is not classified as GDTE, although in fact it should be considered GDTE (predominantly fed by groundwater, peat-forming vegetation, permanently water-logged soil).

Considering the results of the Kazu leja pilot study, it can be concluded that the solutions "not to lose" this type of functioning groundwater dependent ecosystems in the GDTE maps in the future are as follows:

1. The national Annex I habitat interpretation may be revised to include also base-rich fens without typical calciphilous species of *Caricion davallianae*, including fens of secondary origin after peat extraction if there are abiotic conditions typical for rich fens and peat-forming vegetation. The same applies to highly damaged fen habitats (cutaway peatlands) if there is evidence on recovery of peat-forming vegetation. However, this would be a loose interpretation of the EU Annex I habitat manual (European Commission DG Environment, 2013), though deviations for strict habitat interpretation may be found also in the other European countries (e.g. Great Britain, Estonia). Currently the data on fen types other than alkaline fens (7230), their distribution, vegetation, abiotic conditions in Latvia are insufficient.
2. <u>In the case of Kazu leja, both the petrifying spring habitats and fens on the valley bottom are in fact a single GDTE (multipart polygon)</u> (FIGURE 41). However, this would be a deviation from the agreed joint Estonian-Latvian GDTE methodology (see Chapter 4.4.). Moreover, the current habitat data availability in the Gauja basin and in the entire Latvia does not allow identifying the non-calcareous

fens as GDTEs – this is possible only using targeted field surveys, as in the Kazu leja case within GroundEco project.

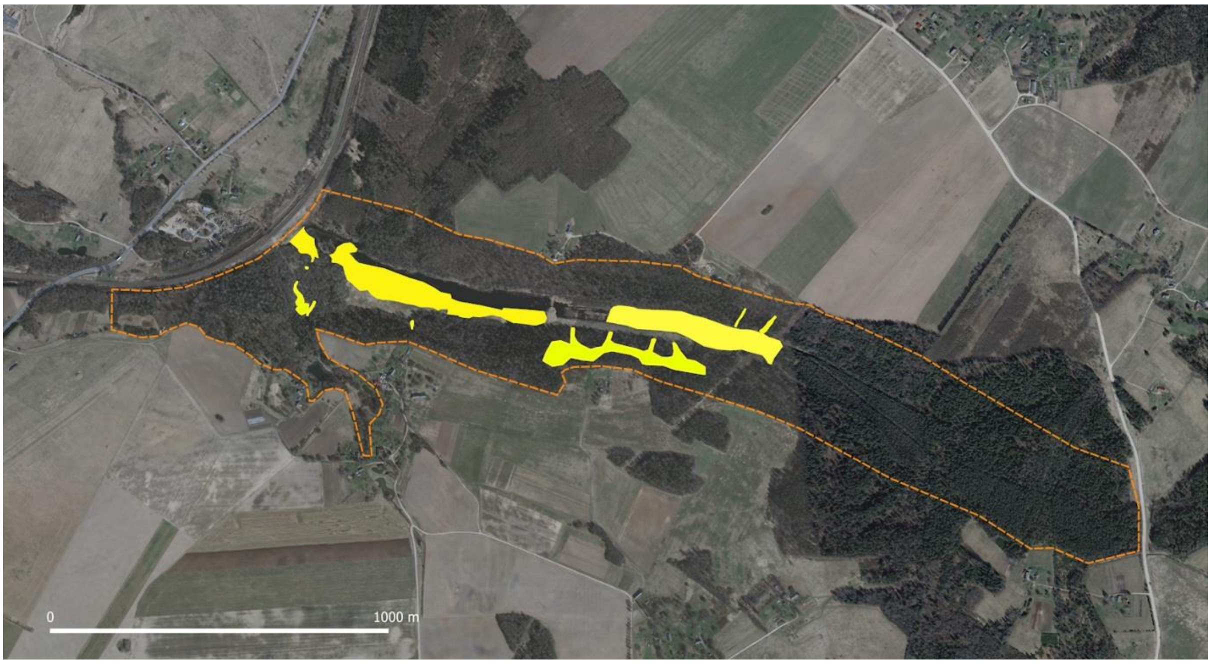


FIGURE 41 **Groundwater-fed habitats in Kazu leja**

The degraded peatland in the eastern end of the valley was not considered a GDTE, because it does not primarily feed from groundwater (as indicated by vegetation) and the peat formation process is disrupted by drainage and peat extraction in the past.

5.2.2.2.3. Vegetation

In the vegetation analysis, 99 relevés were used. In total 128 taxa were identified in the sample plots, 77 of them vascular plant species, 51 moss species. As a result of DCA ordination, two groups of plant communities were distinguished: (1) petrifying springs with tufa formation (spring flushes, banks of spring brooks, spring brook waterfalls) – on the left side in the ordination graph; (2) fens on the valley floor – on the right side of the ordination graph (FIGURE 42). The distinction by location, i.e. local site conditions, is not well pronounced, which suggests that the species composition in the distinct vegetation types is rather homogenous, especially in the fens (right side of the ordination space).

The spring flushes in Kazu leja (2,3 ha) are distinct with moss-dominated vegetation, whereas herbaceous plants may have a minor role in the plant community, especially in shaded sites on the forested slopes (FIGURE 42, FIGURE 37). Phytosociologically the petrifying spring vegetation in Kazu leja belongs to Montio-Cardaminetea class, Montio-Caraminetalia order, Cratoneurion commutati alliance. Affiliation to Cratoneurion commutati community was also identified in an earlier study in Kazu leja by Pakalne and Čakare (2001a, 2001b), the only published paper on spring vegetation in Kazu leja.

In Kazu leja, this type of spring vegetation was found around spring discharges on constantly wet soils with tufa precipitation, usually covering a few square meters, and along spring brooks where the specific vegetation has developed in very narrow strips along the banks of brooks. The plant community is also found in the former tufa mining area, where it does not significantly differ from the natural habitats neither by species composition, nor the vegetation structure. It means that recovery of spring-specific vegetation is possible if appropriate conditions exist, i.e. in this case – continuous groundwater impact, tufa precipitation, stable microclimate. The time period between degradation and recovery in this type of habitat is at least 50–70 years, though it cannot be generalized and may vary from site to site (no other studies on the regeneration potential of petrifying spring vegetation are known).

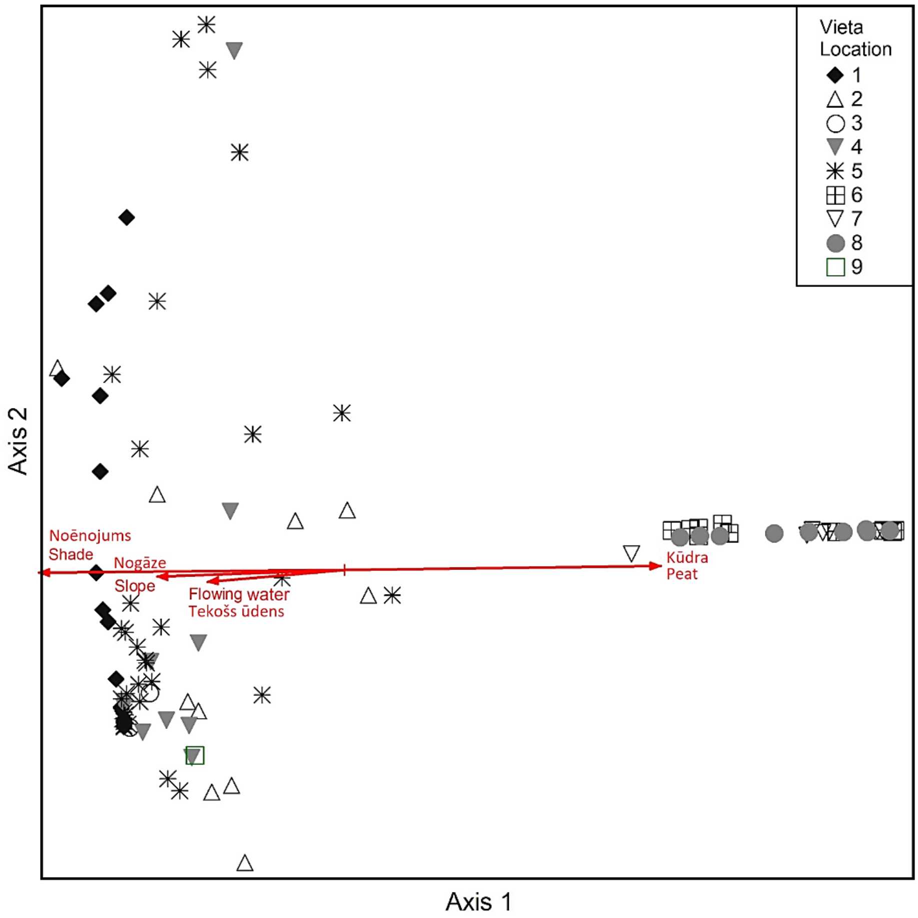


FIGURE 42 **DCA ordination of plant communities in Kazu leja. Locations: 1 – spring flush on the right valley slope (Mazceipi); 2 – the former tufa mining area downstream the Lībāni-Jaunzemji tufa outcrop; 3 – small waterfall downstream the former tufa mining area; 4 – Seven Springs' Waterfall; 5 – spring flushes on the right slope of the valley between Inkuļi and Oļukalni; 6–8 – the fen in the valley bottom.**

In total 95 vascular plant and moss taxa were found in the sample plots, 47 of them vascular plant species, 48 – bryophytes. About half of taxa were frequent in the sample plots, whereas the rest were rare. In shaded locations, the mosses dominate (mainly *Palustriella commutata*, less frequently *Palustriella decipiens* as dominating species). In partly shaded locations in the forest openings, the proportion of herbaceous species was higher (common species are P*oa palustris, Cardamine amara, Geranium robertianum, Mycelis muralis, Brachypodium sylvaticum, Myosotis palustris, Epilobium parviflorum*). Besides *Palustriella* spp., the most frequent moss taxa are *Brachythecium* spp., *Plagiomnium* spp., *Cratoneuron filicinum*, etc. The diagnostic species of the plant community is *Palustriella commutata*.

Locally, the vegetation composition varies in relation with light availability, slope erosion processes, processes in the forest (e.g. tree falling), however, generally the vegetation is relatively uniform in the entire Kazu leja area. The vegetation, especially in the waterfalls, may be frequently damaged by the stream at inundation periods, however, this is a natural process and cannot be considered a damage to the habitat or the vegetation composition, as it recovers relatively quickly.

In the early stage of the pilot study, it was assumed that the vegetation composition in spring flushes may be affected by nitrogen pollution, i.e. as a direct impact of human-caused alteration in the groundwater quality. However, the presence of any nitrophilous species was not recorded. Similarly, it was assumed that the spring vegetation composition may be considerably affected by trampling by visitors in the area around the Seven Springs' Waterfall, however, no significant alteration was found, except for a few individuals of ruderal plant species. Trampling may cause slope erosion and slow down tufa accumulation in the waterfall; however, it cannot be considered a groundwater-related impact.

The fen on the valley floor in Kazu leja (5,2 ha) is of secondary origin, because it has developed in the former peat cutting area (used for these purposes from the 1930s to 1960s). Continuous groundwater supply in most of the former peat cutting area has ensured successful mire ecosystem recovery, though the fen vegetation is still unstable. The area has been drained (both ditches at the foot of the slopes and tile drainage), however, the drainage system no longer functions and does not have a considerable impact on the wetland.

The fen is open, without shrubs and trees, in the previous decades the area has been flooded by beavers. The vegetation is species-poor, dominated by sedges (*Carex rostrata* as the dominant species accompanied by *Equisetum fluviatile*, *Epilobium palustre*, *Naumburgia thyrsiflora*, *Lycopus europaeus*); the moss layer is poorly pronounced, with low number of species (e.g. *Calliergonella cuspidata*, *Calliergon cordifolium*, *Hygroamblystegium riparium*). In total 46 taxa were recorded in the sample plots, 39 of them vascular plant species, 7 – moss species. Most probably, due to its secondary origin (peat cutting in the mid-20th century) the phytosociological affiliation of this community is ambiguous; it is most similar to tall sedge Potentillo-Caricetum rostratae community (Phragmito-Magnocaricetea class, Magnocaricion alliance), occurring in base-rich sites. Although the soil water is alkaline, no specialist species of alkaline fens were found. The dominant species *Carex rostrata* has a wide ecological range (grows both in poor, acidic fens and base-rich alkaline fens). However, in this case *Carex rostrata* was considered the diagnostic species of the plant community, the constant species – *Epilobium palustre*, *Naumburgia thyrsiflora* and *Equisetum fluviatile*.

Due to beaver impact, the fen in the central part of the valley is characteristic with unstable vegetation, it is problematic to classify it using the phytosociological hierarchical system. In the end of the 1990s, it may have been an alkaline fen (secondary, developed after peat cutting?) before the beaver-caused inundation (as suggested, though not clearly described by Pakalne and Čakare (2001a, 2001b).

5.2.2.2.4. Snail fauna in the spring-fed habitats in Kazu leja

During the inventory in 2019, in total 27 snail taxa were found (TABLE 11). The three selected spring-fed habitat types (spring flushes with active tufa precipitation on steep forested slope near Oļukalni; fens with *Carex rostrata* on the valley floor; spring flushes with tufa formation on gentle slope, abundant in shrubs, herbs and mosses at the foot of the Seven Springs' Waterfall (developed after tufa mining)) are represent distinct conditions for snails, therefore only few species were found in all sampling sites. One of them, *Euconulus fulvus*, was found in large numbers. *Euconulus fulvus* may be potentially used as an indicator of this type of fen habitats.

The spring flushes are small and are surrounded by forest vegetation. Therefore, rather many typical forest species are present, e.g. *Vertigo ronnabiensis*.

In terms of snail fauna, the fen with *Carex rostrata* on the valley floor is the most distinct among the sampling sites, inhabited by *Gyraulus rossmaessleri* that indicates constant presence of surface water. Also, *Pisidium* spp., commonly found in freshwater habitats, were found in the *Carex rostrata*-dominated fen and in the shrubby spring flush on the gentle slope downstream the Seven Springs' Waterfall – the least is typical with small spring streams under the dense vegetation cover. Presence of *Pisidium* spp. indicates open water conditions – i.e. the site is characteristic with many small streams.

The number of species and density of individuals varied among the sampling sites. The lowest number of species and density of individuals were in the *Carex rostrata*-dominated fen – it may be explained by lack of suitable feed, as the habitat is abundant in dead litter of sedges (*Carex* spp.), horse-tails (*Equisetum* spp.), willowherbs (*Epilobium* spp.). Slightly higher density of snail individuals was found in the spring flush on the steep slope, dominated by mosses. The snails do not feed on the dominant moss, *Palustriella commutata*, most probably they feed on the dead fallen leaves. In the shrubby spring flush on gentle slope, the population density was high that may be explained by heterogeneous herbaceous vegetation and presence of shrubs and trees. The diversity of feed is essential for diversity of snail species. The number of snail species was similar in the spring flushes on the steep slope and on the gentle slope, as both locations occur under the tree canopies.

TABLE 11

Snail fauna in the sampling sites in Kazu leja

No.	Taxa	Number of individuals			Typical habitats
		K1	K2	K3	
1	*Arianta arbustorum*	3		4	eurybiont
2	*Carychium minimum*	66	2	821	wetland species
3	*Carychium tridentatum*	1		4	forest species
4	*Cepea hortensis*	4		1	mesophilous forest species
5	*Clausiliidae juv.*	42		5	forest species
6	*Cochlicopa lubrica*	5		66	eurybiont
7	*Columella aspera*	4			forest species
8	*Columella edentula*	5		46	wetland species
9	*Euconulus fulvus*	129	39	102	eurybiont
10	*Galba truncatula*			10	standing freshwater species
11	*Gyraulus rossmaessleri*		104		freshwater species
12	*Helicidae juv.*	9		43	forest species
13	*Macrogastra ventricosa*	1			forest species
14	*Nesovitrea hammonis*	1		1	mesophilous forest species
15	*Nesovitrea petronella*	15		10	hygrophilous species
16	*Perforatella bidentata*			1	hygrophilous species
17	*Pisidium sp.*		25	48	freshwater species
18	*Punctum pygmaeum*	109	1	144	mesophilous forest species
19	*Succinea sp.*		19	35	wetland species
20	*Vallonia costata*	1		43	species of open habitats and forests
21	*Vallonia pulchella*			1	hygrophilous species of open habitats
22	*Vertigo antivertigo*	7	144		wetland species
23	*Vertigo pusilla*	1		1	forest species
24	*Vertigo ronnebiensis*	8			forest species
25	*Vertigo substriata*	28		28	forest species
26	*Vitrea crystallina*	40		513	forest species
27	*Zonitoides nitidulus*	19	8		wetland species

K1 – spring flushes with active tufa precipitation on steep forested slope; K2 – fens with Carex rostrata on the valley bottom; K3 – spring flushes with tufa formation on gentle slope, abundant in shrubs, herbs and mosses

Before the field inventory, it was assumed that there may be snail species that are specific for petrifying springs with tufa formation and thus they may be used as indicators of the Annex I habitat 7220* (i.e. GDTE quality). Nevertheless, no snail species, exclusively related to petrifying spring habitats, were recorded in Kazu leja sampling sites. The species composition in Kazu leja wetland habitats represent a variety of wetland and forest habitats, with few species of freshwaters and some eurybiont species with broad tolerance to various environmental conditions.

5.2.2.3. Naturalness and recovery potential of spring and fen habitats and its role in GDTE identification

Naturalness of spring and fen habitats and their recovery potential is an essential question in terms of GDTE identification. The joint Estonian-Latvian methodology (see Chapter 4.4.) states that heavily degraded ecosystems shall not be considered GDTEs (e.g. cutover peatlands, heavily drained areas). However, the Kazu leja pilot study shows that GDTEs may be heavily deteriorated, but in certain conditions capable of regenerating. Thus, we suggest that <u>the ecosystem regeneration capability, either self-regeneration or</u>

targeted restoration, is the key indication in considering whether a damaged wetland or spring ecosystem may be considered GDTE. The regeneration success cannot be estimated shortly after the deteriorating impact, e.g. mining, drainage, as the process may take several decades.

In Kazu leja, the time period since cessation of tufa and peat extraction covers approximately 70 years. In the largest tufa extraction plot near Lībāni and Jaunzemji, the mining impact is still visible, though the vegetation composition and structure do not significantly differ from natural and slightly disturbed petrifying spring habitats.

In the other location ca. 1 km to the east from Lībāni (near Oļukalni), the vegetation composition and structure of the petrifying spring habitats suggest that these are natural habitats. However, there is a documented tufa extraction also in this location (Rozenšteins, Lancmanis, 1928; Melnalksnis et al., 1955), which means that the current tufa deposits and the characteristic vegetation may have recovered after mining. Nevertheless, the mining technique in this location is not known – perhaps a low intensity manual extraction without disrupting the spring discharge is the reason for successful post-mining regeneration.

In both locations, the current condition of the spring habitats allows classifying them as the Annex I habitat 7220* and GDTE.

In the early stage of the pilot study, the fen on the valley floor raised doubt on its relevance to Annex I habitats and GDTE, unless the site history was explored. In this pilot study, the land use history helped to understand the current vegetation composition which did not seem typical for the particular conditions. The atypical species composition for a base-rich spring-fed fen most probably is due to the relative instability after heavy disturbance (peat extraction). After exploring the site, we consider that the valley fen is in fact a GDTE, though there is lack of the habitat-characteristic species and it has been severely damaged. Similarly, to the petrifying springs in Kazu leja, the fen ecosystem is successfully recovering, though not yet fully stabilized.

5.2.3. Geology and hydrogeology

5.2.3.1. Methods

The site investigation was generally organized by following recommendations for the use of conceptual models for groundwater assessment (European Communities, Guidance Document No.26, Guidance on risk assessment and the use of conceptual models for groundwater). In short, the guidelines recommend a structured interactive three-step approach gradually refining understanding of the site in question. Three main components are Step-1: conceptual model; Step-2: risk assessment and Step-3: risk management. It is suggested that each step involves quantitative and qualitative assessment. In the guidelines it is stressed that qualitative understanding (conceptual model) is backed up by quantitative indicators that in turn form the basis for defining qualitative (conceptual) understanding of the GDTEs site in question. It is stipulated that any investigation and management measures are backed up by conceptual model and justified by quantitative measurements.

Based on the initial conceptual model (see chapter 5.2.1.1. Description of Kazu leja) of the pilot site a field campaign was designed that included quantity (level and discharge) and quality (chemical composition) monitoring of surface and groundwater. Six water sampling campaigns were organized in coordination with Estonian partners: April 23, May 30, July 17, September 23 and November 27 in 2019 and March 3 in 2020. In total 57 water samples were collected and analyzed for basic chemical composition as well as biogenic and micro elements and water stable isotope ratios. In addition, water temperature was monitored in 2 springs and 2 surface water objects as well as soil temperature distribution was monitored using a vertical multilevel temperature probe. In addition, infrared remote sensing approach using UVA (unmanned aerial vehicle) was tested.

5.2.3.1.1. Observation points

Ground, spring and surface water quantity and quality were monitored during the project (FIGURE 44; TABLE 12, TABLE 13). Springs as groundwater discharge sites: initially one spring from the main group of the spring (Spring-1) and a small spring at the S slope of the Kazu Leja (Spring-2) were selected. Starting from May 2019 (second sampling stage) a spring at the base of Kazu Leja (Spring-3) and during the last sampling stage, February 2020, large spring on the N slope of the Kazu Leja were included in the programme as well. Surface water: initial sampling programme included River Triečupīte at the railway culvert (Q1_railway) and confluence stream of water from the main group of springs - stream Seven Springs' Brook (Q2_pond), in May 2019 samples were collected from River Triečupīte upstream the confluence with Seven Springs' Brook (River_1), during subsequent sampling campaigns River Triečupīte were sample near Well_2. A ditch collecting water from the tile drainage system was sampled as well on occasions when there was flowing water (Drainage_outlet). Groundwater was sampled in three shallow monitoring wells (augers): Well_1, Well_2 and Well_3. Water samples for stable isotope analysis were collected at a precipitation trap and soil water sampler (Soil_well_3). In addition, plant samples were collected for water stable isotope analysis, but no meaningful results were obtained.

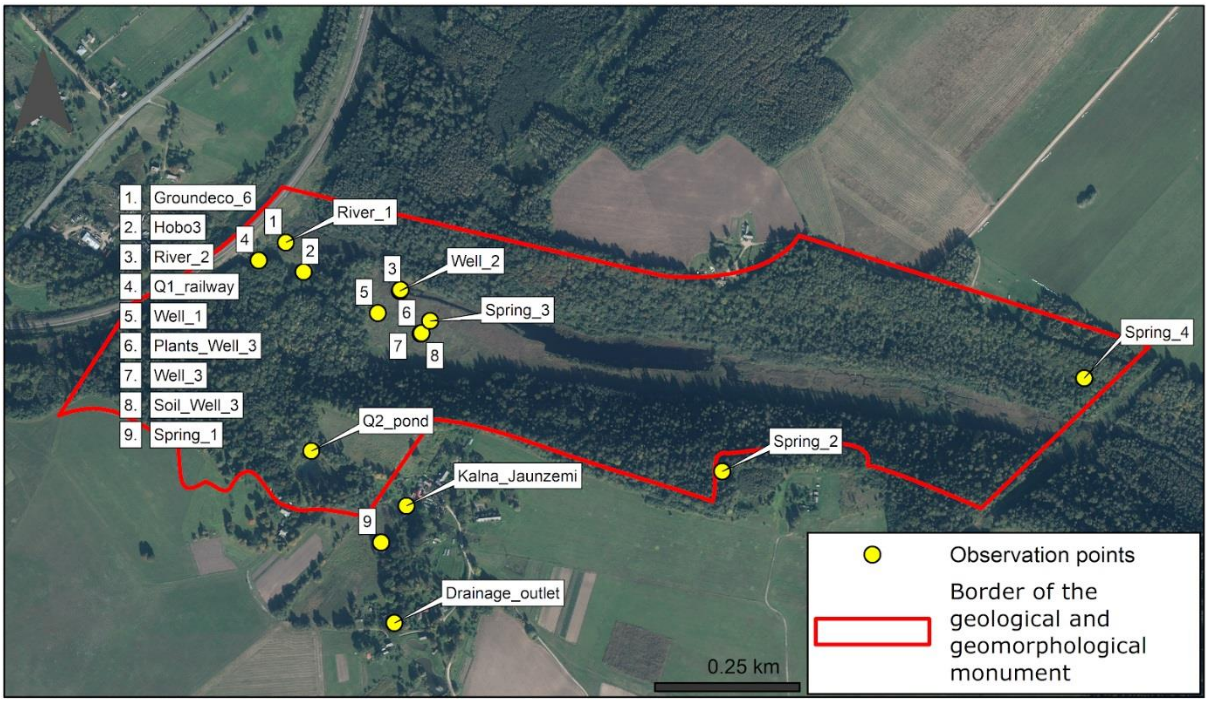


FIGURE 43 **Location of observation points**

Coordinates of the observation points are summarized in TABLE 12, site description is given in TABLE 13 and measured parameters at each location are given in TABLE 14. Geological description of monitoring wells is in TABLE 15.

TABLE 12

Coordinates of the observation points and coordinate reference systems (BalticTM, EPSG:25884; WGS84; EPSG:4326)

Site ID	X, Baltic TM	Y, Baltic TM	Z, Baltic TM	Lat, WGS84	Lon, WGS84
Well_1	580469.992	6355316.09	67.29	25.337	57.334
Well_2	580507.507	6355354.03	65.78	25.337	57.334
Well_3	580544.317	6355281.16	66.61	25.338	57.333
Soil_Well_3	580544.715	6355280.36	66.61	25.338	57.333
Precipitation_Well_3	580545.244	6355281.48	66.61	25.338	57.333
Spring_1	580475.5	6354920.3	100.09	25.337	57.33
Drainage_outlet	580498.501	6354781.19	107.65	25.337	57.329
Spring_2	581063.051	6355043.05	88.19	25.346	57.331
Q2_pond	580355.4	6355078.2	91.78	25.335	57.332
Q1_railway	580264.6	6355406.8	65.61	25.333	57.334
River_2	580508.9	6355356.2	65.84	25.337	57.334
River_1	580311.496	6355437.75	65.84	25.334	57.335
Spring_3	580559.977	6355302.08	66.21	25.338	57.334
Spring_4	581687.1	6355203.6	94.39	25.357	57.332
Kalna_Jaunzemi	580519.43	6354983.3	97.69	25.337	57.331
Hobo3	580341.9	6355386.7	66.88	25.335	57.334
groundeco_6	580311.5	6355437.75	65.83	25.334	57.335

TABLE 13

Description of observation points

Site ID	Water type	Site description
Well_1	Groundwater	Shallow monitoring well, filter from 3.1 to 4.1 m below ground surface
Well_2	Groundwater	Shallow monitoring well 1.0 to 2.0 m below ground surface
Well_3	Groundwater	Shallow monitoring well 1.3 to 2.3 m below ground surface
Precipitation_Well_3	Precipitation	Precipitation trap near Well_3
Spring_1	Spring	Spring
Drainage_outlet	Surface water	Drainage ditch
Spring_2	Spring	Spring
Q2_pond	Surface water	Spring culvert
Q1_railway	Surface water	Railway culvert
River_2	Surface water	River within Kazu Leja, near well 2
River_1	Surface water	River within Kazu Leja just before confluence with Seven-sprig-stream, groundeco-6 Diver for temperature mixing model
Spring_3	Spring	Spring
Spring_4	Spring	spring
Kalna_Jaunzemi	Spring	Dug well ~2m deep, water table above ground surface, and outflowing as a small spring
Hobo3	Surface water	Stream from main group of springs
Groundeco_6	Surface water	River before entrance of the stream from main group of springs, near railroad, temperature measurement for mixing estimation

TABLE 14

Summary of field and laboratory measurements

Site ID	Level	EC	T	Q	Quality
Well_1	yes	yes	yes		yes
Well_2	yes	yes	yes		yes
Well_3	yes	yes	yes		yes
Precipitation_Well_3				yes	
Spring_1					yes
Drainage_outlet				yes	yes
Spring_2			yes		yes
Q2_pond				yes	yes
Q1_railway				yes	yes
River_2 *	yes	yes	yes		yes
River_1*			yes		yes
Spring_3*			yes		yes
Spring_4*				yes	yes
Kalna_Jaunzemi*					yes
Hobo3*			yes		
Groundeco_6*			yes		

Level – automatic level logger in the field
EC –automatic specific electrical conductivity logger in the field
T – automatic temperature logger in the field
Q – manual stream discharge or precipitation measurement in the field
Quality – samples collected for laboratory measurement
* – limited number of samples or short measurement period

TABLE 15

Description of monitoring wells

Site ID	Filter depth, below soil surface (m)	Geology, depth, below soil surface
Well_1	3.1 to 4.1	0.00–0.15 m – soil 0.15–0.32 m – tufa, white, loose 0.32–4.1 m – sands, white fine or mixed (from 1.5 m) with silt, thin clay lamina at 0.5 m, a piece of dolomite at 2.5 m depth 3.0 m - groundwater table
Well_2	1.0 to 2.0	0.00–0.20 m – sludge with plant remnants 0.20–0.30 m – tufa, white, loose 0.30–1.10 m – peat, well to medium decomposed 1.10–1.20 m – tufa 1.20–2.1 m – sands, fine flowing with clay and silt
Well_3	1.3 to 2.3	0.00–0.15 m – soil 0.15–0.30 m – sands 0.30–1.10 m – mixture of tufa and sands, light colored 1.10–2.0 m – fine sands with fine material (silt, clay) admixture, 2.0–2.4 m – sand with clay and dispersed organic matter, dispersed peat and peat 2.4–2.7 m – mixed grained, flowing sands 0.43 m - groundwater table

5.2.3.1.2. Surface watershed (catchment)

The territory was selected considering the local geological structure and land surface elevation (FIGURE 44). The valley of River Gauja forms a natural boundary to the N and W from Kazu leja site. Deep incised valleys of River Rauna entering River Gauja and its tributary River Vaive forms a natural boundary to the E of the Kazu leja site. To the south of the Kazu leja site is undulating plain and, further, the slope of Vidzeme Upland. There is a choke-point on the plato some 10 km south from Kazu leja formed by incised valleys of River Vaive and River Rakšupe. This was selected as the southern extent of the wider examination area.

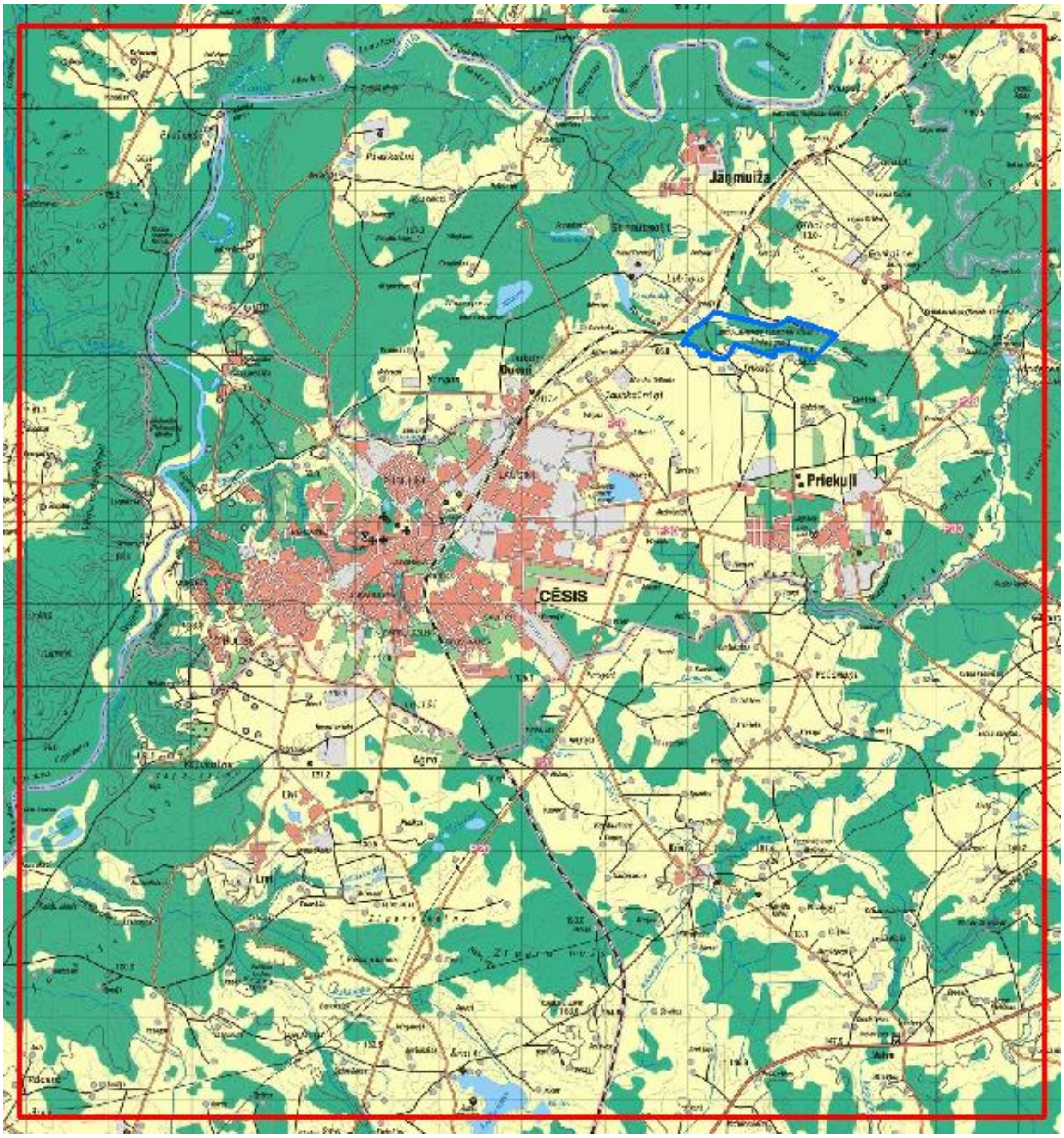


FIGURE 44 **Extent of the initial conceptual model of Kazu leja GDTE site (min X: 571900, min Y: 6345800, max X: 584300, max Y: 6359000 in Baltic TM coordinate system)**

Digital elevation model is necessary to delineate catchment areas. Digital elevation model (FIGURE 45) was compiled from three data sources:

1. digitized topographic map (scale 1:10 000, (TOPO 10K PSRS). Data covers eastern side of model territory; resolution 2 m.
2. Lidar data provided by LĢIA covers western side of the model territory, resolution 1 m.
3. DEM from Latvian Geospatial Information Agency (LGIA), resolution 20 m (LĢIA, 2018).

LĢIA DEM and EU-DEM (Copernicus Land Monitoring Service, 2014) was compared to a digitized topographic map. LĢIA DEM showed better results with mean difference of -2,35 m and standard deviation of 3,44 m. EU-DEM has lower accuracy with mean difference of -5,01 m and standard deviation of 7,29 m. The largest errors in EU-DEM were found in forested territories.

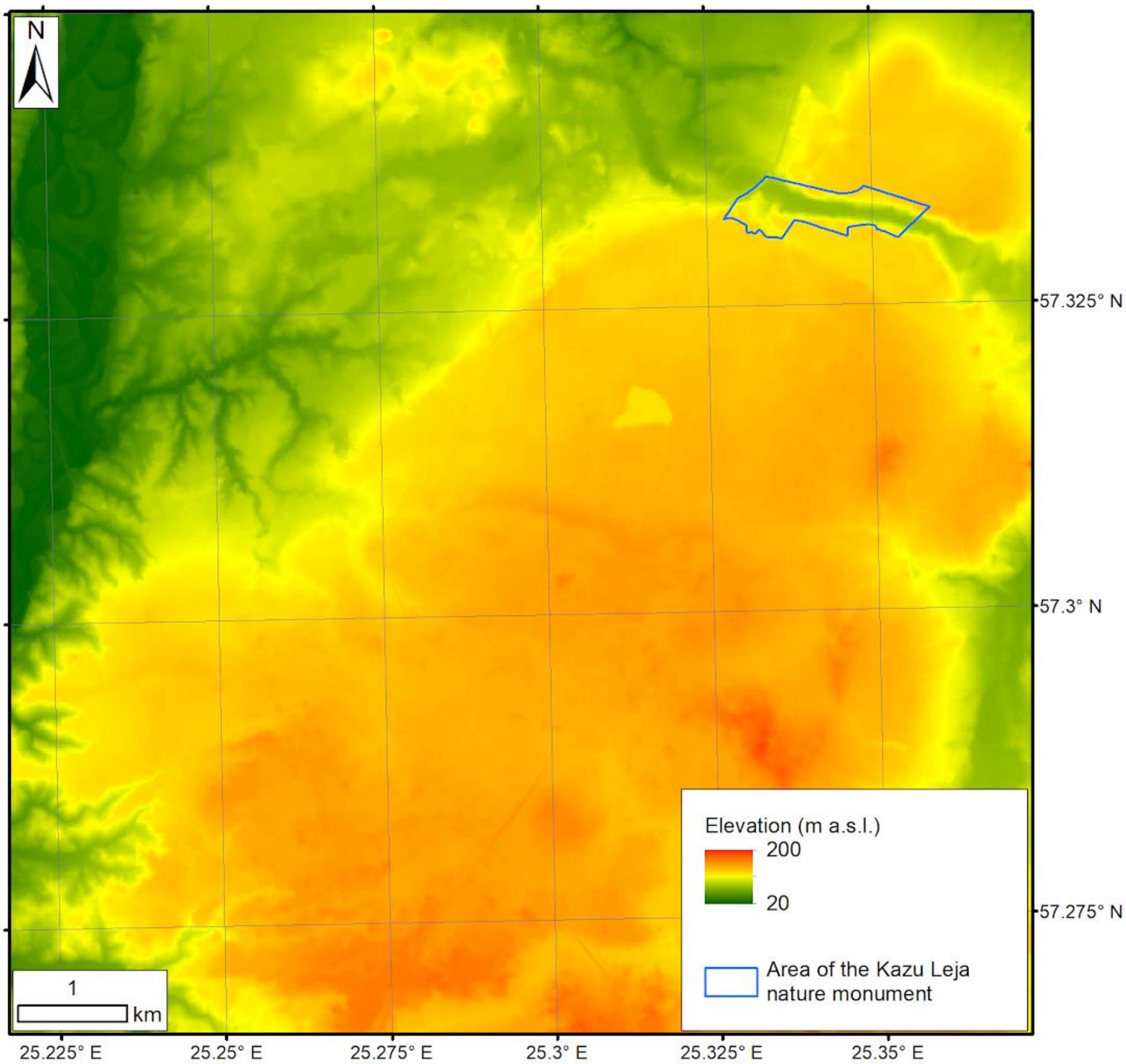


FIGURE 45 **Digital elevation model (DEM) of Kazu leja GDTE site, actual area of Kazu leja is marked with blue line**

To delineate surface watershed, firstly, unwanted depressions were filled in the DEM by GRASS GIS tool r.fill.dir. Then, all watershed basins were delineated by r.watershed tools. Parameter "Minimum size of exterior watershed basin" = 5000. Output: Watershed basins, drainage directions and stream segments. Next – the specific watershed that affects Kazu leja territory was calculated by r.water.outlet tool. Input = drainage directions from previous step and coordinates of the point of interest (lowermost point in the GDTE area) X: 580323, Y: 6355450 (580323, 355450 in LKS92). The result is watershed based on topography where surface runoff would be concentrated in Kazu leja. The final version of delineated watershed was smoothened by applying GRASS GIS tool v.generalize and QGIS tool Simplify. The area of the resulting watershed is 4,73 km^2.

5.2.3.1.3. Subsurface catchment

The subsurface catchment of the springs was estimated using a conceptual approach. Given the sub-horizontal nature of the bedrock geology, it was assumed that the groundwater would flow in the D3pl aquifer to the nearest location where the aquitard at its base was outcropping at the land or rather sub-quaternary surface. The location of hypothetical groundwater water divide can be estimated by constructing the Voronoi diagram. The Voronoi diagram is a partition of the plane into regions closest to each of a given set of points. In our case the points are nod-points of the modeled distribution boundary of the aquitard.

Distribution of impermeable layer (aquitard) at the base of the Pļaviņas aquifer (fractured and certified dolomites) were modeled according to the methodology by Popovs et al. (2015). Linear interpolation was used to obtain intersection of land surface (DEM) and surface of aquitard. Voronoi polygons attributed to the nod-points within surface catchment of the Kazu leja was attributed as contributing their associated subsurface runoff to the springs of Kazu leja.

5.2.3.1.4. Groundwater table in D3Pl dolomite aquifer

Groundwater table in D3Pl dolomite aquifer was not measured directly, instead it was estimated from the water table fluctuations in flooded Lauciņi dolomite quarry near Kazu leja. It is reasonable to assume that the water table in the quarry matches the groundwater table in the aquifer. The water table was estimated by deciphering water line position from aerial photography at 10 locations on the coastline and reading respective elevation on the digital elevation model constructed from LIDAR data.

5.2.3.1.5. Groundwater and surface water level monitoring

Surface and groundwater level monitoring was done by Diver®CDT groundwater level loggers (submersible pressure probes) that include a sensor for electrical conductivity. Measured pressure of the above water column was compensated for fluctuations of atmospheric pressure using readings from barometric Diver® groundwater level loggers installed in Well_3.

5.2.3.1.6. Discharge

Stream discharge was measured with salt dilution slug test. Known mass of concentrated table salt solution was introduced in the stream. The electrical conductivity was measured at a downstream location where the salt water was expected to be fully mixed within stream flow. Each time a new calibration ratio was measured using stream and salt water samples in the laboratory. Stream discharge was calculated using a simple chemical mass balance equation and linear regression equation relating electrical conductivity and proportion of salt and stream water in the sample. At each occasion between 1 and 10 L of concentrated NaCl solution was introduced into the stream. Diver®CDT groundwater level loggers were used to log specific electrical conductivity with 3 second intervals.

5.2.3.1.7. Chemical composition

Samples for chemical analysis were filtered in the field, colcered in HDPE bottles, transported to the laboratory and conserved with acid to pH < 2 in the laboratory typically within 24 hours after collection as described in TABLE 16. At all times the samples were kept refrigerated at temperature less than 4C. Sample collection and conservation methodology was adapted from methodology issued by the United States Environmental Protection Agency (O'Dell, 1993).

Samples from surface waters were collected directly with 60 to 100 ml syringe. Monitoring wells were purged with a submersible pump followed by sample collection with a peristaltic pump. Electrochemical parameters were measured in an open flow cell with bottom-up water movement direction.

Along with each sample collection in the field water temperature, electrical conductivity, pH, and ORP (oxidation-reduction potential) and dissolved O_2 were determined with appropriate WTW multimeter and probes. In addition, alkalinity was determined in the field by VWR AL-AP field drop counting titration kit.

TABLE 16

Sample collection specification

Bottle	Parameter	Volume (ml)	Filtration (RC 0.45)	Conservation	Parameters determined
1	Cations	15	yes	0,11 ml 63% HNO_3	
2	Anions 1	15	yes	no	
3	Anions 2	15	yes	0,08 ml 98% HSO_4	(NH_4 and NO_3)
4	DOC & Total nitrogen	50	yes	0,28 ml 35% HCl	
5	Total phosphorus	150	no	0,83 ml 98% H_2SO_4	
6	Microcoponds	15	yes	0,11 ml 63% HNO_3	
7	Stable isotopes	>15	No	no	dH, dO

Ionic analyses (Na^+, NH_4^+, K^+, Mg^{2+}, Ca^{2+}, F^-, Cl^-, NO_2^-, Br^-, NO_3^-, PO_4^{3-}, SO_4^{2-}) were performed with high-performance liquid chromatography (HPLC) and macronutrient analyses (C, N, P) using spectrophotometer and carbon/nitrogen analyzer at the Institute of Ecology, Tallinn University.

Microcoponds in water samples were measured by Thermo Scientific Inc. ICP-OES spectrometer iCAP7000 at the Environmental Quality Laboratory, Faculty of Geography and Earth Sciences, University of Latvia. Initial sample run included 30 elements (Al, As, B, B, Ba, Be, Ca, Cd, Co, Cr, Cu, Fe, K, Li, Mg, Mn, Mo, Na, Ni, P, Pb, S, Sb, Se, Si, Sr, Ti, Tl, V, Zn), but only those above detection threshold (Ba, Ca, Fe, K, Mg, Mn, Na, P, S, Si, Sr) were further considered.

Stable isotope ratios of hydrogen and oxygen in water were analyzed in the Laboratory of Environmental Dating at the University of Latvia, Faculty of Geography and Earth Sciences. Isotope ratios are expressed in standard δ-notation relative to the Vienna Standard Mean Ocean Water (VSMOW; Craig, 1961). Both isotope ratios of hydrogen and oxygen were measured using the cavity ring-down laser spectroscopy method (Brand et al., 2009) with a Picarro L2120-i Isotopic Water Analyzer. Reproducibility of stable isotope measurements was less than ±0.1‰ for δ18O and ±1‰ for δ2H. To assure quality of water sampling and processing in the laboratory, internationally-accepted procedures elaborated by the IAEA (Aggarwal et al., 2007) were followed. The laboratory has successfully participated in world-wide open proficiency tests on the determination of stable isotopes in water organized by the IAEA in 2016 (Wassenaar et al., 2018).

5.2.3.1.8. Thermal and visible light imagining from UAV

DJI Inspire 1 with interchangeably mounted visible light DJI Zanusse X3 photo camera and Integrated Thermal Camera DJI Zanusse XT was used.

5.2.3.2. Results and discussion

5.2.3.2.1. Surface and subsurface catchment

The surface catchment area of the Kazu leja was found to be 4,7 km^2 and estimated subsurface catchment area was 3,0 km^2 (FIGURE 46), out of it 1,8 km^2 was attributed to the main group of springs. Notably, the two catchment areas overlap only partly. Bulk of the subsurface water in D_3*pl* dolomite acquire appears to be directed towards the main group of springs forming the origin of Seven Springs' Brook.

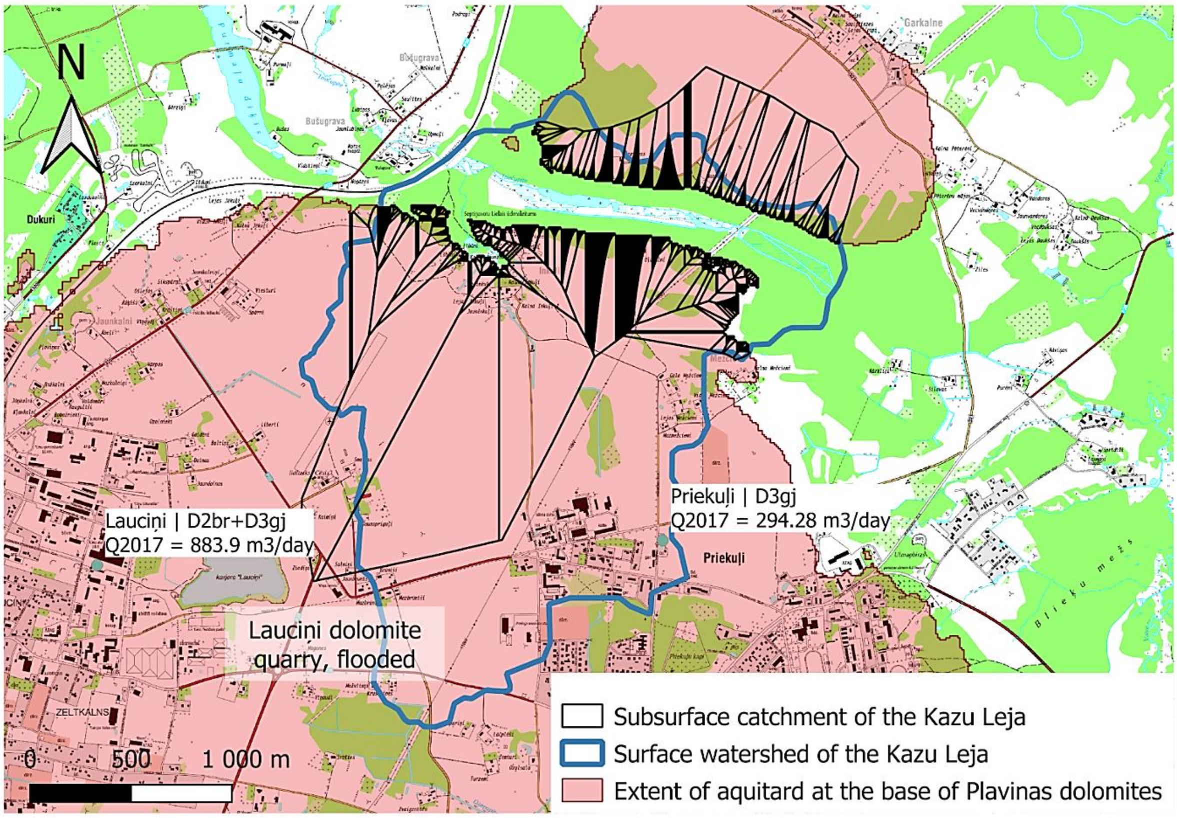


FIGURE 46 **Delineated watershed for Kazu leja, modeled distribution of aquitard at the base of D_3*pl* aquifer and hypothetical subsurface catchment of Kazu leja (open Street base map)**

5.2.3.2.2. Impact assessment

In the past, the GDTEs in Kazu leja were damaged by tufa and peat extraction. Nowadays, these activities are ceased. However, we have identified several other potential human impacts in Kazu leja related with groundwater quality and quantity. These were the nearby municipal groundwater abstraction (FIGURE 46), spring water interception by local homesteads, and extensive arable lands with tile drainage systems around the GDTE site that can capture some of the water flow to the GDTE and can deliver pollutants such as nutrients and pesticides to the GDTE. There was an abandoned dolomite quarry near Cesis as well (FIGURE 46). Theoretically, resuming dolomite extraction could have a direct impact on the spring discharges in Kazu leja. However, that is an unlikely scenario due to the protection status and related national regulations in Gauja National Park.

5.2.3.2.3. Groundwater level and discharge

The reconstructed water table in flooded Lauciņi dolomite quarry between 1997 and 2020 was from 104,9 to 106,2 m asl, e.g. fluctuating by about 1,2 m (TABLE 17). We use this as a surrogate (indicative) groundwater table measurement within Pļaviņas aquifer.

TABLE 17

Estimated water level (m asl) in the abandoned Lauciņi dolomite quarry near Kazu leja

Cycle Np. Of aerial photograph	Date	Water table, m asl
1	30.06.1997	105.75
2	26.04.2003	105.05
3	21.05.2007	104.89
4	11.07.2010	105.08
5	21.06.2013	106.17
6	11.08.2017	105.24
LIDAR	02.04.2019	104.90

Overall, the groundwater table fluctuations reflect sudden rise during precipitation events and gradual drop due to drainage and evapotranspiration. Abrupt changes in the water table in November 2019 and January 2020 were observed in Well_2 and River_2 (FIGURE 47) and absent in other observation points are most likely to construction and destruction of beaver dams.

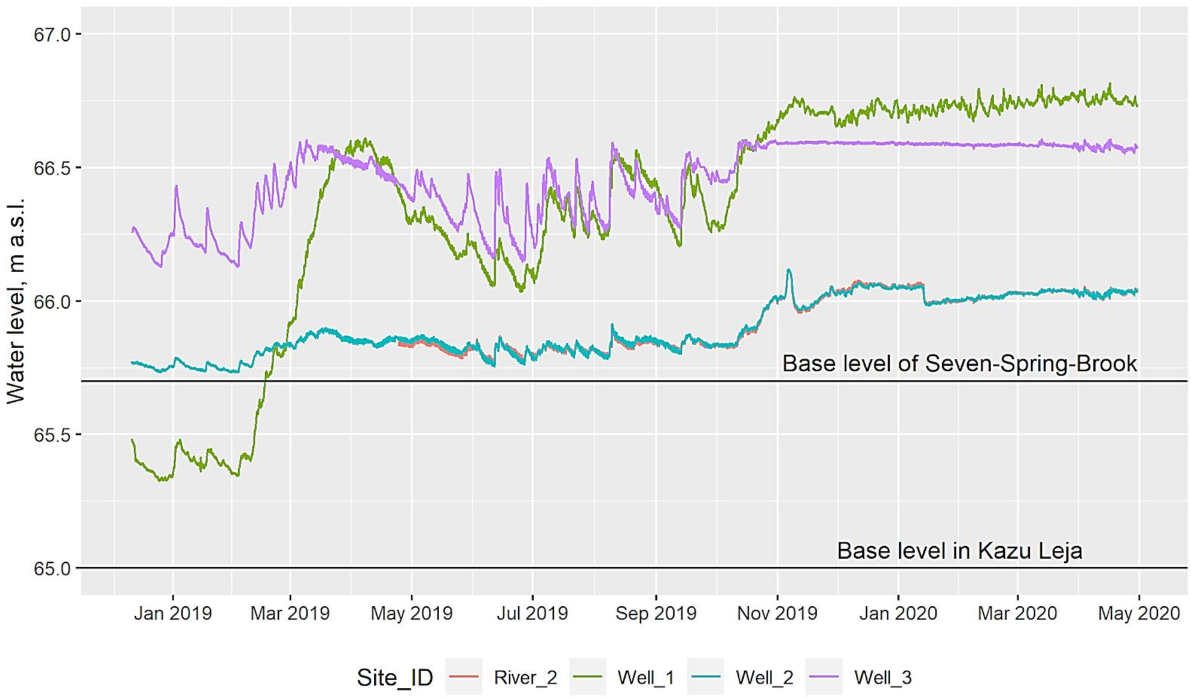


FIGURE 47 **Groundwater and surface water level observations in Kazu leja**

It appears that the groundwater in Kazu leja is infiltrating into deeper aquifers. The water level in Kazu leja at Well_1 (FIGURE 47) in late 2018 and early 2019 was relatively low. In fact, at its lowest point on December 24 2018 it was at 65,30 m asl that was just above the lowest topographical point in Kazu leja – then railway culvert, where the surface elevation was about 65 m asl Apparently the drought of 2018 affected the water table. The water table in Well_1 was below the water table in River Triečupīte in Kazu leja (Well_2 and River_2, FIGURE 47). The elevation of the Seven Springs' Brook passing between the location of Well_1 and railway culvert was about 65,8 m asl, that was higher than the water table in Well_1. Thus, it is likely that groundwater in Kazu leja is seeping into deeper Devonian aquifers discharging within the nearby River Gauja valley.

The difference between observed groundwater table in Well_2 and water table at location River_2 in River Triečupīte next to it was negligible, mostly less than 1 cm. Therefore, we concluded that groundwater in Well_2 located in the fen was hydrologically connected to the River Triečupīte. Due to the close proximity of Well_2 to the river we were not able to make any significant interpretation of overall groundwater movement direction – discharge or recharge.

The observed groundwater table in Well_3 was up to 0.5m below the soil surface in late 2018 and 2019 (FIGURE 47). However, starting from October 2019, the water table reached the top soil level. This pattern reflects the drought conditions in 2018 followed by a period of above-average precipitation in late 2019.

Around the probes installed close to Well-3 upwelling of groundwater was observed in late 2019 and early 2020.

We measure the stream discharge at three locations: Drainage_outlet where the runoff from tile drainage from agricultural fields is collecting; Q2_pond - location on Seven Springs' Brook and Q1_railway – River Triečupīte at its exit from Kazu leja. We have found strong seasonal fluctuation (by a factor of 10) of spring discharges (FIGURE 48) in summer 2019 it was only 4,3 l/s, while in early spring 2020 it reached 68,7 l/s.

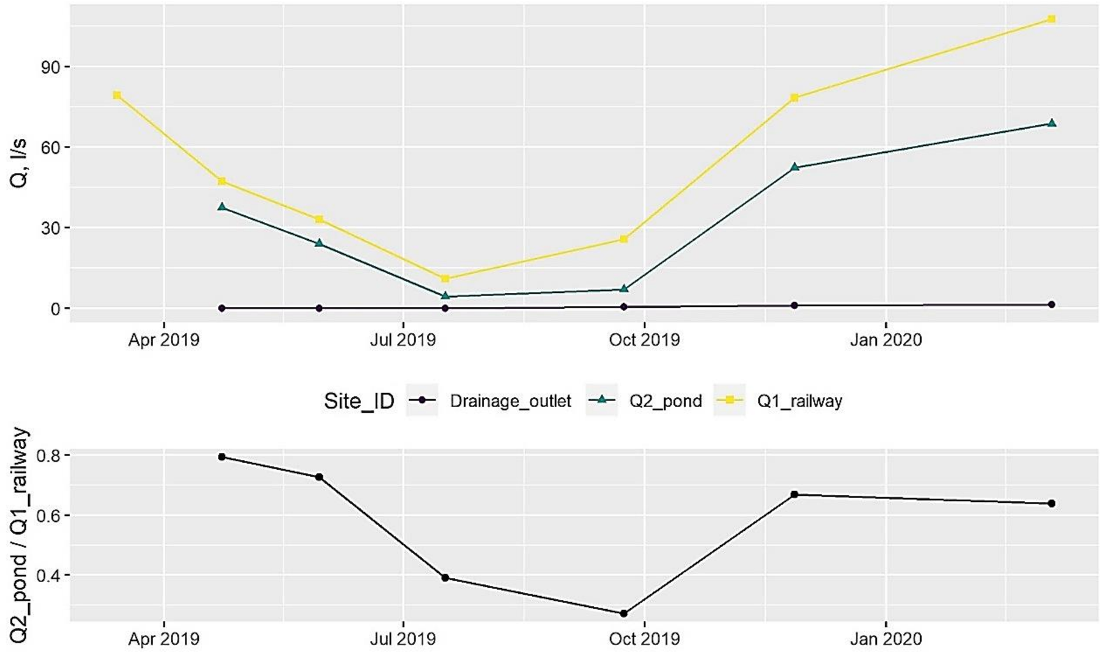


FIGURE 48 **Discharge in Kazu leja**
(Drainage_outlet – where the runoff from tile drainage from agricultural fields is collecting; Q2_pond – location on Seven Springs' Brook; Q1_railway – River Triečupīte at its exit from Kazu leja)

The groundwater discharge to the stream as base flow can be described as a linear reservoir (Hendriks, 2010) where the storage in the reservoir (S, groundwater) is linearly related to discharge from the reservoir (Q):

$$S = \frac{Q}{\alpha} \quad (1), \text{ where}$$

S – storage,
Q – discharge,
α – decay constant.

The observed range of spring discharges within the main group of springs is from 4,3 to 68,7 l/s while the range of groundwater table fluctuation was least 1,2 m (TABLE 17). We have used equation (1) to estimate the value of decay constant α. It can be used as a proxy for average groundwater residence time within the aquifer. The α was estimated to be around 3 years^{-1}. In other words, the average residence time of water in the aquifer was 1/3 years. However, the aquifer might extend below the drainage level. In such a situation the actual average residence time would be larger than estimated. Thus, it can be expected that the aquifer will react rather swiftly to any changes in land use and meteorological conditions.

5.2.3.2.4. Water quality: geochemical facies

All the observations of the chemical composition were subdivided into geochemical facies according to their sampling location (TABLE 18). Geochemical facies were identified by exploratory data analysis and preliminary principal component analysis (not shown here). Results are presented as box plots grouped by geochemical facies and parameter type (FIGURE 49 to FIGURE 51).

One water sample was collected from a shallow dug well of the Kalna Jaunzemji homestead located next to the main group of springs. Its chemical composition was similar to other spring water samples, however the concentrations of phosphorus (TP 1,4 mg/l) and potassium (26 mg/l) was significantly elevated compared to any other samples. This could be due to water contamination during sampling or within the dug well. As there was only one sample collected at this location, it is excluded from further analysis.

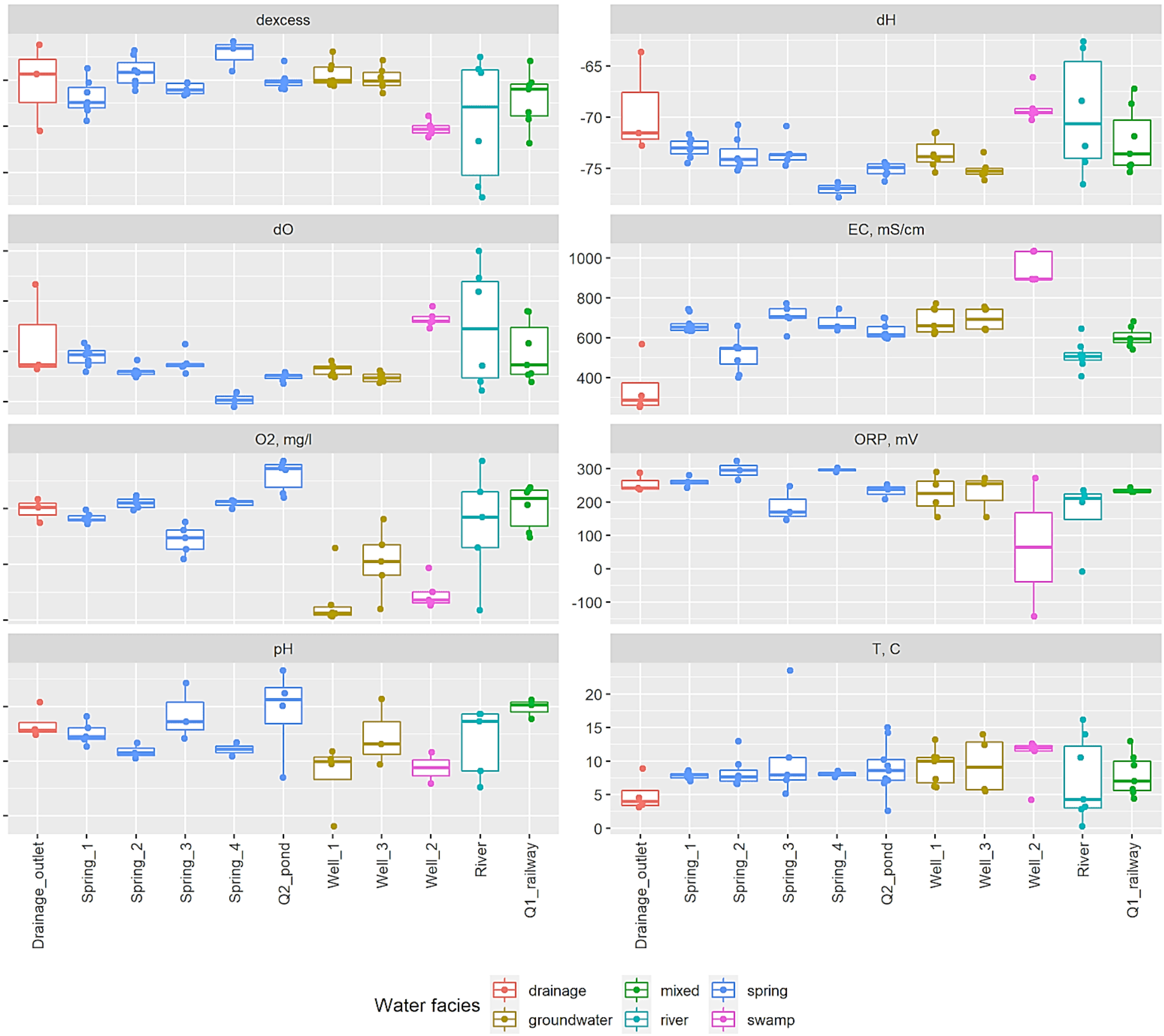


FIGURE 49 **Summary of the field parameters and stable isotope analysis in the water samples from Kazu leja**

TABLE 18

Geochemical facies, short description and corresponding sampling locations

No.	Geochemical facies	Description	Sample site ID	No. of observations
1	groundwater		Well_1	12
2	fen	strongly reduced rich in nutrients, very high alkalinity, high Mg, very high Ca	Well_2	5
3	spring	high alkalinity and Ca and Mg, very high NO3	Spring_1 Spring_2 Q2_pond Spring_3 Spring_4	25
4	drainage	high K and P(?), moderate alkalinity	Drainage_outlet	3
5	river		River	5
6	river and spring	mixing of spring and river facies	Q1_railway	6

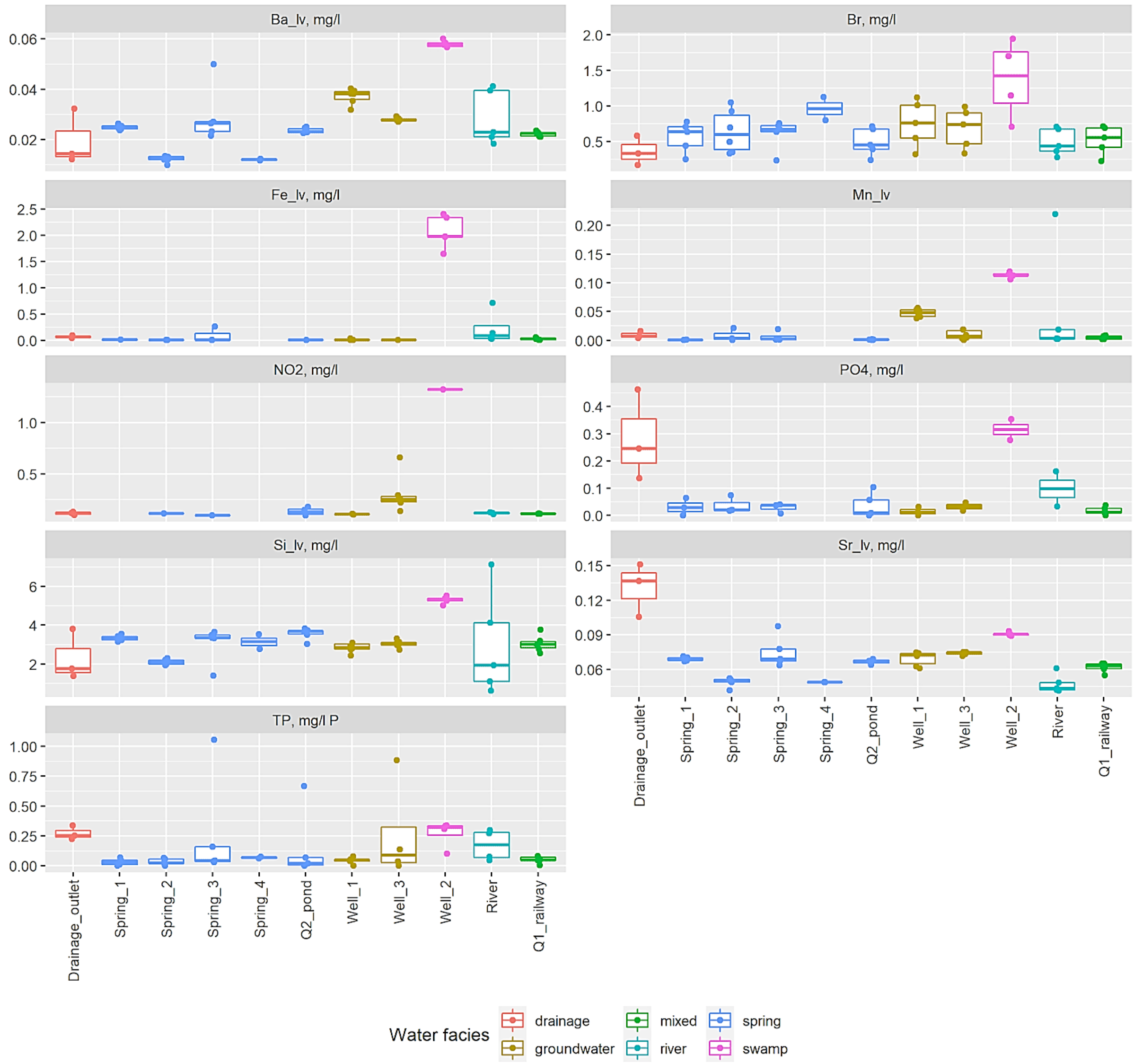


FIGURE 50 **Summary of the micro-component analysis in the water from Kazu leja**

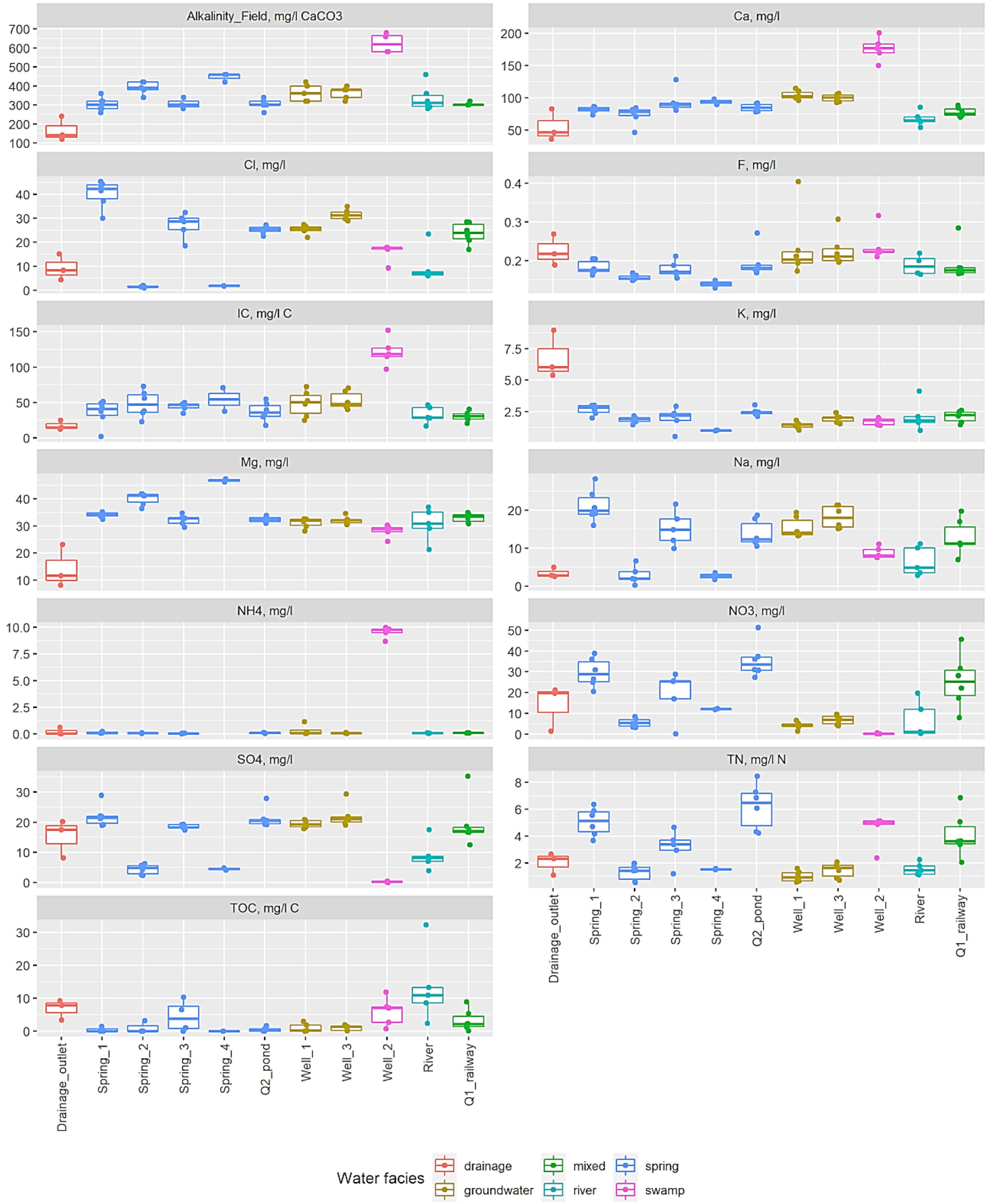


FIGURE 51 **Summary of the field major compound analysis in the water samples from Kazu leja**

5.2.3.2.4.1. Field drainage

Drainage water had an overall lower load of dissolved carbonates (alkalinity, and Ca, Mg ions) compared to most of the other water types (Drainage_outlet, FIGURE 51). This was also reflected in lower electrical conductivity. Load of organic carbon is somewhat higher compared to the other water facies. Chemical composition of drainage water is compatible with fast surface or intermittent runoff of the recent precipitation and short interaction time with minerals.

Elevated concentration of potassium and phosphorus compared to other water types was noted (FIGURE 50) in the drainage water. Potassium loading exceeds even the concentration of sodium (Na). This was interpreted as a direct fertilizer leaking from the cultivated fields. The loading of nitrate was relatively high up to 20 mg/l NO3) but smaller compared to some of the springs.

Notably the loading of Sr is higher in the drainage water as well compared to other water types. That might be due to fertilizer input or, alternatively, differences in mineral weathering in the soil zone mostly composed of glacial loam (drainage water) and dolomites of the Pļaviņas aquifer (spring water).

5.2.3.2.4.2. Springs

Both dissolved oxygen concentration and pH is highest in location Q1_pond, where water initially coming from the main group of springs emerging from a small pond: the average pH in Spring_1 was 7,5 while in downstream location Q2_pond it was 7,9, while average concentration of dissolved oxygen was 9,1 and 13,2 mg/l respectively (FIGURE 49). Green filamentous algae were observed during most field visits at this pond: algae growth was consuming CO_2, resulting in increasing pH value and releasing oxygen, as observed by elevated concentration of this dissolved gas.

Springs in Kazu Leja site had high loading of calcium and magnesium ions and high alkalinity (FIGURE 51) indicative of the dolomite weathering. In fact, the average molar ratio of Mg and Ca ions in Spring_2 and Spring_4 was respectively 0,93 and 0,87 – only a slight excess of Ca over Mg (FIGURE 52). Such a proportion can originate by almost stoichiometric weathering (dissolution) of dolomite ($CaMg(CO_3)_2$). In other springs Ca excess over Mg was greater. That could be due to weathering of calcium carbonate ($CaCO_3$) along with dolomite. In case of Spring_1 and Spring_4 Mg to Ca molar ratio was 0,69 and 0,57 respectively and even lower for groundwater and fen water.

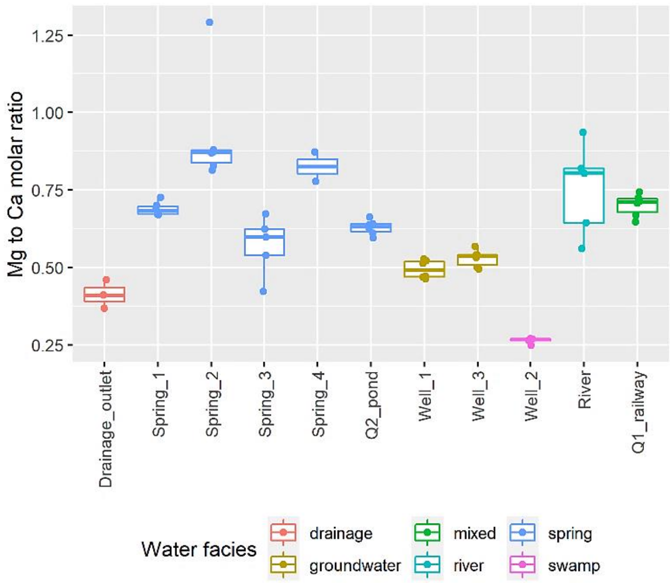


FIGURE 52 **Molar ratio of Mg to Ca in Kazu leja**

Considering other alkali-earth elements we observed an interesting relationship: the concentrations were inversely correlated – higher load of lighter ions was associated with lower load of more heavy alkali earth elements (FIGURE 53). We speculate that this relationship was controlled by the source of ions in the water. If the dolomite weathering is the main source of dissolved minerals then high Mg and low heavy alkali-earth elements – Ba and Sr are expected. However, weathering of calcium carbonate will result in higher loading alkali-earth elements compared to Mg. In fact, the sources of relatively elevated concentration of Sr and Ba in the groundwater and fen water likely was secondary dissolution of freshwater tufa.

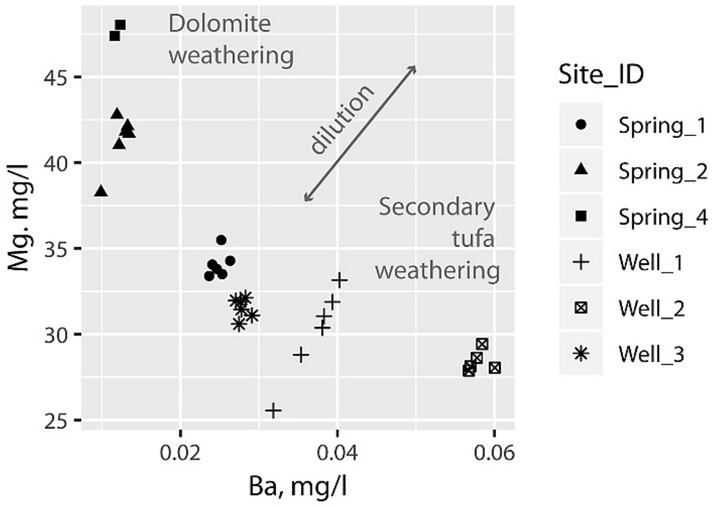


FIGURE 53 **Relationship between light (Mg) and heavy (Ba) alkali earth elements ion concentration in spring and fen water**

Spring_2 had the least EC value for all the springs. That seems to be due to smaller concentrations of Na and Cl as well as NO_3 ions in its waters (FIGURE 51). Concentrations of Na and Cl ions in Spring_2 and Spring_4 were an order of magnitude lower than in Spring_1 (FIGURE 51). The source of elevated chloride ion likely was an artificial source: chlorine-bearing fertilizers or salt application for deicing.

Springs have relatively large range and fluctuations of **isotope ratios** (FIGURE 49) that would be characteristic for event driven recharge or short – few months – residence time of the groundwater. Relatively low d-excess value (FIGURE 49) for Spring_1 suggests its recharge with water affected by evaporative enrichment, that can take place, for example, in short-lived paddles on land surface, due to particular isotope composition of a major precipitation event or contribution of altered snowmelt water to the groundwater recharge.

Spring waters had high loading of nitrates (NO_3, FIGURE 51). At one occasion even the drinking water threshold was exceeded: on 23 April at location Q2_pond it was 51,3 mg/l. The average unweighted NO_3 concentration in the water of the main group of springs at location Q2_pond was 35,6 mg/l. Other springs had somewhat lower nitrate load (FIGURE 51). The loading of the phosphorus was rather low in the spring water not exceeding critical or threshold values.

5.2.3.2.4.3. Groundwater

Groundwater was monitored in two shallow wells: Well_1 and Well_2. Both wells were located in the base of the Kazu leja valley. Generally, the groundwater composition was similar to the spring water composition within the main group of springs. However, the loading of nitrate and dissolved oxygen concentration were lower. In addition, we observed elevated NO_2 concentration in Well_3, indicating ongoing denitrification process: reduction of nitrate.

Ca, Ba and F concentration was slightly elevated compared to springs. The ratios of Mg to Ca and Mg to Ba (FIGURE 53) are lower in groundwater than in spring water. Such observation can be interpreted as additional or secondary dissolution of calcium carbonate (freshwater tufa) in the groundwater.

Well_1 had elevated Mn concentration 0,11 mg/l, compared to less than 0,05 mg/l in all other locations, but this might be the artifact of contamination from metal parts used to suspend the water level logger in the well.

5.2.3.2.4.4. Fen

The highest EC value was found in Well_2 (fen), that is in line with a high load of dissolved calcium and bicarbonate ions (FIGURE 51) that can be the result of intensive weathering of calcium carbonate due to action of organic acids and/or high concentration of CO_2. High loading of Sr and Ba (FIGURE 53) compared to Mg supports this explanation.

The average concentration of ammonia in the fen water was 9,5 mg/l, exceeding the observations in all other locations by an order of magnitude (FIGURE 50). Further, the observed concentration of nitrite ion (NO_2^-) in Well_2 (fen) was the highest in all sampling locations, but the nitrate (NO_3^-) was virtually absent. This might indicate a significant denitrification process of high-nitrate spring or groundwater penetrating into the peat layer.

Other biogenic elements like Si and P loading was among the highest observed in this study as well (FIGURE 50). That can be interpreted as the result of ongoing decomposition of organic matter (peat) releasing nutrients like nitrogen and phosphorus into the pore water.

Furthermore, concentration of elements mobile in anoxic conditions – iron and manganese – were elevated as well. This was compatible with highly reducing conditions within the swamp. In such conditions we would not expect to find any measurable concentration of dissolved oxygen in the water. Therefore, it was highly likely that the observed average O_2 concentration in fen water (2,4 mg/l) was the result of contamination during the sampling.

The bromine (Br) concentration in fen water was highest among observation points. But the reasons for these phenomena were not clear.

The fen water had stable isotope ratios characteristic for enrichment by evaporation (FIGURE 54) in intermediate position between river and groundwater with little seasonal changes. This can be interpreted as a result of relatively slow water exchange between peat and open river water with residence time at least a year.

The fen water composition appears to be affected by peat decomposition; that in turn might be the result of heavy disruptions brought about by peat extraction and drainage works. It appears that the River Triečupīte had been straightened and deepened at some point during the 20th century, but no significant water management activities had been taking place there for at least the last 30 years. Thus, the hydro-biogeochemical system has not reached an equilibrium even decades after the disturbance.

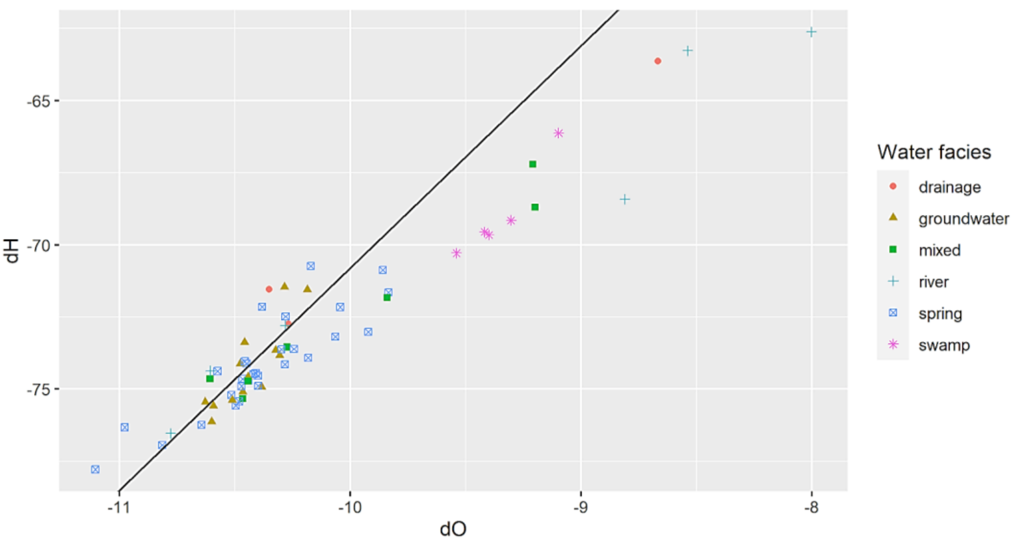


FIGURE 54 **Water stable isotope ratios in Kazu leja, diagonal line visualizing the local meteoric water line as calculated from GNIP stations at Riga, Tartu and Ramata (GNIP, 2020)**

5.2.3.2.4.5. River and mixed water

EC was low for the river water (River, 513 µS/cm) compared to spring and groundwater (630 and 689 µS/cm respectively). River Triečupīte runoff was formed by direct precipitation and groundwater discharge represented by Spring_2, Spring_3 and Spring_4. Thus, it was reasonable to expect that its load of dissolved minerals controlling EC value originating from springs (FIGURE 49) would be diluted somewhat by direct precipitation input. Surface waters in Kazu leja (river in FIGURE 54) demonstrate seasonal enrichment of heavy isotopes due to evaporation, indicating that the water residence time at most was a few months.

We consider the water samples collected from River Triečupīte at the railway culvert mix of the river water delivered by Triečupe and spring water delivered by Seven Springs' Brook. As such it is expected to have chemical composition intermediate between these two sources. An attempt was made to construct a mixing model using measured discharges to validate this assumption. However, the results were rather inconclusive. Using conservative ions such as Cl^- as tracers did not provide predicted results in agreement with observations (FIGURE 55).

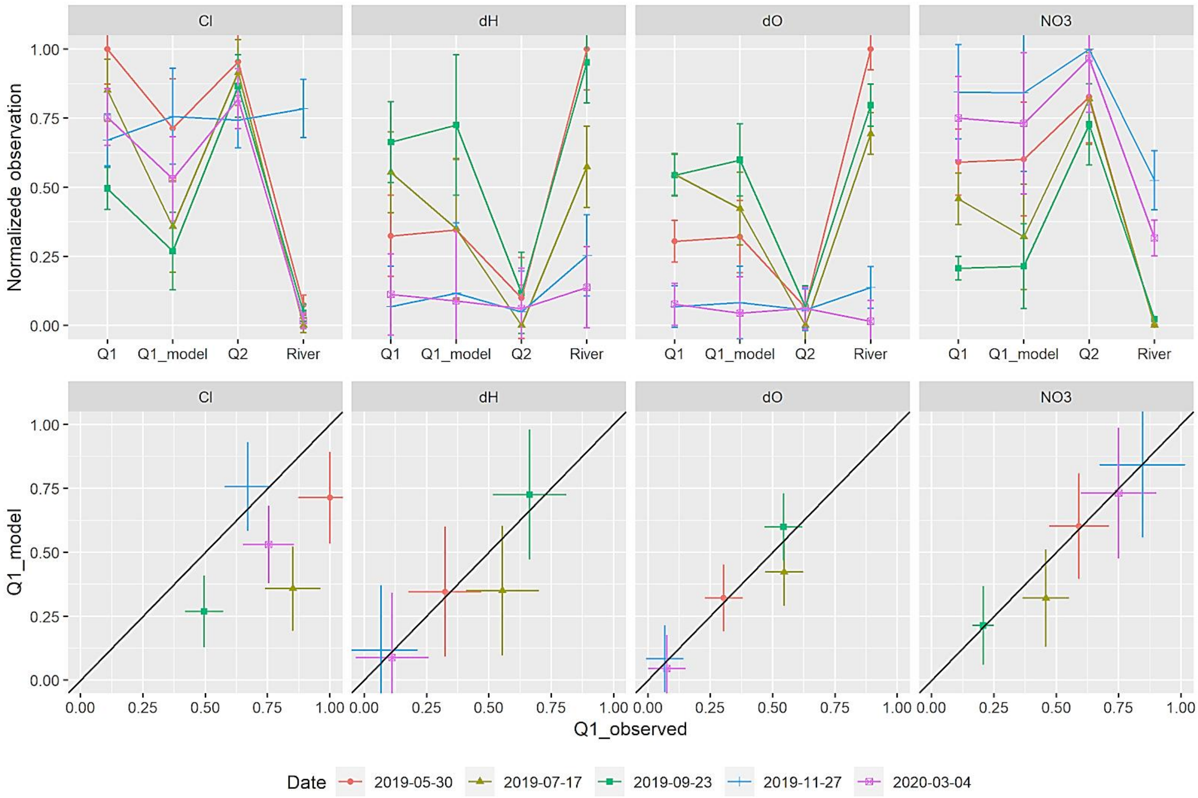


FIGURE 55 **Results of the mixing model considering Cl ion, d18O, d2H and nitrate (NO_3^-) as passive tracers (on the top – relationship between predicted and observed normalized values at location Q1_railway; on the right – predicted and observed values at Q1_railway, River and Q2_pond (Q2))**

The result of the mixing model indicated that our observation campaign did not fully capture some of the geochemical phenomena in Kazu leja. Most likely there was a water source not captured by observations network, though affected by anthropogenic Cl^- input.

5.2.3.2.5. Thermal and visible light imagining from UAV

The Kazu leja area was photographed in both infrared thermal and visible radiation, however it was not possible to compile a proper orthophoto map of the area due to steep terrain conditions and presence of large trees. In thermal images (FIGURE 56) it is possible to distinguish individual groundwater discharge locations; however, the results were not as useful as was expected. Most notably even in an overcast winter day different surfaces exhibit contrasting brightness in thermal radiation making any deciphering of minor, otherwise unnoticed groundwater discharge sites challenging.

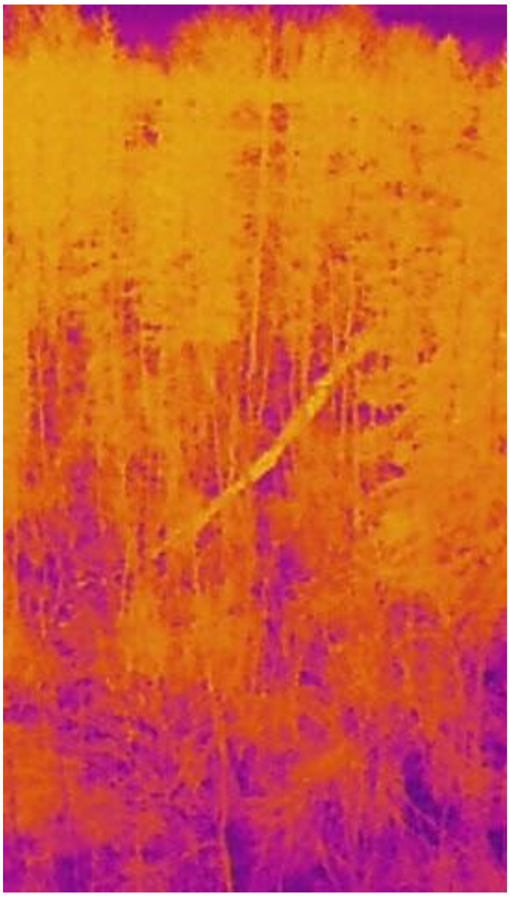


FIGURE 56 **Panorama view in thermal infrared radiation on the southern slope of Kazu leja (the bright diagonal line is a channel of a spring discharging at the upper part of the slope) (Photo: A.Markots, 2019)**

5.2.3.2.6. Surface water quality indicators

Background values of biogenic elements in surface water as reported in Annex 4.3 of the River Gauja basin management plan for 2016–2021 (*LVĢMC, 2015. Gaujas upju baseinu apgabala apsaimniekošanas plāns 2016.–2021.gadam)* are presented in TABLE 19. According to regulation issued by the Cabinet of Ministers No.834, December 23, 2014 "*Prasības ūdens, augsnes un gaisa aizsardzībai no lauksaimnieciskās darbības izraisīta piesārņojuma*" developed in line with the EU Nitrates Directive (91/676/EEK), the threshold value of nitrate in the surface water is 50 mg/l (11.3 mg/l $N\text{-}NO_3$).

TABLE 19

Natural background (reference) values of biogenic elements in surface water as reported in Annex 4.3 of the Gauja river basin management plan for 2016-2021 (LVĢMC, 2015)

Water body type	Dissolved O_2, mg/l	$N\text{-}NH_4$, mg/l	N_{tot}, mg/l	P_{tot}, mg/l
Rapid small river (strauja ritrāla tipa maza upe): Seven Springs' Brook and River Triečupe at the exit from Kazu Leja	>8,0	<0,09	<1,5	<0,040
Slow small river (lēna potamāla tipa maza upe): River Triečupīte within Kazu leja	>7,0	<0,10	<1,5	<0,045

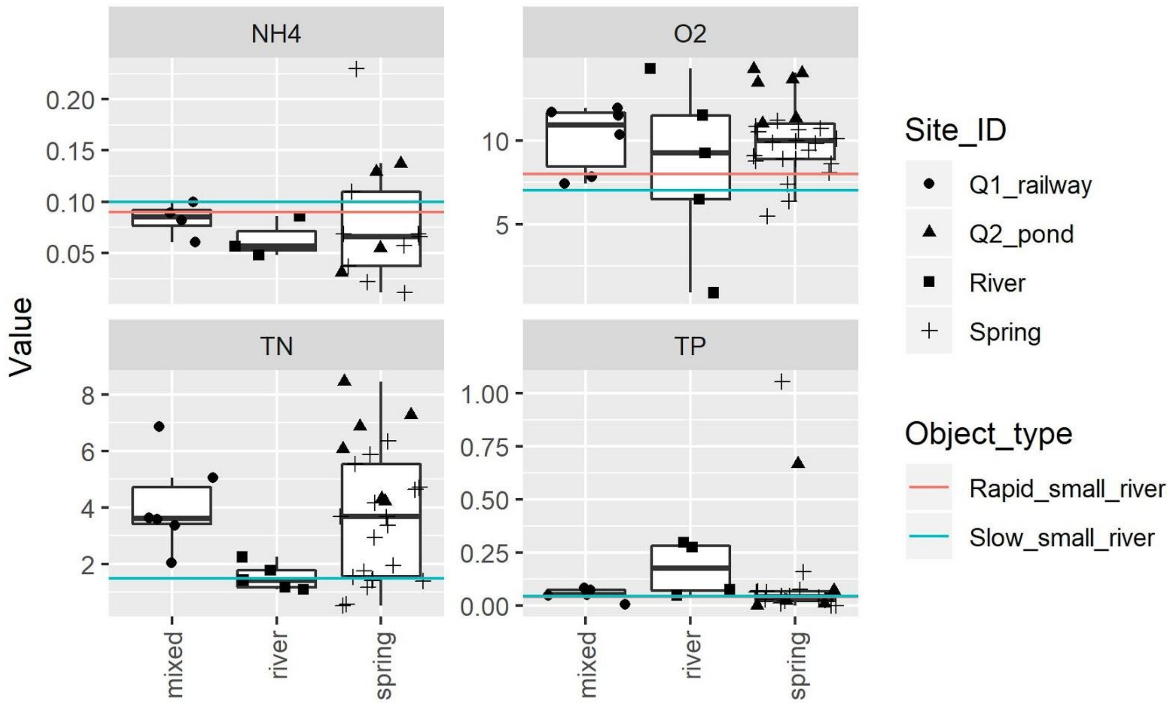


FIGURE 57 **Comparison of the background (natural) concentrations of biogenic elements in the surface waters (horizontal line, LVĢMC, 2015) and those observed in Kazu leja**

Observed oxygen concentration was mostly above the background (natural) values of small rivers (LVGMC, 2015; FIGURE 57). However, in summer 2019 (July 17) observed dissolved O2 concentration in River Triečupīte was only 0,9 mg/l, probably reflecting the influence of algal bloom or disruption of moody river bed sediments.

In most cases, the total nitrogen concentration exceeded the natural background concentration in both the springs and River Triečupīte at its exit from Kazu leja. The nitrate load in spring water was very high as well, in fact at one occasion it exceeded the drinking water quality threshold (50 mg/l). The nitrate load was the highest in the main group of the springs (Spring_1 and Q2_pond) and elevated in other springs draining the dolomites as well (Spring_2 and Spring_4). Most probably, the source of nitrate pollution is the intensively farmed fields surrounding Kazu leja.

However, the nitrogen load in River Triečupīte above the confluence with Seven Springs' Brook was close to the natural background, despite the fact that it receives part of its water from springs (Springs_2, Spring_3 and Spring_4) affected by agricultural pollution. Apparently, within the river headwater noticeable nitrate attenuation either due to dilution, uptake by plants or denitrification was taking place. If this is the case, it can be argued that beaver dams, increasing extent of ponds and fens on the floor of Kazu leja facilitate natural water purification.

In spring water and mixed water in most cases the total phosphorus loading (FIGURE 57) was close to the natural background levels. Most likely, it was due to bonding of phosphate with calcium forming poorly soluble minerals.

However, the background level of total phosphorus (0,045 mg/l) was exceeded in all observations in the water of River Triečupe (FIGURE 57, the average was 0,17 mg/l) above its confluence with Seven Springs' Brook. The median concentration of the total phosphorus in spring water was 0,041 mg/. The two extreme values in springs were observed in Q2_pond where water composition could be affected by decomposition of algae or admixture of drainage water and the other in Spring_3 that was located in the valley floor and water was emanating from peat deposits.

The highest average concentration of total phosphorus (0,27 mg/l) besieged agricultural drainage (Drainge_outlet) was observed in the fen (Well_2). Thus, the high phosphorus load in river water was likely due to the same reason as observed in the fen water – peat decomposition. Within Kazu leja River Triečupīte in most of its course was in direct contact with peat that mostly was disturbed during the previous century, providing plenty of opportunities for enrichment with phosphorus compounds.

5.2.3.2.7. Nutrient runoff: source-pathway-receptor (SRP)

Nitrate load of up to 50 mg/l and average 35,6 mg/l were found in the water from the main group of springs, indicating heavy nutrient leaching from arable lands. In addition, nitrate runoff from arable lands had pronounced seasonality - highest in winter and least in summer (TABLE 20). Taking into account catchment area the corresponding unweighted average intensity of nitrate leaching was approximately 243 kg ha-1 year-1. This is a significant value than that for average nitrogen fertilizer use in Latvia in 2012-2015 expressed as nitrate equivalent: 146 kg ha-1 (Vides aizsardzības un reģionālās attīstības ministrija, 2016).

TABLE 20

Estimated catchment area and area-specific nitrate runoff in Kazu leja

	Catchment area (km^2)		Average discharge (l/s)		Average NO_3^- concentration mg/l		NO_3^- runoff (kg/ha season)	
	D_3*pl* aquifer	Other	Cold season	Warm season	Cold season	Warm season	Cold season	Warm season
The main group of springs	1,8	0,0	52,8	11,8	41,7	29,9	217	26
The rest of Kazu leja	1,2	0,8	25,3	11,5	16,0	0,8	41	1,5

Our monitoring programme does not cover all the springs discharging in Kazu leja, however land use and geological conditions for catchments of all the springs are similar. Therefore, we can speculate that the water composition in non-observed springs was similar. If so, then noticeable retention of reactive nitrogen takes place in the wetlands at the base of Kazu leja during the warmer season of the year, as the nitrogen load in the outflowing water – River Triečupīte – is moderate compared to that observed in springs (FIGURE 58).

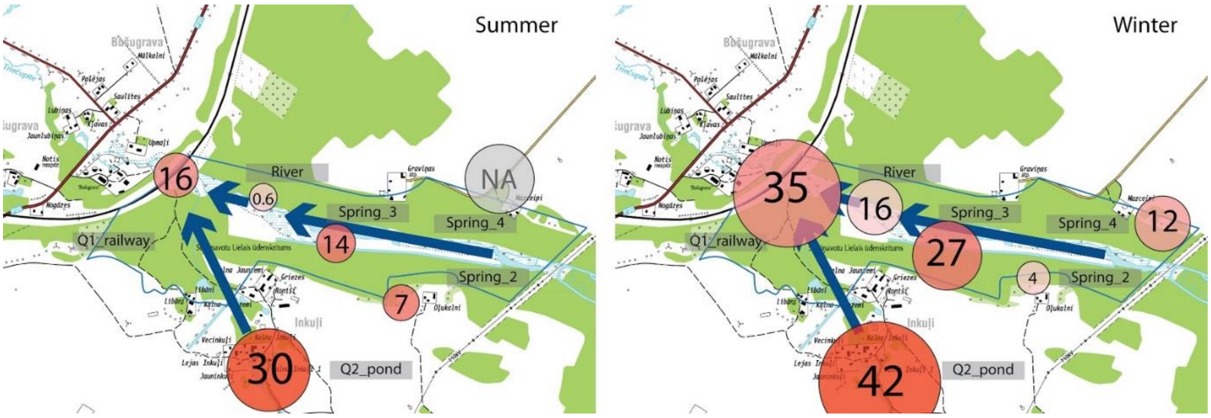


FIGURE 58 **Comparison of average observed concentrations of nitrate (NO_3, mg/l) during the warm season (May-September) and cold season (October-April) in waters of Kazu leja**

5.2.3.3. Conceptual hydrogeological model

Springs distributed along the slopes of Kazu leja were delivering most of the water to Kazu leja site. Springs discharging along slopes of Kazu leja are draining karstified aquifer formed by fissured dolomites of upper Devonian Pļaviņas formation (FIGURE 59). It was speculated that groundwater might discharge from sand interlayers in underling clastic Devonian Amata and Gauja formations as well. However, modelled (Spalviņš, 2016) and observed groundwater head in water abstraction wells opening in the lower part of Gauja formation is below the base of Kazu leja valley, thus groundwater contribution from this source is unlikely. Springs found in the base of Kazu leja apparently are fed by re-infiltrated water from springs located higher on valley slopes. This is indirectly supported by similar stable isotope ratios and chemical composition of the springs on valley slopes and base. It is possible that groundwater at the base of Kazu leja was recharging deeper aquifers.

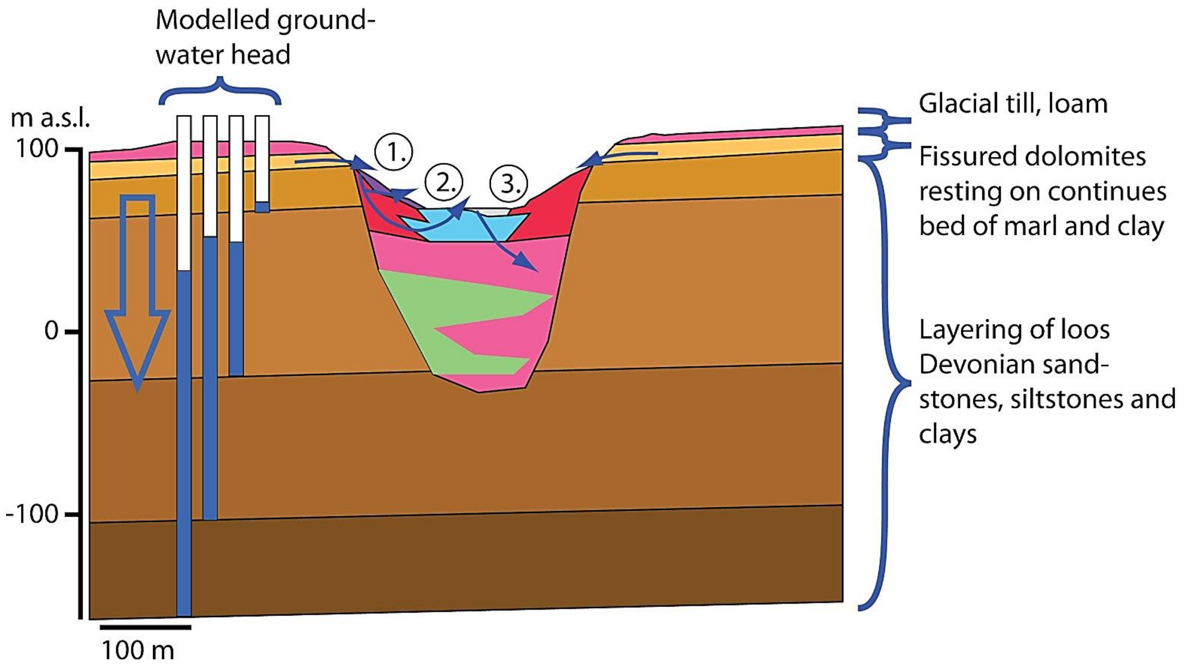


FIGURE 59 **Conceptual hydrogeological model of Kazu leja (1. – springs are discharging from D₃*pl* fissured and karstified dolomite aquifer; 2. – some spring water is re-infiltrated into the quaternary deposits at the slopes of Kazu leja to reappear on earth surface as focused springs or diffuse flow at the base of Kazu leja; 3. – some of the groundwater in Kazu leja likely is percolating towards deeper aquifers; modeled groundwater head from Spalvins et al., 2016)**

5.2.4. Conclusions

1. Primary groundwater source in Kazu leja is unconfined Upper Devonian Pļaviņas aquifer composed of fissured and karstified dolomites. Some of the spring water emerging along upper reaches of Kazu leja was infiltrated into the slope deposits to re-emerge at land surface at the base of the Kazu leja valley. D_3*pl* aquifer recharge were of the Kazu leja springs were the undulating moraine plains (plato) around Kazu leja, with estimated area approximately 3,0 km^2, in addition to 0,8 km^2 of land surface where D_3*pl* aquifer is not present.
2. Springs discharging in Kazu leja had strong seasonality – the highest in winter and the lowest in summer. That may be due to natural fluctuations of groundwater recharge. Average groundwater residence time in D_3*pl* aquifer can be as little as 1/3 years. There was no evidence that groundwater abstraction was reducing groundwater discharge in Kazu leja.

3. From a hydrological and ecological point of view, Kazu leja can be considered as a complex GDTE that consists of cascading petrifying springs and spring-fed fens that are inter-linked by brooks, ponds and subsurface flow of re-infiltrated groundwater. The history of land use and the current condition of ecosystems suggest that this type of groundwater dependent ecosystems may be capable of recovery if the groundwater supply is not disrupted.
4. The main anthropogenic pressure related with groundwater to the GDTEs was nutrient leakage from intensively cultivated arable land that makes up the bulk of the groundwater recharge area. Unusually high nitrate concentration in spring water was found, most likely originating from fertilizer applied to the arable lands in the spring catchment. Nitrogen concentration appears to be facilitated by well aerated conditions of the D_3*pl* aquifer. Nitrate concentrations were lower during the summer and higher in winter. It appears that almost complete reactive nitrate retention was taking place in wetlands of Kazu leja during summer. However, this process did not affect water discharging from the main group of springs as it bypassed most of the wetlands in Kazu leja. However, no evidence of deteriorating nitrogen impact on the explored biodiversity elements (vegetation, mollusk fauna) was found.
5. Applying source-pathway-think approach nitrogen retention in Kazu leja was identified. It is a useful concept, facilitating the understanding of interactions in the GDTEs and deserves to be part of site studies.
6. In water of spring-fed fen we found evidence of secondary dissolution of tufa increasing concentration of alkali earth elements (Ca, Sr and Ba) and indications of peat decomposition releasing of nutrients (N and P) in the spring-fed fen groundwater at the base of Kazu leja. Evaporative signature and lack of distinct seasonality in stable isotope ratios indicate that the water residence time in wetland exceeds one year.
7. Ratios of alkali earth element concentrations (Mg, Ca, Sr, Ba) had provided some unusual insight into weathering of carbonate sediments (dolomite, tufa) in Kazu leja, contributing to our understanding of the ongoing processes.
8. Chloride ion is often considered as a conservative parameter, useful for tracer studies, however it appears that due to wide application of chloride salts in agriculture and road deicing, have significantly increased its concentration in some parts of the hydrological system, rendering it as inappropriate as conservative tracer. Water balance calculations using discharge and chemical tracer measurements can help to assess if all significant water sources have been identified.

5.2.5. Assessment of Kazu leja GDTE according to the schemes of quantitative and qualitative effect

The collected dataset allows to evaluate if the changes in groundwater quantity and quality of the groundwater body D6 of Pļaviņas-Amula Upper Devonian multi-aquifer system in the Gauja-Koiva river basin has caused significant damage to the GDTE complex in Kazu leja.

5.2.5.1. Quantitative effect

Step 1 – *Is there any piece of evidence that water level in the GDTE is considerably lower than it has been previously or is typical to analogous ecosystems, or the conservation status of the GDTE is worse than “good” according to the assessment based on the Habitats Directive?*

The assessment of the quality of the spring habitat patches in Kazu leja varied from medium to good (4-point scale from low to excellent). The excellent status was not awarded because some spring habitats are of secondary origin, still not fully recovered and/or are under trampling impact, and/or are invaded by alien plant species (none of these impacts are related with groundwater). Only in one patch, the spring flush and characteristic moss cover was completely dry in spring-autumn 2019 due to prolonged drought, e.g. the characteristic vegetation and tufa precipitation was unfavorably affected by reduced groundwater discharge. In the other patches, the water discharge amount was reduced, but without visible changes in vegetation structure and composition, i.e. the ecosystem did not quickly respond to temporary drop of water amount. Significant fluctuation of the discharge from the main group of springs was observed: in summer 2019 it was only 4,3 l/s while in early spring 2020 already 68,7 l/s. It is likely that other springs had similar magnitude of discharge variations. In early 2019 the groundwater table within Kazu leja was noticeably below the surface of the nearby fens. However, the fens there receive their water from springs located along the slopes of the Kazu leja and did not appear to suffer from drought.

The vegetation and other habitat characteristics recorded in summer 2019 may lead to a conclusion that the overall habitat conservation status of petrifying springs (habitat 7220*) in Kazu leja is good. However, evidence of reduced amount of groundwater discharge in one of the spring flushes is a caution that the water amount discharged in the springs may be lower than before, i.e. the typical spring vegetation has developed on tufa deposits, the typical species are present, but throughout the summer 2019 the spring flush was dry.

In a "normal situation", since the fen would not be considered an Annex I habitat, its condition would not be assessed within Natura 2000 monitoring. In early 2019, the groundwater table in the valley was noticeably below the surface of the fen. However, the fens there receive their water from springs located along the slopes of the Kazu leja and did not appear to suffer from drought. No evidence of deterioration in the fen was found.

Considering the low amount of precipitation in 2018 and first half of 2019, we can assume that the drought impact may be temporary, without a long-term impact on GDTE. But this cannot be proved without long-term monitoring or at least previous habitat survey data.

Verdict – considering the evidence of groundwater-related impact on ecosystem, though only in one of the 7220* habitat patches, we can move to Step 2.

Step 2 – *Is there relevant groundwater abstraction in the vicinity of the GDTE? Groundwater abstraction rates considered relevant depend on the distance from the GDTE.*

It is proposed that simple as possible assumptions are used to initially assess if groundwater abstraction can have impacts on GDTEs (FIGURE 15, Step 2). One possibility is to estimate the area needed for the groundwater recharge to compensate for groundwater abstraction.

There are several municipal groundwater abstraction sites near Kazu leja (FIGURE 60) and one of them (Priekuļi) are within surface catchment area of the Kazu Leja. At these sites' groundwater is abstracted from Middle-Devonian GWB A8 Gauja and Burtnieks formations. It was estimated that the area needed to replenish the groundwater resources by natural recharge (FIGURE 60) does not overlap significantly with the subsurface catchment area of the springs discharging in Kazu leja.

Verdict – There is no evidence that the municipal groundwater abstraction from Middle-Devonian A8 GWB has caused significant reduction of the groundwater discharge within Kazu Leja.

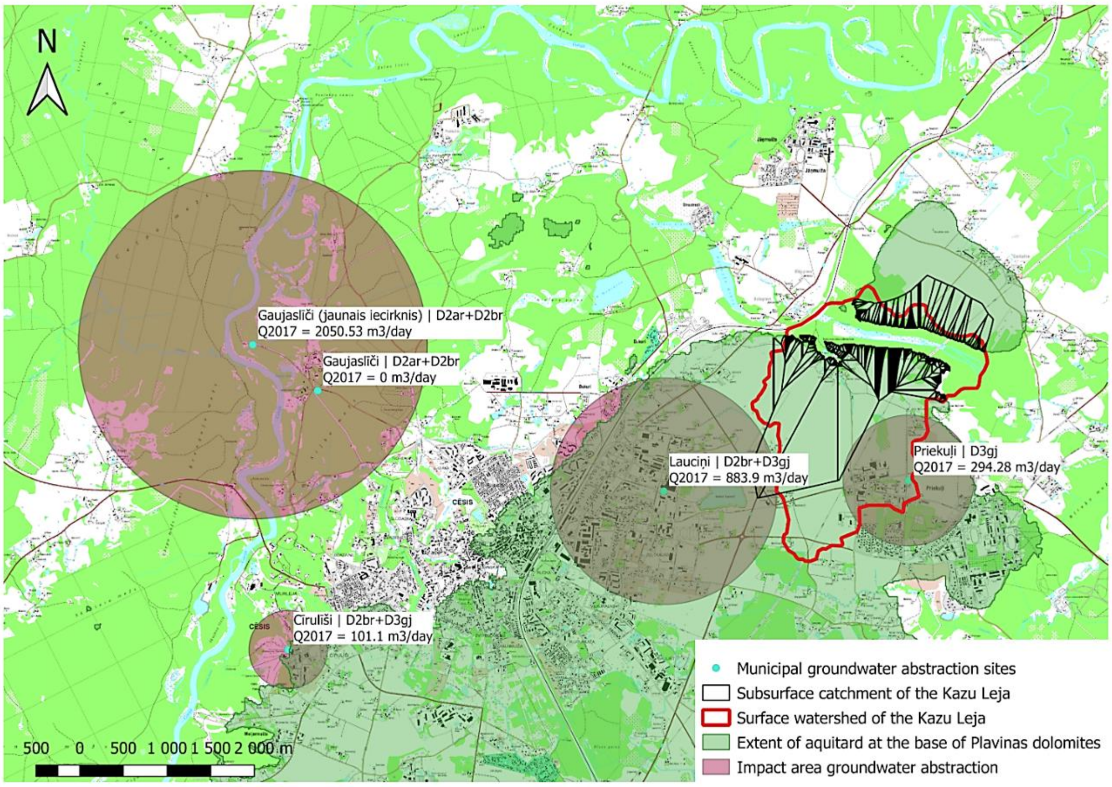


FIGURE 60 **Estimated areas for replenishment of the groundwater resources around municipal water abstraction sites: groundwater abstraction rate as in 2017 and groundwater recharge equal to 1/4 of the subsurface runoff from ERA5 reanalysis**

5.2.5.2. Qualitative effect

Step 1 – *Is there any piece of evidence that the GDTE is in a considerably worse condition than it has been previously or is typical to analogous ecosystems, or the conservation status of the GDTE is worse than "good" according to the assessment based on the Habitats Directive?*

No. The species composition in the spring flushes did not indicate increased nutrient level or other potential impacts related with pollution. However elevated concentration of nitrate ion (NO_3^-) in spring water was detected during the pilot study.

Verdict – In a "normal situation", within Natura 2000 monitoring using the parameters in the standard data form for petrifying springs, the impact of increased nitrogen level would not be, most probably, detected. This requires water sampling and analysis in the laboratory. As elevated nutrient load was detected by chemical analysis it was decided to move on to the next step of the assessment scheme.

Step 2 – *Is there any relevant and possibly polluting human activities in the vicinity of the GDTE?*

Yes, there are fertilized fields within the recharge area of the springs. Fertilizers applied to fields may enrich shallow aquifers with nutrients.

Verdict – there is evidence that the spring water chemical composition was affected by human activity.

Step 3 – *Has the deterioration of the conservation status of the GDTE been caused by changes in the water chemistry (N_{tot}, P_{tot}, nitrates, etc.)?*

GDTE assessment did not reveal deterioration of its status due to changes of water composition (increased Ntot).

Verdict – increased fertilizer loading appears not to affect the status of GDTE at this location, as it was already determined at Step 1.

5.2.6. 3D geological model of the Kazu leja site

Three dimensional (3D) geological models are a useful tool for both communicating complex geological information and forming a basis for more elaborated modeling tasks such as modeling subsurface movement of water – hydrogeological modeling. Here we present a 3D visualization of the conceptual geological model of the Kazu leja site. The aim of this work was to demonstrate the complexity of subsurface environment and to enhance communication between stakeholders.

We were building on our previous experience in construction geological and hydrogeological model of the Baltic Artesian Basin (BAB) spanning the territory of three Baltic states. The visualization presented here was prepared by using WebGL technology and three.js JavaScript library implemented in QGIS qgis2threejs plugin. The interactive geological model is available at the website of University of Latvia: https://www.gprm.lu.lv/kazugrava-3d/.

5.2.6.1. Methodology

Initial geological structure of Kazu leja was created within the *MOSYS* modelling system (Virbulis et al., 2012). MOSYS modelling environment was applied in development of the 3D geological model of the Baltic Artesian Basin (BAB) (Virbulis et al., 2012, 2013). This is a script-based modelling process. Since during the development of the geological model of the BAB majority of available geological data sources were already collected and processed, initial geological structure of the Kazu leja site was created as sub-model of the BAB on the utilizing previously developed algorithm. Additions to this algorithm includes the updated extent of model area, updated model stratification and particular new data sources.

Initial geological structure was built layer-wise on triangular mesh. Topography of each layer was interpolated from data denoted in borehole lithologic and stratigraphic descriptors. Linear interpolation in surface construction was used. To eliminate the effect of layer thickness change in presence of glacial erosion, only those borehole descriptors describing non-eroded layer surfaces were used (Popovs et al., 2015). Erosion effect on geological structure was considered through topological integration of the entire depositional sequence against Sub-quaternary. As a result, topological relations of all model layers were controlled automatically by relationships between sedimentary layers and Sub-quaternary erosion interfaces – modelled topography of subjacent layers are wedged out in intersects with erosion surfaces or terminated along fault lines (Popovs et al., 2015).

To develop web-based visualization of 3D geological structure of Kazu leja site, *WebGL* technology (https://get.webgl.org/) and *three.js JavaScript* library (https://threejs.org/) implemented in *qgis2threejs QGIS* (https://github.com/minorua/Qgis2threejs) plugin was used. It was accomplished by rasterizing 3D vector data of previously constructed finite element model structure through simple GIS routines – exported elevation values of each surface for all nodes of triangular mesh was linearly interpolated on a regular grid. Created georeferenced DEMs of all model layers were stacked together in proper geological order. Further all elements of the developed model were exported to data format suitable for web publishing allowing to access the entire model and interact with it using a simple setting panel for all interested parties.

5.2.6.2. Model structure

The Kazu leja local topographical structure represents about 3 km long, 1 km wide and up to 42 m deep valley like depression connecting the valleys of Gauja and Vaive rivers. Topography of valley partly mirrors paleo-incision system incised into dolomites of upper Devonian Pļaviņu Fm which outcrops in valley banks and sandstones, siltstones and clays of underlying middle Devonian Amata and Gauja Fms. Following the scarce borehole data information and previous geological interpretations, it is believed that the deepest part of the valley reaches the deposits of middle Devonian Burtnieki Fm.

As a model extent the surface watershed of River Triečupīte emanating from Kazu leja with an area of about 6 km2 was chosen. To represent the majority of topographic features and wedging of sedimentary layers along paleo incision, resolution of initial triangular mesh is defined with an average edge of triangle 5 m long. To maintain the characteristics of stepped topographic surface, 1 m grid resolution was chosen in translation of model structure to GIS environment.

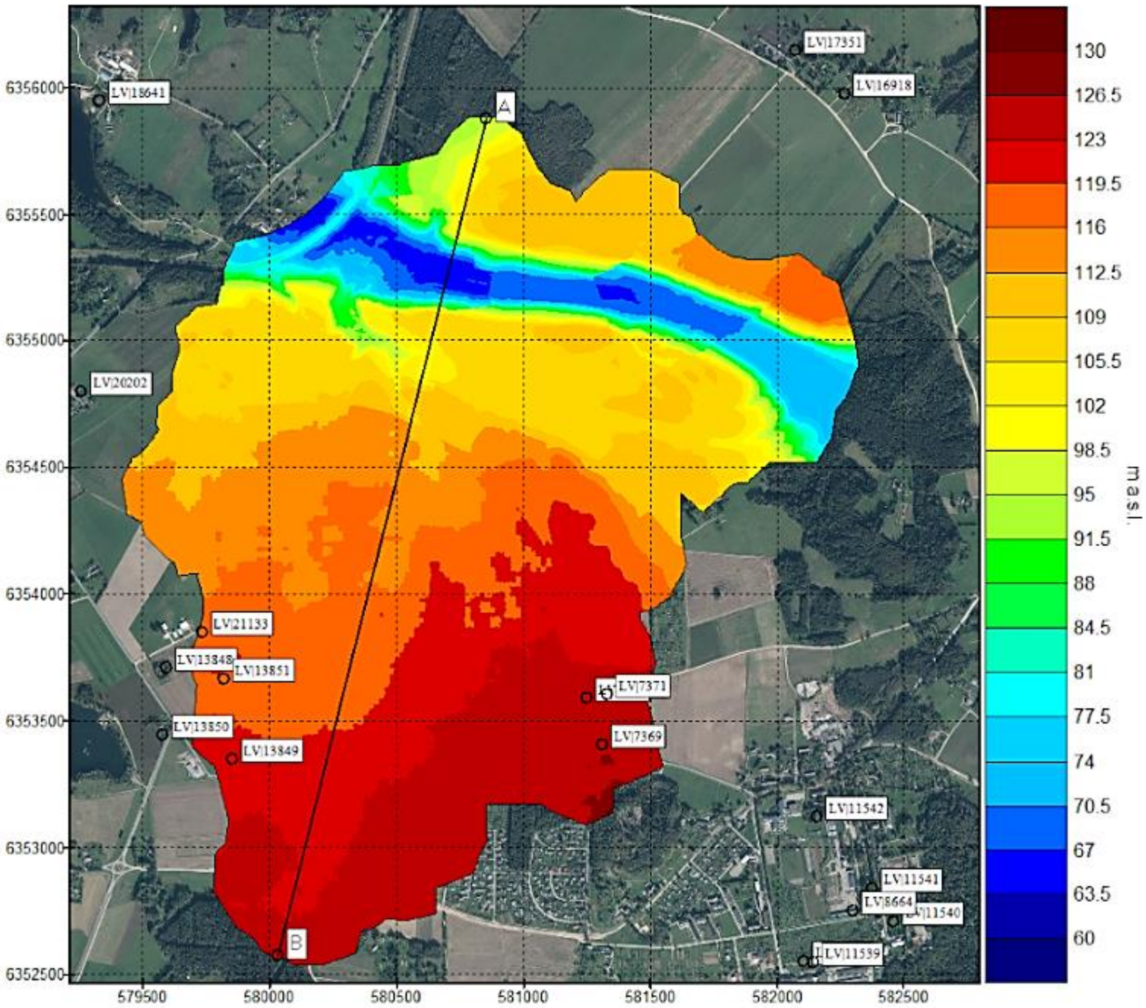


FIGURE 61 **Representation of Earth surface elevation in 3D geological model, locations and identifiers in the national wells database maintained by LEGMC of the geological wells used for model construction**

Considering stratigraphy, lithological composition of deposits as well as traceability in borehole descriptors, geological structure is represented with 6 model layers (TABLE 21). However, territory is described with a limited number of boreholes (FIGURE 61) limiting the degree of spatial interpretation of geological complexity, particularly distribution of sand, silt and clay sediments in making up Amata, Gauja and Burtnieks formation of the Upper and Middle Devonian.

TABLE 21

Model layers and their general description

Stratigraphic unit	Layer name	Average thickness (m)	Lithology
Quaternary	Q	5-10 (up to 100 in area of paleo incision)	moraine diamicton, glacial outwash sediments, peat and freshwater tufa
Upper Devonian Pļaviņas Fm	D3pl	5-25	fissured and karstified dolomites, marls

Stratigraphic unit	Layer name	Average thickness (m)	Lithology
Upper Devonian Amata Fm	D3am#	3-5	loose sandstones, siltstones and clay, often with calcareous cementation
Upper Devonian Amata Fm	D3am	5-30	loose sandstones, siltstones and clay
Middle Devonian Gauja Fm	D2gj	80-105	loose sandstones, siltstones and clay
Middle Devonian Burtnieki Fm	D2br	70-90	loose sandstones, siltstones and clay

Quaternary sediments in the model structure were represented with a unified layer although in borehole logs often two general layers can be divided (sandy sediments on top of moraine). However, scarce borehole data coverage does not allow spatially viable interpretation of any interlayers in Quaternary. Thickness of Quaternary generally varies between 5 to 10 meters increasing up to 100 m along paleo incision. Buried valley itself, as denoted in borehole LV|18641, is believed to be filled with interlayers of sandy and clayey deposits possibly interconnecting the bedrock aquifers. It was assumed that similar lithological layering is traceable throughout the paleo incision.

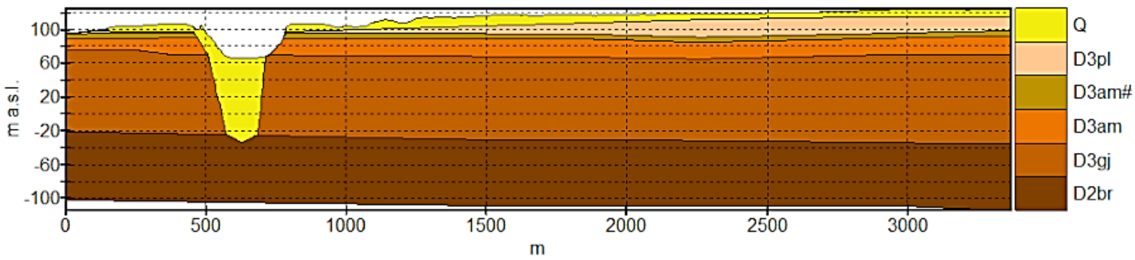


FIGURE 62 **Cross-section through the model area (location of cross-section in FIGURE 61)**

Pļaviņas Fm of fissured dolomites is well traced throughout model area and expressed as one model layer.

Continuous bed of 3 to 5 m thick impermeable marl and clayey deposits in upper part of Amata Fm (D3am# layer) separates corresponding sequences of Pļaviņas Fm and Amata Fm (D_3*am* layer).

Entire succession of Burtnieki, Gauja and Amata Fms is formed of loose sandstones, siltstones with many clayey interlayers. Following stratigraphic descriptors in borehole logs, it is possible to divide this succession in three layers corresponding to Amata (D_3*am* layer), Gauja (D_3*gj* layer) and Burtnieki (D_2*br* layer) Fms, however, heterogeneity of interlayering within this sequence cannot be interpreted with continuous layer system from available borehole data.

Particular attention was paid to the interpretation of paleo incision. Since borehole data alone was too scarce for viable interpretation, information from geological maps in scale of 1:50 000 (Mironovs et al., 1973) was used. At first surfaces of D_2*ar* to D_3*pl* layers were constructed only from borehole data bypassing the eroded parts along paleo incision. Further, by reading elevation values of each layer along distribution extents denoted in geological map, elevation values of the bottom of paleo incision was obtained, which coupled with the data from borehole logs were interpolated to obtain the elevation of Subquaternary surface. In result, continuous distribution of modelled topography of depositional layers were wedged-out at the intersection with Subquaternary surface.

For better representation, web visualization was supplemented with aerial photo (ORTOFOTO 6) and hydrological network of major water flows and water bodies together with spring locations overly over models' topographic surface (FIGURE 63).

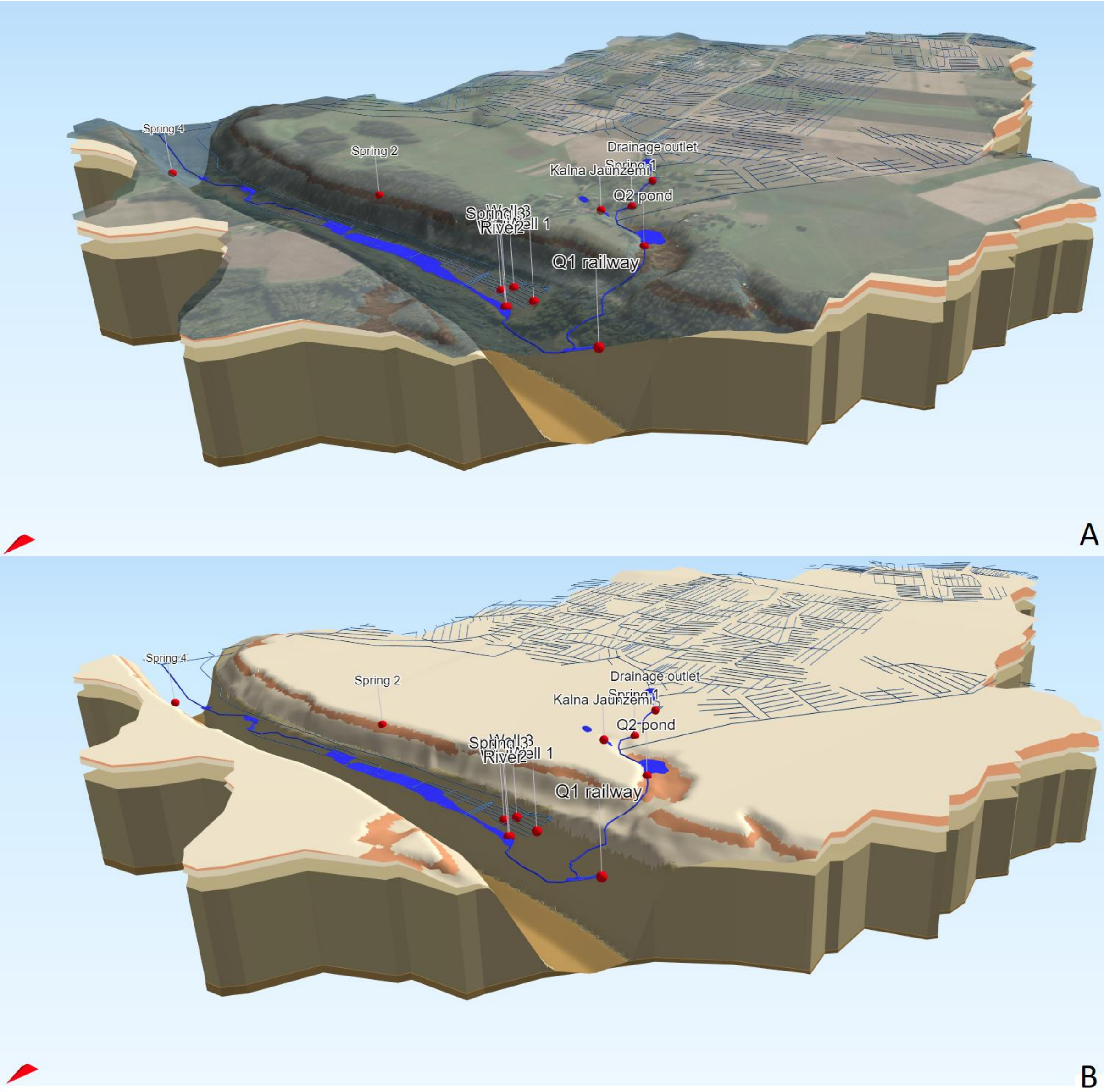


FIGURE 63 **3D geological model of Kazu leja. View from NNE. A - view with Q layer and orthophoto overlay, B - view without Q layer (available at https://www.gprm.lu.lv/kazugrava-3d/).**

5.3. Development of low-cost do it yourself sensors

5.3.1. Introduction

GDTE identification and investigation is a complex process that can involve a wide variety of tools and techniques. Most common methods and tools for hydrological investigations in GDTE are automatic groundwater level measurement, water chemical composition analysis and discharge measurements. However, the cost of the equipment can be high thus limiting the assessment in tight budget conditions. Also, commercially available tools are limited to the features they already have and more specific needs cannot be always satisfied.

Low-cost, do-it-yourself (DIY) are getting more and more attraction in the science community as these sensors can overcome the disadvantages shown by commercial sensors. Firstly, low-cost sensors are of lower cost than their commercial counterpart. Secondly, these sensors usually are flexible and features can be added as necessary. Thirdly, by using a low-cost sensor approach, it is possible to create a tool that doesn't exist yet, but it is needed for specific purposes. GDTE assessment is quite specific in a general available scientific tool viewpoint therefore the last two advantages of low-cost sensors are most important as new tools can be invented for the need.

Low-cost sensors have their disadvantages too, if compared to the commercial sensors. The most important limiting factor of low-cost sensors is the need for specific knowledge – i.e. technical, electronics and programming skills are necessary. The development of low-cost sensor systems can take a lot of time from idea to working unit, but even after great effort involvement the successful and working results are not guaranteed. Despite the cons, it is still worth investing effort to create new low-cost sensors for specific needs as they can unfold new and less expensive opportunities for GDTE assessment.

5.3.2. Low-cost sensor setup

Two main directions of low-cost sensor development and application was decided during GroundEco project:

1) groundwater temperature measurement with high vertical and temporal resolution;
2) differential groundwater level measurements at two different depths.

High resolution temperature profile at GDTE site combined with groundwater level difference between two depths can yield information about groundwater movement characteristics within GDTE. Such data over a longer period of time can shed a light on ecosystem's dependence on groundwater resources and how vulnerable the system is over seasons.

5.3.2.1. Vertical temperature sensor unit

Vertical temperature profile sensor unit was developed for the project for the needs to monitor groundwater temperatures at multiple depths simultaneously. The sensor unit was constructed using low-cost electrical components and sensors. Principal scheme of the sensor unit is pictured at FIGURE 64. The unit is powered by a Li-ion battery, which is charged by a solar panel, thus it can work if enough energy is acquired from the sun.

The unit consists of 5 temperature probes that are installed at different depths on pilot sites. Temperature readings are recorded every 5 minutes on a microSD card and precise date and time is recorded with the help of a real time clock. The brain of the unit is an Arduino-type microcontroller that is programmed in Arduino IDE. The code is developed for the specific unit needs based on open source code provided by "the Cave Pearl Project", E. Mallon (https://thecavepearlproject.org/).

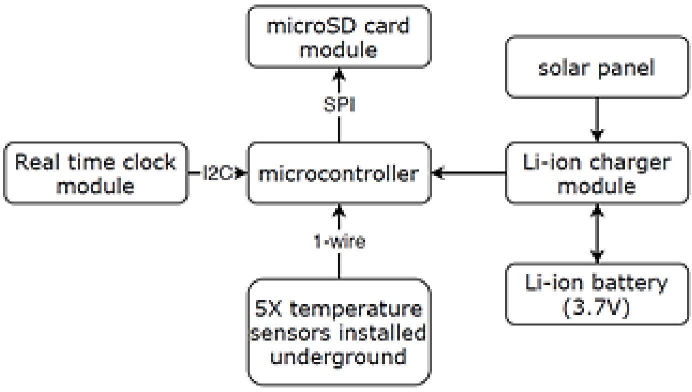


FIGURE 64 **Principal scheme of vertical temperature profile sensor components**

All components were soldered on a printed circuit board (PCB) thus ensuring rigidity when deployed on field. Most electronical parts are kept in a waterproof enclosure (FIGURE 65) with a plug connection for temperature sensors and solar panel.

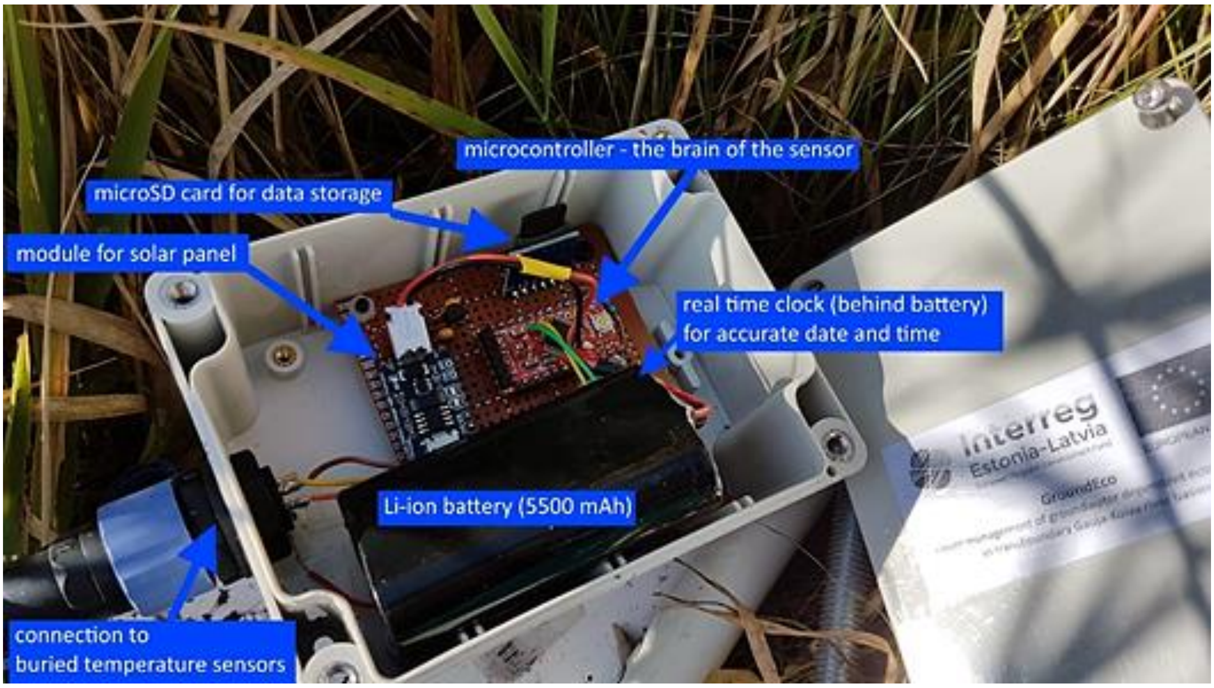


FIGURE 65 **Sensor unit components during installation process in Kazu leja**

Five temperature sensors for each vertical temperature profile unit was installed at different depths to record subsurface temperatures at depths of: 25 cm, 50 cm, 100 cm, 150 cm and 250 cm below ground surface.

All temperature sensors were bath-calibrated prior sensor unit construction to ensure more precise temperature readings than stated by the manufacturer. As a result, accuracy of 0,1–0,2°C was achieved instead of manufacturer's stated accuracy of 0,5°C.

5.3.2.2. Differential pressure sensor unit

Differential pressure sensor unit was envisioned as a tool to record groundwater level differences between two depths at a single place and high resolution. Groundwater movement is driven by hydraulic gradient: if shallow groundwater has a higher level than deeper groundwater, then groundwater flows downwards and vice versa. Furthermore, if precise hydrogeological properties are known, it is possible to calculate groundwater flux. Such information can help to assess the ecosystem's dependency on groundwater.

Initially an automatic differential pressure sensor unit was planned for long term deployment on the field similarly. However, laboratory tests during the prototyping stage revealed shortcomings that prevented this idea. It was found that sensor signal is unstable under changes in temperature and environmental atmospheric pressure as well as gas diffusion in the tubes resulted in significant signal drift over a few minutes. Therefore, an alternative approach was sought.

Eventually a manually operated differential pressure sensor was developed with an LCD screen and buttons to reset the sensor after each measurement. Operator of the sensor unit should reset the unit prior each sampling and manually submerge sensor tubes in two piezometers simultaneously by keeping both tubes in a level. The piezometers should have screen intervals at depths of interest so groundwater level within each piezometer will represent the level at specific depth.

The main part of the sensor unit is a differential pressure transducer/sensor that has two ports to connect flexible tubes. Differential pressure sensor signal is read by a high-resolution analog-to-digital converter (ADC), that, in turn, passes the reading to the microcontroller and eventually the reading is displayed on LCD screen (FIGURE 66, FIGURE 67). Differential pressure sensor units need to be calibrated once per field visit in the laboratory for precise groundwater level measurements.

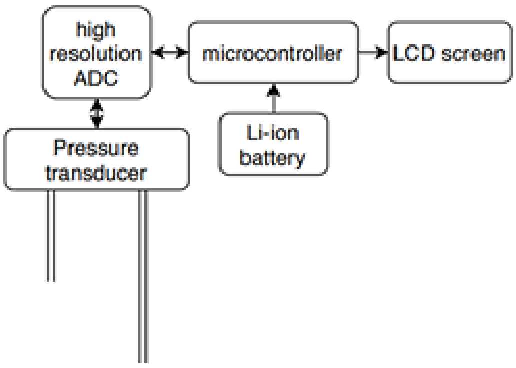


FIGURE 66 **Principal scheme of different pressure sensor components**

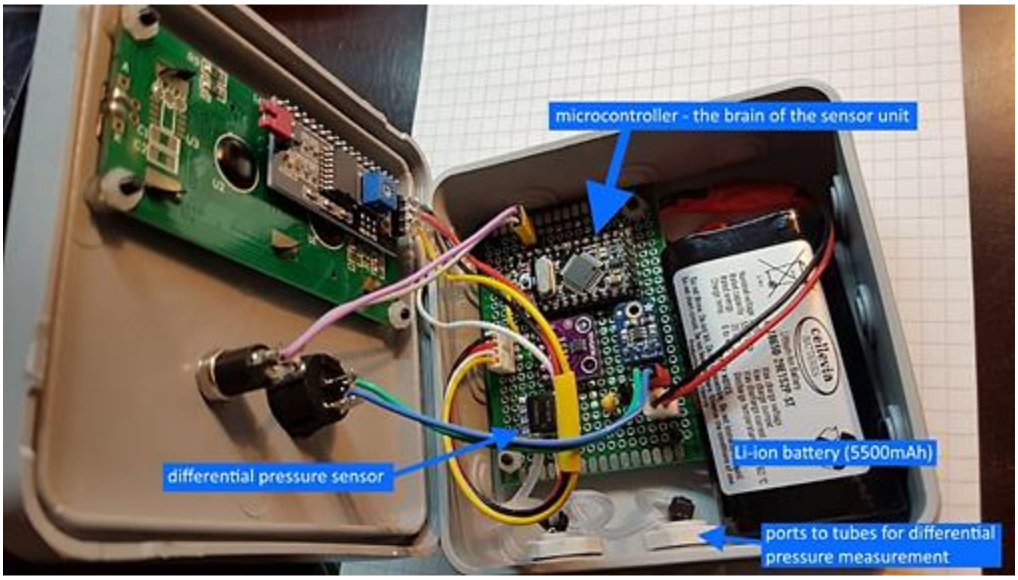


FIGURE 67 **Setup of differential pressure probe**

5.3.3. Results of the low-cost sensor units

Two vertical temperature profile sensor units were constructed and deployed in Kazu Leja and Matsi pilot sites. Kazu Leja site was equipped with the sensor in April 2019, while a second sensor was deployed on Matsi site in November 2019. The first sensor unit deployed in Kazu Leja site showed technical issues during summer 2019 that resulted in loss of data, however, the shortcomings were eliminated and the second improved sensor unit in Matsi site worked as expected. Simultaneous temperature measurements in both sites allowed to compare hydrothermal properties of both sites and reveal the differences that can be critical for GDTE sustainability.

Groundwater temperatures measured at different depths for the same period of time are compared in FIGURE 68, showing distinct temperature fluctuations on both pilot sites. Temperature signals in a compared period of 2 months (FIGURE 68) are covering cold months (December 2019, January 2020) therefore shallow temperature sensors have lower temperatures than deeper ones. The deepest temperature probe located at depth of 2.5 meters at Kazu Leja site shows rapid temperature drop from 8,3 to 6,2°C over the two-month period, while at Matsi site temperatures at the same depth and time period show drop just from 7,38 to 7,25°C. These results indicate that groundwater temperatures at Kazu Leja site has greater impact from surface processes than at Matsi site.

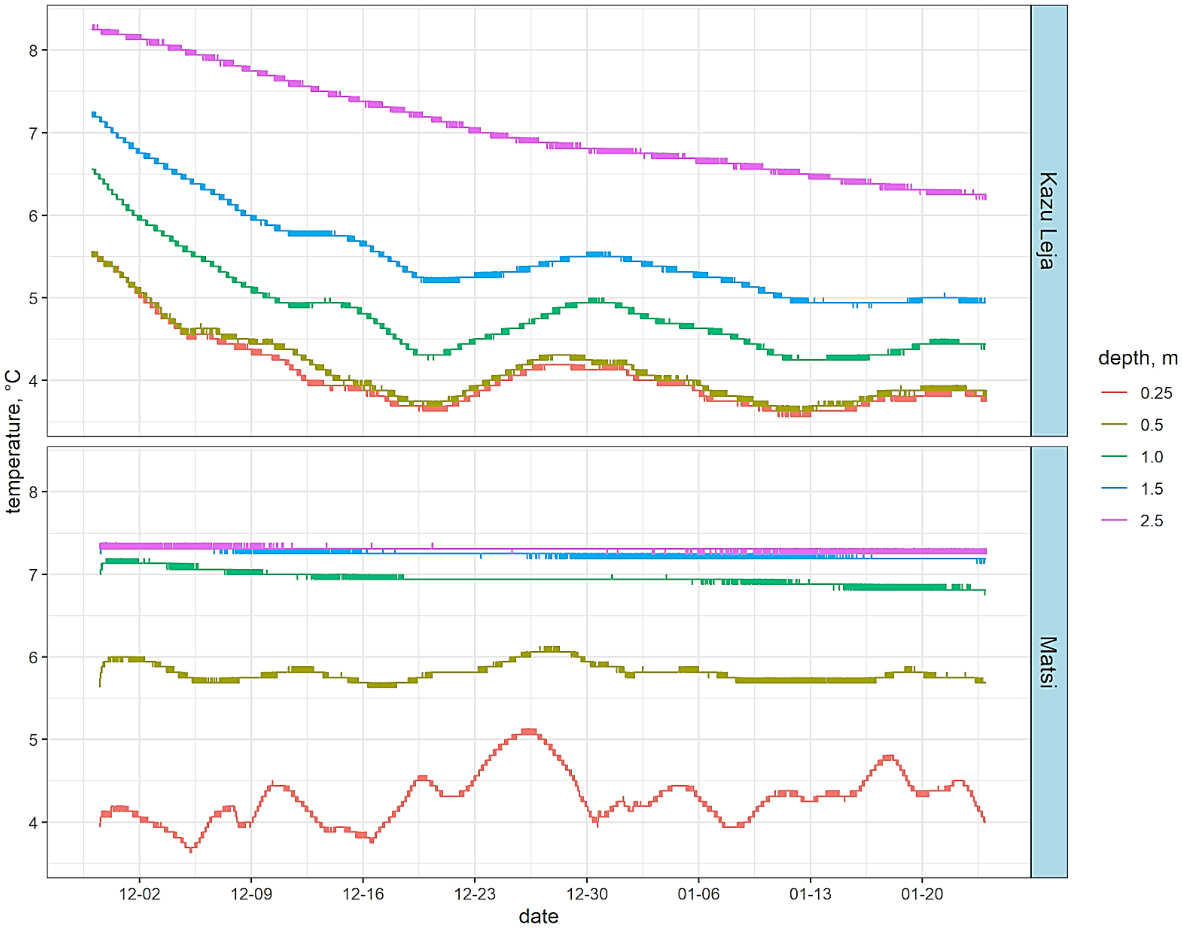


FIGURE 68 **Groundwater temperature time series in both pilot sites at multiple depths**

Similar pattern can be observed, when the same temperature time series are interpolated with depth between sensor probes and plotted in a temperature-colored graph (FIGURE 69). The graph clearly reveals that surface/shallow temperature signals in Kazu Leja site propagate freely up to 1,5 m depth, but are still distinguishable at the deepest sensor in depth of 2,5 m. Matsi site, however, has different characteristics and surface temperature signals travel only short distance, rapidly diminishing at 0,5 m and reaching maximum of 1,0–1,5 m in depth (FIGURE 68).

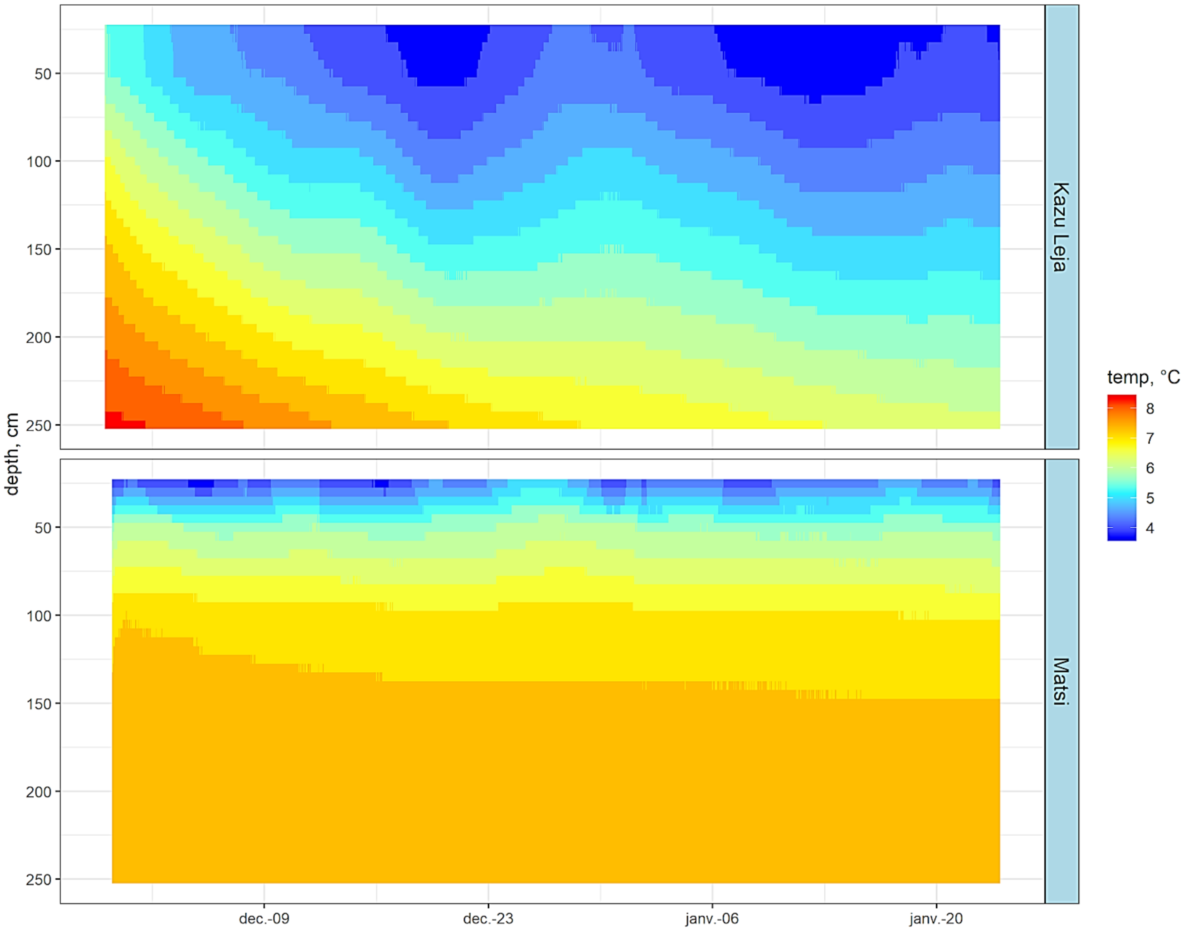


FIGURE 69 **Interpolated groundwater temperature profile for both pilot sites**

Features of groundwater dynamics are revealed when daily aggregated vertical temperature profiles are compared for both sites (FIGURE 70), indicating upward movement of groundwater at Matsi site while Kazu Leja site has no clear indications of vertical groundwater movement. The upward groundwater flow is characterized as a curved temperature profile (line) with depth instead of straight line, thus indicating that deeper located groundwater having higher temperature penetrates upwards. Furthermore, daily temperature profiles at Matsi site aren't scattered a lot, whereas at Kazu Leja site these profiles are much more scattered and have a drift over time due to impact from surface temperatures.

When GDTEs are considered, a low-cost vertical temperature profile sensor unit in a simplistic manner can indicate that deeper located groundwater has greater impact on Matsi ecosystems and smaller impact on Kazu Leja site. However, these results can be attributed to local points where the sensor units are installed and cannot account for the whole pilot site. Moreover, significant lateral groundwater flow can distort the interpretation as the temperature signals may come with groundwater from adjacent territories. Nevertheless, such low-cost sensor units proved to be useful to assess the ecosystem's dependency on groundwater and might reveal anthropogenically induced negative changes if such groundwater temperature monitoring is performed in a long term.

Differential pressure sensor unit was operated manually as it wasn't possible to construct it in a way that it could take accurate readings automatically. As a result, measurements were made solely during field visits and acquired data sets are scarce and rare. However, acquired data allowed to estimate the accuracy of the sensor unit. Taking into account all possible errors, the accuracy of the sensor unit is in range from 0.3 to 0.5 mm, if groundwater level difference between two piezometers is measured. Acquired field data suggests that differential groundwater level between a piezometer at 220 cm depth and a piezometer at 110 cm depth in Kazu Leja site can change as much as 49.4mm suggesting dynamic groundwater system with changes in groundwater flow over time. The vast amplitude of measurement results indicate that significant changes occur between the field visits that cannot be taken into account by using this sensor unit. The developed sensor unit could be applicable for sites where groundwater system is more stable over time, but not in the Kazu leja site. However, it is not possible to predict a system's behavior before actual measurements are taken, therefore even the developed sensor unit can be of help if less is known about the GDTE site of interest.

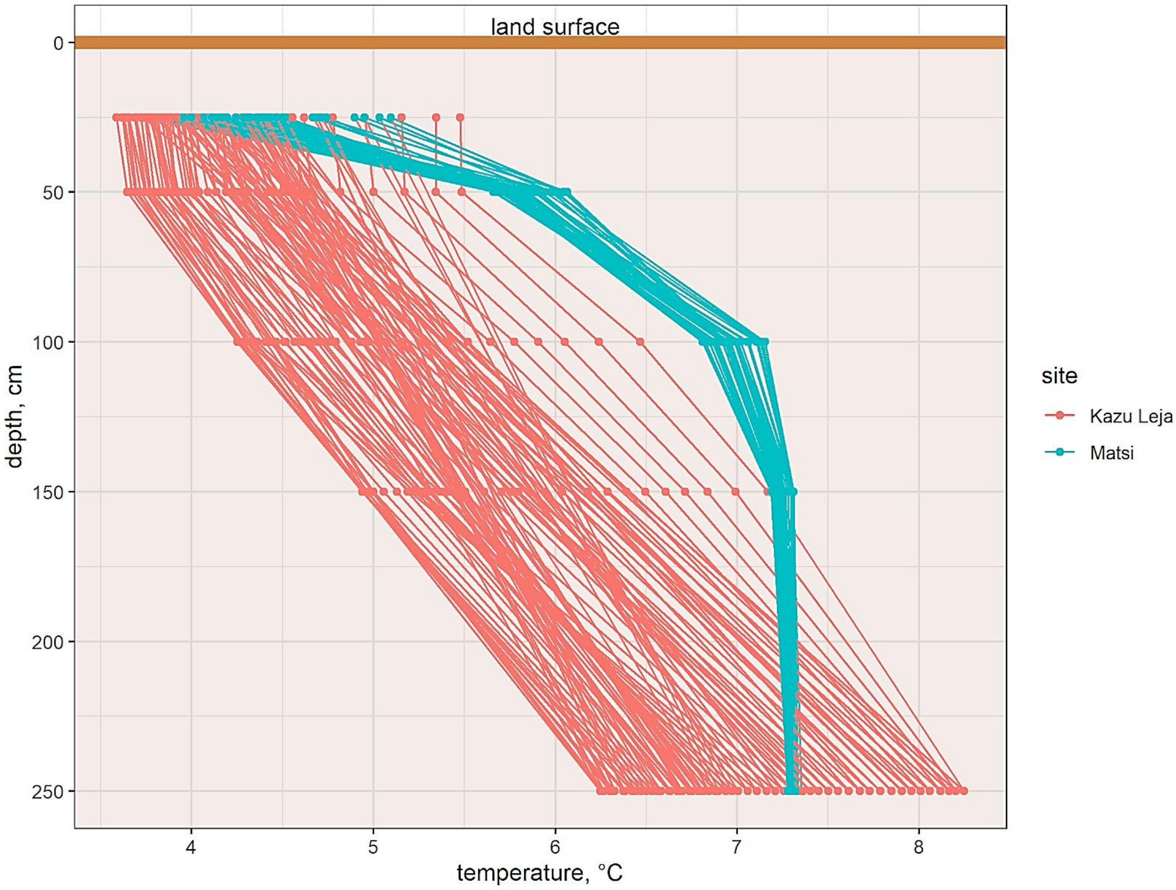


FIGURE 70 **Daily vertical temperature profiles for both pilot sites**

6. Conclusions and recommendations

6.1. GDTE identification, monitoring and assessment

1. The basic information for GDTE identification is and will be obtained from habitat maps, i.e. data acquired during surveys of the habitats listed in the EU Habitats Directive, and maps and data layers are available in state information systems of Latvia and Estonia. However, the habitat layers are constantly changing and will change in the future due to new inventories. Consequently, **identification of GDTEs is an interactive process and GDTE maps should be periodically re-evaluated** based on improved or new data. In Estonia, national scale inventory of habitats is expected to repeat within ~15 years. In Latvia, national scale inventory is currently ongoing (to be completed in mid-2021) and re-mapping could take place within 10 or more years, still there is no precise timetable. Re-mapping is costly and time consuming (usually several years), thus changes in habitat quality should be detected by representative and targeted monitoring programmes.
2. Regular monitoring of GDTE habitats would provide necessary data for reporting and directly benefit the implementation of both the Water Framework Directive (WFD) and Habitats Directive.
3. Current condition of protected habitats (including GDTE habitats) is recorded during the first survey but should be re-evaluated at least once in 6 years (in accordance with reporting periods of EU Habitats Directive). **For GDTE status assessment, monitoring (inventory) once in 6 years period is acceptable** as changes in such ecosystems are not rapid and monitoring is costly.
4. Estimation of the habitat status (quality) strongly depends on the expert judgement and determination of the habitat quality in a particular habitat patch slightly differs in Estonia and Latvia. Even though there are certain indicators that clearly show the habitat condition, still the expert must have sufficient knowledge and experience. Thus, **capacity building of experts working with GDTE assessment** and harmonization of knowledge base and approaches between neighboring countries **must be encouraged.**
5. GroundEco project focused only on the Gauja-Koiva river basin, not on the entire countries' area. Therefore, selected minimum threshold areas (1 ha or 10/20 ha for Latvia/Estonia) might be revised when significant GDTEs will be identified at country scale. The thresholds for minimum areas might be larger at national scale to ensure balance between desirable monitoring intensity and cost-efficiency. Monitoring of every GDTE patch is not possible and not necessary, and **deterioration of the status of GWB can be observed by monitoring the number of selected representative sites.**
6. In Estonia, the habitat monitoring system (also covering GDTE habitats) is established, but currently is re-evaluated and will change. In Latvia, the previously used habitat monitoring methodology and established transects (randomly chosen throughout the country, proportionally covering all habitat types) must be re-evaluated after completing national scale mapping (Nature Census project, to be completed in mid-2021), thus monitoring coverage in GDTE habitats is unknown.
7. **Neither in Estonia, nor in Latvia the state (or any other) monitoring programmes regarding groundwater and GDTEs are complete.** Currently the locations of groundwater monitoring networks and GDTEs do not overlap. Established state groundwater monitoring networks in Estonia and Latvia were designed to serve different purposes (e.g. to detect regional trends in groundwater quality or quantity, assess impacts of diffuse pollution) and provide valuable long-term datasets. The locations of wells cannot be moved, and current monitoring frequency must not be interrupted. However, an effort should be made to improve groundwater monitoring networks by **establishment of new wells and selection of new springs for national monitoring needs considering locations of GDTEs.** A conceptual hydrogeological model should be developed for each monitoring station

serving as a tool for planning future monitoring programs, evaluating the results, accumulation of knowledge and institutional memory.

8. **Parameters to be monitored by national groundwater monitoring networks should be related to the type of (potential) pressure.** Targeted monitoring is cost-effective (both money and time) and provides relevant, site specific data for future assessment needs (analysis). Wise planning of monitoring is necessary to consider knowledge on the current situation and planned land/ resource use intentions (potential impacts).
9. The balance between costs of three major parts in the water monitoring programme (sampling, laboratory analysis and data evaluation) should be improved. This should be considered when developing water monitoring programmes based on available nation funding. If one of the three parts is inadequately funded, then investments in two other parts are wasted. For example, often samples are collected and analyzed in the laboratory, but results are not properly evaluated on a regular basis. Monitoring only for the purpose of monitoring is not useful: a conceptual hydrogeological model should be developed for each groundwater monitoring station, defining the purposes of the monitoring, justifying parameters to be monitored.
10. **Currently, there are no thresholds for GDTEs quality assessment** (nitrates, phosphates or other). As threshold values can be site-specific and habitat type-specific, they must be based on several studies and comparison of similar sites. Each type of monitored GDTE should have its site/type specific quality parameters in addition to basic/major ion composition. A reference site(s) for each significant GDTE type should be selected, with minimal anthropogenic impact, so that it can be used to evaluate the state of other sites impacted by human activities. **Estimation of thresholds will not be possible based on data provided by national monitoring only; the data and knowledge should come from scientific studies.**
11. **Conceptual model (understanding) for any new monitoring site (groundwater or GDTE) should be developed as the first step in planning of the monitoring programme.** Conceptual model is a good tool to design GDTE monitoring programmes: identify inflows, outflows and important elements in the system, planning of monitoring points accordingly (e.g. retention of (harmful) trace metals in GDTE can be an ecosystem service. Retained trace metals can be released into the environment if GDTE is degraded, e.g. due to decreased groundwater supply, therefore, downstream-outflow water quality should be monitored as well). Groundwater monitoring, weather stations and surface water monitoring stations should be at the same or close-by locations, so the preference is given to sites where these conditions can be met.
12. Detailed monitoring of water quality should be continued for at least a few (preferably 3–4) years to consider a site as a good GDTE reference site. Then the sampling frequency and number of parameters can be decreased based on locally identified pressures.
13. GDTEs mostly depend on small, local groundwater systems with no real transboundary character, still **transboundary groundwater monitoring between Estonia and Latvia is encouraged to harmonize monitoring approaches and improve the usage potential of shared national groundwater datasets and encourage transboundary cooperation.** New groundwater monitoring station could be installed at the Latvian-Estonian border for transboundary groundwater monitoring and laboratory intercalibration needs. In addition, laboratory standards can be shared, or independent experts/certified laboratories attracted to evaluate the performance of national laboratories. **Intercalibration of ecological/habitat monitoring is suggested as well.**
14. **Major compounds** for ionic charge balance calculation (typical data quality assessment) **should be measured in all water samples.** If any problems are identified it will allow us to understand the ongoing hydro-geo-chemical processes.

15. Collection of water samples is time consuming and expensive, thus the number of measured parameters could be increased providing extra useful information at nearly the same costs. For example, ion chromatography can determine multiple parameters on a single sample run without extra expenses, but often only a short list of parameters is reported and stored in national databases according to monitoring programmes. The communication between experts working on development of monitoring programmes, further data analysis and laboratories performing the sample analysis should be improved.
16. **Low cost sensor units can help to understand the ecosystem's dependency on groundwater.** Vertical temperature profile sensors can reveal vertical groundwater flow. It is suggested to install the sensors at the following depths: 0.25 m, 0.5 m, 1.0 m, 1.5 m and 2.5 m. But the depths can vary and should depend on the knowledge of local conditions. Springs should be monitored by single temperature sensors to provide knowledge about seasonality and residence time of spring water.

6.2. Recommendations for GDTE assessment according to the quantitative and qualitative assessment schemes application

The first step before application of assessment schemes is the determination of GDTE dependence on a specific GWB. If the significant GDTEs have been determined according to the methodology described in section 4.3.1, then the GWB on which GDTE depends may be identified solely based on spatial overlay in most of Estonia and Latvia. Respectively, GWB closest to the surface below the GDTE is the one that supports the ecosystem, and the one whose status assessment should consider that GDTE.

However, the determination of the contributing GWB is not always straightforward. In cases where separate Quaternary GWBs have been delineated, it must be verified whether groundwater that supports the GDTE is from Quaternary or from the topmost bedrock (artesian) GWB, or both. Only then GDTE can be joined with the correct GWB for further GWB status assessment. Such cases are rather rare in Estonia and Latvia. Currently, there are four such GWBs in Estonia and two in Latvia. There are no significant GDTEs on the two of the four Estonian Quaternary GWBs (Prangli and Meltsiveski), therefore, specific differentiation in Estonia is necessary only for significant GDTEs situated on Männiku-Pelguranna and Vasavere GWBs. In Latvia there are 2 Quaternary GWBs. There are no significant GDTEs in one of them, still it is only partly located in the project area (Gauja river basin), thus the part located in the Daugava river basin was not assessed and it is unknown if significant GDTEs are present. And the second Quaternary GWB is located outside the project area in Daugava's river basin, thus it is also unknown if there are significant GDTEs.

In these cases, the determination of the representative GWB should be based on conceptual understanding of the site (hydrogeological characteristics) and most likely will involve expert judgment, which ideally can be supported by water chemistry and water level data. E.g. Vertical temperature profile sensors can reveal vertical groundwater flow, but single temperature sensors in springs can indicate seasonality and residence time of spring water.

If there is no relevant information to develop conceptual understanding, initially it can be assumed that significant GDTE depends on both GWBs, Quaternary and the topmost bedrock GWB. However, if significant GDTE is in unfavorable status and assessment schemes show that damage might be caused by groundwater, a detailed investigation must be carried out (see investigation processes of pilot sites in section 5.1 un 5.2).

The proposed quantity and quality assessment schemes (see sections 4.4) are more flexible than initially proposed schemes by European Commission (2009). The reasons for that are strict reporting deadlines (RBMPs of WFD) and often missing long-term (or any) data on water levels and chemistry in GDTE sites. On the one hand, assignment of confidence levels allows us to fulfil basic reporting needs in cases of missing data, but on the other hand, highlights problem areas where monitoring networks must be established. It is also important to remember that GDTEs can be damaged by a variety of pressures which are often not related to groundwater, thus other potential causes must be ruled out first prior costly investments into monitoring networks and water sampling. Therefore, in the first steps of assessment schemes, it is recommended to consider indirect data, such as analysis of any evidence (historical maps, pictures, experience of locals) that could indicate negative pressures (observed as GDTE already in bad status or shows a potential risk) on GDTE coming from a GWB.

Currently temperature, pH and specific conductance of mire/ spring water are not a part of habitat status assessment but could provide first insight in water quality and functioning of the site, and therefore is recommended. Although this requires equipment and training for the habitat experts who will carry out the field surveys, the costs are rather low (such equipment is easy to use and do not require large maintenance costs).

Significant groundwater abstraction (also dense abstraction network) near GDTE can lower the groundwater levels and decrease groundwater inflow to GDTE, thus deteriorating its status. Direct indicator of negative groundwater abstraction is actual evidence of damage in GDTE and observed changes in groundwater levels compared to annual mean levels (such data are often missing). Commonly used groundwater quantitative management approaches in the GDTE sites are volumetric allocations, buffer/well exclusion zones, groundwater level triggers and groundwater rate of declined triggers. In the assessment scheme of quantitative effect, we recommend to use a simple volume-based assessment approach which allows us to identify cases when groundwater abstraction can potentially have a negative impact on GDTE. This is a rather simple approach which can be applied for general groundwater management needs, but it does not provide results with high confidence. In-depth investigation is needed to gather more site-specific data, and more sophisticated measures should be elaborated on a case-by-case basis in line with existing regulations.

If there is no relevant GW abstraction in the vicinity, also other human activities may lower the groundwater level and have a negative impact on the GDTE. Deep ditches reaching the mineral sediments below the peat and mines could be the reasons. Appropriate investigation (using maps, databases, *in-situ* observations) of the presence of such pressures should be carried out. Also, beaver activities can influence hydrological and hydrogeological regimes (evidence observed in both project pilot areas) and should be considered when analyzing groundwater level data series.

As groundwater quality threshold values for GDTEs are missing and their establishment is a costly and long-term process, the usage of indirect data (such as fertilization amounts, locations of polluted sites and land cover data) is strongly encouraged. Consequently, the cooperation intensity between institutions and sharing of data from various national databases should be improved.

If the status of significant GDTE is worse than good and there is supporting evidence (from direct and/ or indirect data) that groundwater could be responsible for that, quantity and quality water monitoring should be established in the watershed area of GDTE to carry out in-depth investigation. Detailed description of approaches used and lessons learned are described in sections 5.1 and 5.2 of project pilot sites.

Development of monitoring networks should be based on conceptual understanding of the site and identified pressures. Consequently, the catchment area (watershed) of the GDTE should be identified and studied using direct and indirect data. Monitoring of the quantitative dependency of GDTEs should be based on long-term observations of surface and groundwater levels. Levels should be monitored at least two times a day using automatic loggers. First, water quality monitoring should be carried out seasonally (four times a year). The parameters to be monitored (water level, nutrients) must be GDTE specific, depending on the pressures determined during development of conceptual understanding (e.g. nutrients for agricultural pressures). After the first screening of water quality, the list and frequency of measured parameters must be re-evaluated. If there is evidence that GDTE is only supported by groundwater not fed by GWB, then quality monitoring is not necessary. If the analysis shows that damage may occur through excess nutrient inflow, then just water quality monitoring must be continued. For the GDTEs to which no potential pressures through anthropogenically induced changes apply, establishing a monitoring network is not necessary.

6.3. Future prospects

Next steps to be taken in both countries:

1. Identification of significant GDTEs in the whole country using the methodology developed in the GroundEco project.
2. Evaluation whether the number of significant GDTEs that have been identified is manageable. If the number is high, then the minimum areal thresholds for considering a GDTE significant, have to be raised and the country-wide evaluation has to be performed again. As a result, a manageable number of significant GDTEs that will be considered in the groundwater status assessment, will be derived.
3. Analysis of potential pressures expressed through groundwater on each significant GDTE. Based on existing data on groundwater abstraction, mine dewatering, land usage, etc., the theoretical possibility, whether anthropogenically induced changes in groundwater may damage the GDTE, have to be evaluated. If it is determined that there can be no damage neither through groundwater quantity or quality, then there is no need to establish a specific monitoring network for the GDTE. Similar assessments have to be made in the future, if new groundwater abstraction facilities, mines, etc. are planned in the vicinity of a significant GDTE.
4. Developing a monitoring scheme and establishing a monitoring network for significant GDTEs that are at risk of being damaged through anthropogenically induced changes in groundwater quantity and/ or quality. The parameters to be monitored (water level, nutrients) have to be GDTE-specific, depending on the actual pressures.

The first two of these steps may be performed as a single study and the last two as another study. In order to guarantee a fluent and reliable assessment of the status of groundwater bodies according to the test of groundwater dependent terrestrial ecosystems, some other requirements have to be fulfilled in both countries:

- The status of all significant GDTEs has to be assessed at least once every six years according to the Annex I habitat status assessment procedures. That ensures there is up-to-date information on the status of all GDTEs, even if no specific monitoring network is established according to the assessment of potential pressures. The availability of this data would enable passing step 1 in the assessment schemes of quantitative and qualitative effect.
- Threshold values for nitrogen and phosphorus have to be developed for all groundwater dependent habitat types. These thresholds would enable passing step 3 in the assessment scheme of qualitative effect. Currently, even if nutrient levels were monitored in GDTEs, it is impossible to say if a certain concentration of nitrogen or phosphorus is too high or acceptable for that GDTE.

Reference monitoring sites for all groundwater dependent habitat types have to be established in areas where there is no human impact through groundwater, or the impact is as small as possible. These sites would provide background information for questionable cases where it is not clear whether the damage to the GDTE has been caused by anthropogenic or natural processes.

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ANNEXES

Annex 1

Characteristics of groundwater bodies delineated in Latvia (in Gauja river basin)

Groundwater body, associated river basin district		**Area**	
Q1 Daugava River Basin District – 195 km² (60%) Gauja River Basin District – 129 km² (40%)		Total area: 324 km² *Gauja-Koiva river basin: 96 km² (30%)*	
Physico-geographical characteristics	Groundwater body area is covered by plains. In western and central parts there is the Seaside lowland with Rīgava plain, and in eastern part - Midland Lowland with Ropaži plain. ***Absolute height***: 0.0-29.1 m above sea level ***Average annual rainfall***: 650-700 mm ***Average air temperature***: summer - +17.0°C, winter - -5.0°C		
Characterization of aquifers	**Type of aquifers, dominant lithology**	***Dominant material***: sand ***Aquitard***: aleirite ***Dominate***: porous material	
	The main characteristics of aquifers	***Water conductivity coefficient (km)***: 26-3004 m²/d (mostly 1500-2000 m²/d) • ***Baltic Ice Lake*** (lgQ_3ltv^b) aquifer: 564-3004 m²/d (mostly 2000 m²/d) in area of groundwater well fields Baltezers and Zaķumuiža • ***Glaciolimnic*** (lgQ_3ltv) aquifer: 2080 m²/d in area of groundwater well field Baltezers • ***Glaciolimnic and Baltic Ice Lake*** ($lgQ_3ltv – lgQ_3ltv^b$) aquifers: 260-1428 m²/d in area of groundwater well field Baltezers • ***Baltic Ice Lake and Holocene*** ($lgQ_3ltv^b – Q_4$) aquifers: 26-2170 m²/d (highest values in area of groundwater well field Baltezers: 1070-2170 m²/d, mostly up to 1500 m²/d; in direction of Gulf of Riga: 26-170 m²/d • ***Glaciolimnic and Holocene*** ($lgQ_3ltv – Q_4$) aquifers: 182-2490 m²/d (mostly 1500 m²/d) in area of groundwater well fields Baltezers un Zaķumuiža	
	Thickness of aquifers	***Quaternary thickness***: 26-97 m (average – 49 m)	
Vulnerability		***High risk of contamination***: 92% of the area ***Medium risk of contamination***: 8% of the area	
Land use from CORINE Land Cover 2018		***Common types***: Coniferous forests – 40% Discontinuous urban fabrics – 10% Water bodies – 10%	Pastures – 7% Transitional woodland-shrubs – 6% Complex cultivation patterns – 5% Other – 22%
Nitrate vulnerable zone		***Distribution***: 79% of the area (central and north-eastern part)	
Groundwater recharge	**Main recharge mechanisms**	***Dominate***: rechange by precipitation ***Infiltration***: 117 000 m³/d	
	Average annual precipitation	***Average annual precipitation***: 610 mm/m² (Riga-University station)	
	Recharge and discharge areas	***Recharge area***: all of the area ***Discharge area***: Gulf of Riga, Rīgava coastal plain	
Monitoring	**Number of monitoring stations, number of wells**	***Quantity monitoring***: 5 stations: Carnikava (1 well), Jugla (1 well), Kalngale (2 wells), Piukas (1 well) and Rīga (5 wells). Total: 10 wells. ***Quality monitoring***: 4 stations: Carnikava (1 well), Jugla (1 well), Kalngale (2 wells) and Piukas (1 well). Total: 5 wells. • ***Operational monitoring***: monitoring station Jugla (1 well) • ***Surveillance monitoring***: correspond to quality monitoring *Gauja-Koiva river basin*: ***Quantity monitoring***: 2 stations: Carnikava (1 well) and Piukas (1 well). Total: 2 wells. ***Quality monitoring***: 2 stations: Carnikava (1 well) and Piukas (1 well). Total: 2 wells. • Surveillance monitoring: correspond to quality monitoring	

	Types and frequency of observations	***Quantity monitoring indicators***: groundwater level from ground surface (m) ***Quality monitoring indicators***: physico-chemical parameters (all stations), main ions (all stations), heavy metals (all stations), pesticides (stations Carnikava, Kalngale and Piukas) and pesticide active substances used in Latvia (stations Carnikava and Piukas). ***Frequency***: from 1 to 4 times a year, varying from 1 time in 4 years to every year. • ***Operational monitoring indicators***: physico-chemical parameters (both stations), main ions (both stations) and heavy metals (both stations) • ***Surveillance monitoring indicators***: correspond to quality monitoring **Gauja-Koiva river basin**: ***Quantity monitoring indicators***: groundwater level from ground surface (m) ***Quality monitoring indicators***: physico-chemical parameters (both stations), main ions (both stations), heavy metals (both stations), pesticides (both stations) and pesticide active substances used in Latvia (both stations). ***Frequency***: from 1 to 4 times a year, varying from 1 time in 4 years to every year. • Surveillance monitoring indicators: correspond to quality monitoring
Groundwater resources	**Groundwater well fields**	6 groundwater well fields **Gauja-Koiva river basin**: 3 groundwater well fields
	Groundwater abstraction in well fields	23 892.65 m^3/d **Gauja-Koiva river basin**: 13 649.46 m^3/d
	Calculated resources in well fields	70 000 m^3/d **Gauja-Koiva river basin**: 45 100 m^3/d
	Recharge amount	***Dominate***: Groundwater downstream flow ***Recharge***: 160 000 m^3/d ***Groundwater balance***: 10 000 m^3/d

Groundwater body, associated river basin district			Area
D6 Gauja River Basin District – 4375 km² (89%) Daugava River Basin District – 516 km² (11%)			Total area: 4891 km² Gauja-Koiva river basin: 4375 km² (89%)
Physico-geographical characteristics	Groundwater body is characterized by variable relief - in the western part, the plain stretches, the middle part and the eastern part are formed by medium-high and high hills, while the rest of the area consists of wavy plains. In the western part there is the Midland lowland, which includes the Midland lowering and the Ropaži plain, in the central part - the Vidzeme highland, which includes Mežole mound, Piebalga mound and Augšgauja lowering. The eastern part is occupied by the Tālava lowland, which includes Trapene plain and Alūksne highland with Veclaicene and Maliena hillsides, Vaidava downfall and Gulbene hillocks. In the north-south direction, the terrain changes from a less-scattered area around Smiltene to a severely scattered area around Vecpiebalga, but in the west-eastern direction it changes from a flat area around Ropaži to a severely scattered around Dzērbene and Taurene neighborhood, from a flat around Lejasciems to scattered around Veclaicene. ***Absolute height***: 90-265 m above sea level ***Average annual rainfall***: 600-850 mm (Vidzeme highland - over 850 mm) ***Average air temperature***: summer – from +16.5°C to +17.0°C, winter – from -7.0°C to -6.0°C		
Characterization of aquifers	**Type of aquifers, dominant lithology**	***Dominant material:*** sandstone and dolomite ***Aquitard:*** dolomite, aleirolite and clay ***Dominate***: porous material ***Quaternary material***: moraine loam, sand and clay	
	The main characteristics of aquifers	***Water conductivity coefficient (km)***: 26-3580 m²/d (mostly 700 m²/d) • Quaternary (Q) aquifer: 11-113 m²/d (mostly 30 m²/d) • Katleši-Ogre (D_3*kt-og*) aquifer: 35 m²/d • Daugava (D_3*dg*) aquifer: 177-383 m²/d • Pļaviņas (D_3*pl*) aquifer: 26-571 m²/d (can reach 2107-3580 m²/d) • Salaspils (D_3*slp*) aquifer: 38-1224 m²/d • Pļaviņas-Salaspils (D_3*pl+slp*) aquifer: 190 m²/d	
	Thickness of aquifers	***Bedrock thickness***: 0.1-105 m (average - 30 m) ***Quaternary sediment thickness***: 1-20 m (Ropaži Plain), 5-25 m (Trapene Plain), 75-135 m (Mežole mound) (average - 50-60 m)	
Vulnerability		***Low risk of contamination***: 4% of the area (eastern part of Trapene plain) ***Medium risk of contamination***: 84% of the area ***High risk of contamination***: 12% of the area (western part of Mežole mound and Ropaži plain) ***Land use in high-risk area***: non-irrigated arable land, pastures, complex cultivation patterns and lands principally occupied by agriculture, with significant areas of natural vegetation (less in proportion – discontinuous urban fabrics, mineral extraction sites and industrial or commercial units)	
Land use from CORINE Land Cover 2012		***Common types***: Mixed forests – 26% Coniferous forests – 17% Transitional woodland-shrubs – 16%	Pastures – 12% Non-irrigated arable lands – 11% Other – 12%
Nitrate vulnerable zone		***Distribution***: 13% of the area (western part)	
Groundwater recharge	**Main recharge mechanisms**	***Dominate***: rechange by precipitation ***Infiltration***: 1 792 000 m³/d	
	Average annual precipitation	***Average annual precipitation***: 786 mm/m² (Alūksne, Gulbene, Zosēni, Sigulda stations)	
	Recharge and discharge areas	***Recharge area***: central part of Vidzeme highland, and eastern part of Alūksne highland ***Discharge area:*** western part of Ropaži plain and eastern part of Trapene plain	

Monitoring	**Number of monitoring stations, number of wells**	***Quantity monitoring***: 3 stations: Dzērbene (1 well), Velēna (2 wells) and Virāne (3 wells). Total: 6 wells ***Quality monitoring***: 2 stations: Velēna (2 wells) and Virāne (3 wells). Total: 5 wells. 6 springs: Bānūžu spring, Dāvida dzirnavu spring, Mežmuižas spring, Saltavots spring, Vecstrautu spring and Zīļu spring. • ***Operational monitoring:*** correspond to quality monitoring Gauja-Koiva river basin: as listed above
	Types and frequency of observations	***Quantity monitoring indicators***: groundwater level from ground surface (m) ***Quality monitoring indicators***: physico-chemical indicators (both stations and all springs), main ions (both stations and all springs), heavy metals (both stations and all springs), pesticides (both stations and all springs) and pesticide active substances used in Latvia (both stations and all springs) ***Frequency***: from 2 and 4 times a year, varying from 1 time in 4 years to 2 times in 4 years • ***Operational monitoring***: correspond with quality monitoring Gauja-Koiva river basin: as listed above
Groundwater resources	**Groundwater well fields**	5 groundwater well fields Gauja-Koiva river basin: as listed above
	Groundwater abstraction in well fields	1090.61 m³/d Gauja-Koiva river basin: as listed above
	Calculated resources in well fields	3884 m³/d Gauja-Koiva river basin: as listed above
	Recharge amount	***Dominate***: Groundwater downstream flow ***Recharge***: 1 792 000 m³/d ***Groundwater balance***: 3 000 m³/d

Groundwater body, associated river basin district		Area
A8 Gauja River Basin District – 8 866 km² (32%) Daugava River Basin District – 18 483 km² (68%)		Total area: 27 349 km² *Gauja-Koiva river basin: 7927 km² (29%)*

Physico-geographical characteristics	Groundwater body terrain is diverse because of its wide area. Western is covered by plains and wavy plains, in north and east, plains and wavy plains alternate with small hills, medium high and high hillsides, while in central and south-eastern part basically consists of small hills, as well as medium and high hills. In western part there is Midland Lowland, which includes Tīreļi, Ropaži, Taurkalne and Metsepole plains, Upmale hilltop, Lejasdaugava valley, lower part of Central Latvia, and the Idumeja highland, which include Limbaži wavy plain, Augstroze hillfort and Gauja River ancient valley. In northern part of territory there is Tālava lowland, which includes Trikāta elevation, Trapene, Burtnieks and Seda plains, as well as the Alūksne highland, which includes Veclaicene and Maliena lowlands, Gulbene hillock and Vaidava downfall. In central part of territory there is Vidzeme highland, which includes Mežole, Piebalga and Vestiena hillocks, Aumeisteri hillside, as well as Augšogre and Augšgauja lowlands. To east lies East Latvian lowland, which includes Jersika and Lubāna plains, Arona hills and Adzele elevation. In eastern part is Mudava lowland, which include Abrene descent and Zilupe plain, and in south-east – Latgale highland, which includes Burzava, Rāznava, Feimaņi and Dagda hills, as well as Malta and Rēzekne declines. Terrain in north-south direction changes from severely sectioned at Stalbe neighborhood to mildly sectioned at Līgatne neighborhood and plain at Jumprava neighborhood, as well as from the easily-sectioned at Valka neighborhood to plain area at Bilska neighborhood and from sectioned at Smiltene neighborhood to the heavily sectioned at Vestiena neighborhood. Western-eastern area changes from flat at Carnikava neighborhood to sectioned at Skujene neighborhood, from heavily sectioned at Jaunpiebalga neighborhood to sectioned at Galgauska neighborhood to plain at Viļaka neighborhood. ***Absolute height***: 0.0-311.5 m above sea level ***Average annual rainfall***: 600-850 mm (central part – over 850 mm) ***Average air temperature***: summer – from +16.5°C to +17.0°C, winter – from -5.0°C to -7.0°C	
Characterization of aquifers	**Type of aquifers, dominant lithology**	***Dominant material:*** sandstone ***Aquitard:*** aleirolite and clay ***Dominate***: porous material
	The main characteristics of aquifers	***Water conductivity coefficient (km)***: 23-1100 m²/d (mostly 800 m²/d) • Quaternary (Q) aquifer: 150-869 m²/d • Amata (D_3am) aquifer: 23-536 m²/d (generally below 200 m²/d) • Gauja (D_3gj) aquifer: 130-1046 m²/d (mostly not exceeding 800 m²/d) • Gauja-Amata (D_3gj+am) aquifer: 215-606 m²/d • Arukila (D_2ar) aquifer: 34-910 m²/d (mostly 400 m²/d) • Burtnieki (D_2br) aquifer: 36-835 m²/d • Arukila-Burtnieki (D_2ar+br) aquifer: 125-742 m²/d
	Thickness of aquifers	***Bedrock thickness***: 115-302 m (average - 210 m)
Covering sediments	**Lithology**	***Quaternary material***: moraine loam, sand, gravel and clay ***Overlaying aquifer complex***: Pļaviņas-Amula aquifer complex ***Overlaying aquifer complex area***: 22 372 km² (82%)
	Thickness	***Quaternary thickness***: 5-25 to 55-190 m (average – 60-80 m) ***Overlaying aquifer complex***: Pļaviņas-Amula aquifer complex ***Overlaying aquifer complex area***: 22 372 km² (82%)
Vulnerability		***Low risk of contamination***: 4% of the area (western part of Ropaži plain, central part of Gauja River Valley, eastern part of Seda plain) ***Medium risk of contamination***: 11% of the area ***High risk of contamination***: 3% of the area (northern part of Idumeja highland, Tālava lowland) ***Land use in high-risk area***: non-irrigated arable lands, pastures, complex cultivation patterns and lands principally occupied by agriculture, with significant areas of natural vegetation, as well as smaller discontinuous urban fabrics and industrial or commercial units.

		Part of groundwater body is covered by overlaying Pļaviņas-Amula aquifer complex. Thickness of Quaternary sediments and Pļaviņas-Amula aquifer complex, which may be variable, determines vulnerability, so the level of protection of the groundwater body may also vary from relative to very well protected.
Land use from CORINE Land Cover 2012		***Common types***: Mixed forests – 19% Pastures – 14% Transitional woodland-shrub – 14% Coniferous forests – 13% Non-irrigated arable lands – 13% Complex cultivation patterns – 7% Broad-leaved forests – 7% Other – 13%
Nitrate vulnerable zone		***Distribution***: 8% of the area (western part)
Groundwater recharge	**Main recharge mechanisms**	***Dominate***: rechange by precipitation ***Infiltration***: 6 875 000 m^3/d
	Average annual precipitation	***Average annual precipitation***: 677 mm/m2 (Riga-University, Lielpeči, Skrīveri, Sigulda, Cēsis, Zosēni, Alūksne, Gulbene, Lubāna, Rēzekne stations)
	Recharge and discharge areas	***Recharge area***: central part (Vidzeme highland), the north-eastern part (Alūksne highland), north-western part (Idumeja highland), south-eastern part (central and northern part of Latgale highland) ***Discharge area:*** Gulf of Riga and cross-border area
Monitoring	**Number of monitoring stations, number of wells**	***Quantity monitoring***: 16 stations: Akmens tilts (2 wells), Bajāri (1 well), Baldone (4 wells), Baltezers (4 wells), Carnikava (3 wells), Dzērbene (2 wells), Imanta (4 wells), Inčukalns (4 wells), Jugla (4 wells), Kalngale (3 wells), Piukas (3 wells), Riga (10 wells), Salaspils (1 well), Stirniene (1 well), Upesciems (6 wells) and Valka (1 well) . Total: 53 wells ***Quality monitoring***: 14 stations: Akmens tilts (2 wells), Baldone (3 wells), Baltezers (4 wells), Carnikava (3 wells), Dzērbene (1 well), Imanta (3 wells), Inčukalns (4 wells), Jugla (3 wells), Kalngale (3 wells), Piukas (3 wells), Salaspils (1 well), Stirniene (1 well), Upesciems (3 wells) and Valka (1 well). Total: 35 wells. 6 springs: Briņķu saltavots spring, Dukuļu spring, Ķērpju spring, Līdumnieku spring, Lielā Ellīte spring, Rūcamavots spring. • ***Operational monitoring***: 2 stations: Akmens tilts (2 wells) and Imanta (3 wells). Total: 5 wells. • ***Surveillance monitoring***: correspond to quality monitoring ***Gauja-Koiva river basin***: ***Quantity monitoring***: *4 stations: Carnikava (3 wells), Dzērbene (2 wells), Inčukalns (4 wells), Piukas (3 wells). Total: 12 wells* ***Quality monitoring***: *4 stations: Carnikava (3 wells), Dzērbene (1 well), Inčukalns (4 wells), Piukas (3 wells). Total: 11 wells.* *6 springs: Briņķu saltavots spring, Dukuļu spring, Ķērpju spring, Līdumnieku spring, Lielās Ellītes spring, Rūcamavots spring.* • ***Surveillance monitoring***: *correspond to quality monitoring*
	Types and frequency of observations	***Quantity monitoring indicators***: groundwater level from ground surface (m) ***Quality monitoring indicators***: physical-chemical indicators (all stations and springs), main ions (all stations and springs), heavy metals (all stations and springs), pesticides (stations Carnikava, Dzērbene, Inčukalns, Kalngale, Piukas and Upesciems; all springs), pesticide active substances used in Latvia (stations Carnikava, Dzērbene, Inčukalns, Kalngale, Piukas and Upesciems; all springs) and other pollutants (stations Akmens Bridge, Baldone, Imanta, Jugla, Salaspils, Stirniene and Valka). ***Frequency***: from 1 to 4 times a year, varying from 2 times in 6 years to 1 time in 2 years. • ***Operational monitoring indicators***: physical-chemical indicators (both stations), main ions (both stations), heavy metals (both stations) and other pollutants (both stations). ***Frequency***: once a year, varying from 1 time in 6 years to 2 times in 4 years • ***Surveillance monitoring***: correspond to quality monitoring ***Gauja-Koiva river basin***: ***Quantity monitoring indicators***: *groundwater level from ground surface (m)*

		Quality monitoring indicators*: physical-chemical indicators (all stations and springs), main ions (all stations and springs), heavy metals (all stations and springs), pesticides (all stations and springs) and pesticide active substances used in Latvia (all stations and springs).* ***Frequency****: from 1 to 4 times a year, varying from 2 times in 6 years to 1 time in 2 years.* • ***Surveillance monitoring****: correspond to quality monitoring*
Groundwater resources	**Groundwater well fields**	99 groundwater well fields *Gauja-Koiva river basin: 29 groundwater well fields*
	Groundwater abstraction in well fields	47 371.64 m^3/d *Gauja-Koiva river basin: 13 899.16 m^3/d*
	Calculated resources in well fields	182 566 m^3/d *Gauja-Koiva river basin: 58 848 m^3/d*
	Recharge amount	***Dominate***: Groundwater downstream flow ***Recharge***: 6 875 000 m^3/d ***Groundwater balance***: 923 000 m^3/d

Groundwater body, associated river basin district		Area
P Gauja River Basin District		Total area: 4 394 km² *Gauja-Koiva river basin: 235 km² (5%)*
Characterization of aquifers	**Type of aquifers, dominant lithology**	***Dominant material***: sandstone ***Aquitard***: aleirolite and clay ***Dominate***: porous material
	The main characteristics of aquifers	***Water conductivity coefficient (km)***: 132-650m²/d
	Thickness of aquifers	***Bedrock thickness***: 40-100 m
Covering sediments	**Lithology**	***Quaternary material***: moraine loam, moraine loam with sand-gravel-pebble interstices ***Overlaying aquifer complex***: Arukila-Amata aquifer complex
	Thickness	***Quaternary thickness***: 1-75 (Vidzeme coastline) to 15-50 m (Limbaži wavy plain) (average – 60-80 m) ***Overlaying aquifer complex***: Arukila-Amata aquifer complex
Vulnerability		***Low risk of contamination***: 100% of the area
Groundwater recharge	**Main recharge mechanisms**	***Dominate:*** rechange mostly outside the borders of Latvia ***Infiltration***: calculations have not been made, because Ķemeri-Pärnu aquifer is not included in hydrogeological model
	Recharge and discharge areas	***Recharge area***: located in a cross-border area ***Discharge area:*** Gulf of Riga and Baltic Sea
Monitoring	**Number of monitoring stations, number of wells**	***Quantity monitoring***: 1 station: Aloja (2 wells). Total: 2 wells ***Quality monitoring***: 2 stations: Aloja (2 wells) and Seda (1 well). Total: 3 wells. • ***Surveillance monitoring***: correspond to quality monitoring ***Gauja-Koiva river basin:*** no monitoring stations
	Types and frequency of observations	***Quantity monitoring indicators***: groundwater level from ground surface (m) ***Quality monitoring indicators***: physical-chemical indicators (both stations), main ions (both stations), heavy metals (both stations), pesticides (both stations), pesticide active substances used in Latvia (both stations) and other pollutants (station Aloja). ***Frequency***: once a year, varying from 1 time in 6 years to 1 time in 4 years. • ***Surveillance monitoring***: correspond to quality monitoring ***Gauja-Koiva river basin:*** no monitoring stations
Groundwater resources	**Groundwater well fields**	5 groundwater well fields *Gauja-Koiva river basin: none*
	Groundwater abstraction in well fields	631.68 m³/d *Gauja-Koiva river basin: none*
	Calculated resources in well fields	5126 m³/d ***Gauja-Koiva river basin***: *none*
	Recharge amount	***Dominate***: Groundwater downstream flow ***Recharge***: 6 875 000 m³/d ***Groundwater balance***: 923 000 m³/d

Groundwater body at risk, associated river basin district	Area
A11 Gauja River Basin District – 11 km^2 (100%)	*Total area: 11 km^2* *Gauja-Koiva river basin: 11 km^2 (100%)*

Physico-geographical characteristics		Groundwater body area is covered by plains. Groundwater body is located at the Ropaži plain in Central Latvia lowland, in its north-eastern part. ***Absolute height***: 10.0-42.7 m above sea level ***Average annual rainfall***: 750-800 mm ***Average air temperature***: summer - +17.0°C, winter - -5.0°
Characterization of aquifers	**Type of aquifers, dominant lithology**	***Dominant material***: sandstone ***Aquitard***: aleirolite and clay ***Dominate***: porous material ***Quaternary material***: sand and moraine loam
	The main characteristics of aquifers	***Water conductivity coefficient (km)***: 20-750 m^2/d • Quaternary (Q) aquifer: 20-100 m^2/d (around 40 m^2/d in the vicinity of northern ponds and around 60-100 m^2/d in the vicinity of southern ponds) • Gauja (D_3gj) aquifer: 100-750 m^2/d (highest values 700-750 m^2/d in the vicinity of southern ponds, reduced to 250-450 m^2/d in the vicinity of northern ponds, 100-200 m^2/d in the vicinity of River Gauja)
	Thickness of aquifers	***Bedrock thickness***: 70-120 m (average - 95 m) ***Quaternary thickness***: 10-15 m in north to 15-25 m in south (average - 10-20 m)
Vulnerability		***Low risk of contamination***: 12% of the area (northern part) ***Medium risk of contamination***: 75% of the area ***High risk of contamination***: 13% of the area (southern part) ***Land use in high-risk area***: coniferous and mixed forests as well as transitional woodland-shrub areas (none of which pose a threat to groundwater quality)
Land use from CORINE Land Cover 2012		***Common types***: Coniferous forests – 58% Mixed forests – 6% Transitional woodland-shrub – 17% Other – 19%
Nitrate vulnerable zone		***Distribution***: 100% of the area
Groundwater recharge	**Main recharge mechanisms**	Resolution of the model does not allow to accurately determine the groundwater balance in the area
	Average annual precipitation	***Average annual precipitation***: 809 mm/m^2 (Sigulda station)
	Recharge and discharge areas	***Recharge area***: all of the area ***Discharge area***: River Gauja
Monitoring	**Number of monitoring stations, number of wells**	***Quantity monitoring***: 1 station: Inčukalns (3 wells) ***Quality monitoring***: 1 station: Inčukalns (3 wells) Gauja-Koiva river basin: as listed above
	Types and frequency of observations	***Quantity monitoring indicators***: groundwater level from ground surface (m) ***Quality monitoring indicators***: physico-chemical indicators, main ions, heavy metals, pesticides and pesticide active substances used in Latvia ***Frequency***: from 1 to 4 times a year, varying from 1 time in 4 years to 2 times in 4 years. Gauja-Koiva river basin: as listed above
Groundwater resources	**Groundwater well fields**	1 groundwater well field Gauja-Koiva river basin: as listed above
	Groundwater abstraction in well fields	No groundwater abstraction Gauja-Koiva river basin: as listed above

	Calculated resources in well fields	300 m^3/d **Gauja-Koiva river basin**: as listed above
	Recharge amount	Resolution of the model does not allow to accurately determine the groundwater balance in the area

Annex 2

Characteristics of groundwater bodies delineated in Estonia (inside Koiva river basin)

Middle Devonian groundwater body in Koiva river basin district (no. 25)
(from Marandi et al. 2019)

No.	River basin district	Groundwater body group	Aquifer system	County	Area (km^2)
25	Koiva	Devonian	Quaternary, Middle-Devonian	Valga County, Võru County	1322

			Figure No.
Hydrogeological description	*Lithological composition of the aquifer- forming rocks*	Aquifers related to the groundwater body are hosted by brownish fine-grained sand- and siltstones of the Middle Devonian Aruküla, Gauja and Burtnieki Formations that contain interlayers of clay. The groundwater body also comprises the aquifers in Quaternary sediments overlying the Middle Devonian rocks. The most important of those Quaternary aquifers are the aquifers hosted by glaciofluvial gravels and sands in the Haanja and Varstu parishes in the Võru County. These aquifers are usually located as lenses in a more widespread till complex (Perens et al., 2012).	1
	Thickness of the groundwater body	Mostly 150–280 m, reaching up to ~280 m in its southern border, depending on the thickness of the Quaternary sediments covering the bedrock.	1
	Overlying aquitard	The western part of the groundwater body is overlain by relatively impermeable loamy till with a hydraulic conductivity of 0.1–1.0 m/d. In the eastern part of the groundwater body, the bedrock sequence is covered by a clayey Snetnaja Gora-Amata aquitard with a thickness of 8–10 m (Perens & Vallner, 1997).	1
	Underlying aquitard	The underlying aquitard associated with the groundwater body is the regional Narva aquitard in the clayey siltstones, clays and dolomitic marls of the Middle Devonian Narva Formation. The aquitard has a transversal hydraulic conductivity of 10^{-4}–10^{-5} m/d on average, locally decreasing down to ≤10^{-6} m/d.	1
	Groundwater levels and potentiometric surface	In the Haanja Upland the potentiometric surface lies at depths of 112–122 m from the ground surface (120–123 m asl). In the depressions between the uplands the potentiometric surface occasionally reaches above the ground surface. In the unconfined Quaternary aquifers, the groundwater levels are located at depths of 0.5 to 57 m below the ground surface (on average 10 m; Perens et al., 2012).	2
Hydrodynamics	*Groundwater flow directions*	The direction of the groundwater flow is mainly determined by the Haanja and Karula Uplands. The groundwater flows to the south and east from the Karula Upland and to the south and west from the Haanja Upland. A local discharge area is found in the valley of the Mustjõgi river located between the two uplands.	2
	Hydraulic conductivity	The lateral hydraulic conductivity of the bedrock aquifers varies from 1 to 3 m/d and the hydraulic conductivity of the Quaternary aquifers is 5 m/d on average (Perens et al., 2012). The transmissivity of the aquifers is very variable, having values of 30–300 m^2/d, that increase southwards together with an increase in the thickness of the aquifer-forming rocks. The hydraulic gradient of aquifers in the vicinity of the south Estonian uplands vary from 0.0001 to 0.1 and the lateral groundwater flow velocity in the sandstones has been estimated to be 0.001-0.005 m/d (Perens et al., 2012). The estimated lateral groundwater	1

		flow velocity in Quaternary sediments is ~0.001-0.15 m/d (in gravel up to 10-15 m/d; *Ibid.*).	
	Recharge and seasonal variations in groundwater levels	Groundwater flows from the recharge areas in the Karula and Haanja Uplands to discharge areas in the lower lying regions. The amount of recharge is dependent on the thickness and composition of the Quaternary sediments. In areas with loamy overburden and waterlogged soils the recharge is low or does not occur at all. Seasonal variations in groundwater levels are mostly between 0.3 and 2 m.	1
Composition of groundwater	***Chemical composition***	Groundwater in the groundwater body is mainly of Ca-HCO_3^-type with low TDS concentrations varying from 200 to 600 mg/l. The Cl^- concentrations are ≤25 mg/l. NO_3^- concentrations are low and are below 10 mg/l. In terms of drinking water quality, the most important characteristic of groundwater is its high natural Fe concentrations (about 4.5 mg/l on average in the sandstones and 1.0-3.0 mg/l in the Quaternary sediments; Perens et al., 2012). The natural SO_4^{2-} concentrations are low and do not exceed 20 mg/l. Groundwater in the groundwater body is usually compliant with drinking water quality standards, bar groundwater with higher iron concentrations.	3, 4, 5, 6
	Conceptual model on the hydrochemical evolution of groundwater	The chemical composition of groundwater in the groundwater body has mainly evolved through the dissolution of carbonate cement in the Devonian sandstones by infiltrating meteoric water. In deeper aquifers, dolomite dissolution also plays an important role, which is evident in the increasing Mg^{2+} concentrations. The Ca-HCO_3^- type groundwater found in the Quaternary sediments is related to carbonate minerals found in tills. High iron concentrations in groundwater indicate that aquifers associated with the groundwater body are under reducing conditions. The sulphate probably originates from pyrite oxidation.	7
Groundwater dependent ecosystems (TLÜ Institute of Ecology, 2015)	**Groundwater dependent river water bodies**	Mustjõgi river to Antsla-Litsmetsa road (Mustjõgi; 1031000_1); Kolga river (Kolga; 1158400_1); Pärlijõgi river from Saarlase dam to river mouth (Pärlijõgi_2; 1155700_2). The groundwater dependent river water bodies related to the groundwater body have a good status.	
	Groundwater dependent lake water bodies	*Groundwater dependent lake water bodies related to the Quaternary aquifers of the groundwater body:* • Hanija lake (VEE2150700); • Kikkajärv (VEE2152100); • Liivajärv (Paganamaa Liivajärv; VEE2152300); • Maiori lake (VEE2152000); • Mudajärv (Paganamaa Mudajärv; VEE2152310); • Murati lake (VEE2155900); • Palujüri lake (VEE2150800); • Sarapuujärv (Paganamaa Sarapuujärv; VEE2152200); • Sarise lake (VEE2154800); • Väiku-Palkna lake (VEE2151710). The status of the Murati lake is moderate due to pH values and water transparency. Other groundwater dependent lake water bodies have a good status, or their status has not been assessed.	
	Groundwater dependent terrestrial ecosystems	Spring fens of the Mustjõe river valley. Groundwater dependent terrestrial ecosystems related to the groundwater body have been influenced by drainage canals and occasionally also by beaver dams.	
Status assessment (Perens et al., 2015)	**Quantitative status**	Good	
	Chemical status	Good	
	Status	Good	

Groundwater resources (m^3/d)	Natural resources (NR)	536689
	Allocated groundwater resources (AGR)	-
	Groundwater abstraction (2017; GA)	164
	Free unallocated resources (AGR-GA)	-164
	Minimal estimate of the natural groundwater resources available for allocation (NR-AGR)	536689
	Minimal estimate of the natural groundwater resources available for abstraction (NR-GA)	536525

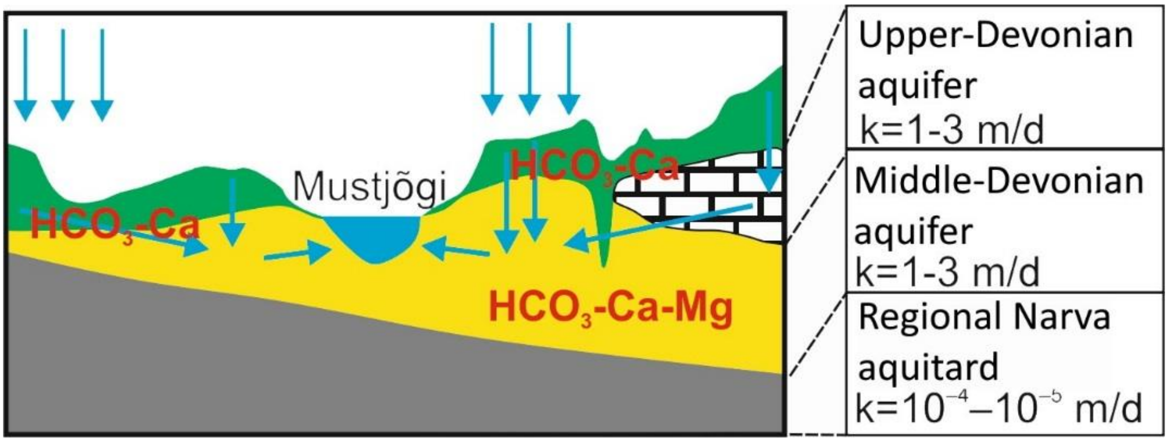

FIGURE 1 **A cross-section of a conceptual model for the Middle Devonian groundwater body in Koiva river basin district (No.25) and the Upper Devonian groundwater body (No.26)**

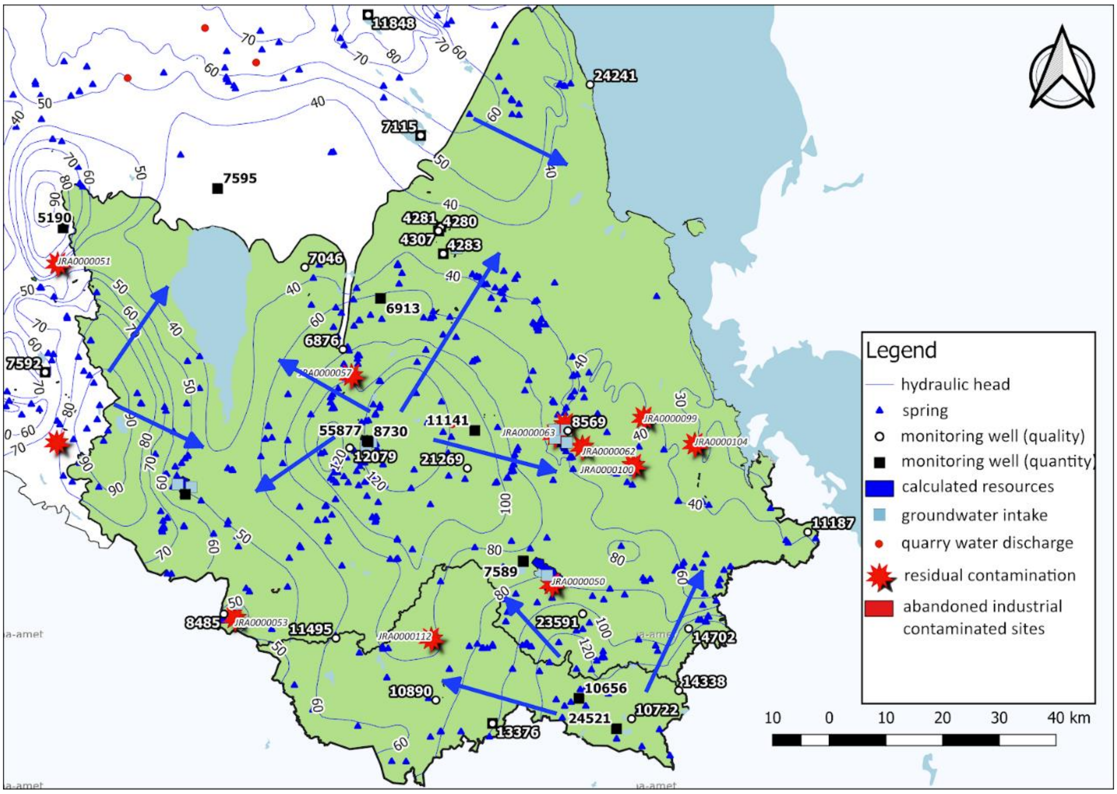

FIGURE 2 **A planar view of the conceptual model for the Middle Devonian groundwater body in the East Estonian river basin district (No.24) and for the Middle Devonian groundwater body in Koiva river basin district (No.25). Conceptual model depicts the hydraulic heads, the main groundwater flow directions, locations of springs and monitoring wells, groundwater intakes and abstraction wells together with the contaminated sites or abandoned industrial sites (pressure no 1.5). The hydraulic heads are taken from Perens et al. (2012)**

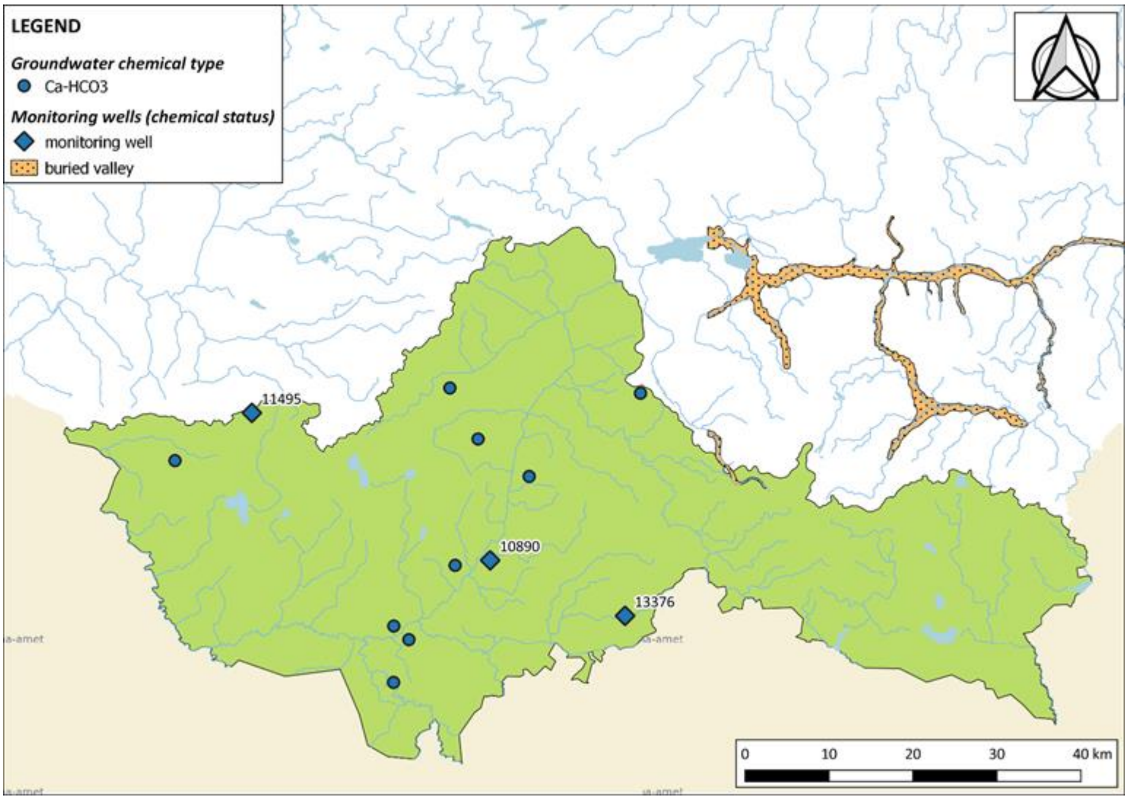

FIGURE 3 **Groundwater chemical types in the Middle Devonian groundwater body in Koiva river basin district together with the position of the monitoring wells for chemical status assessment (shown as diamonds)**

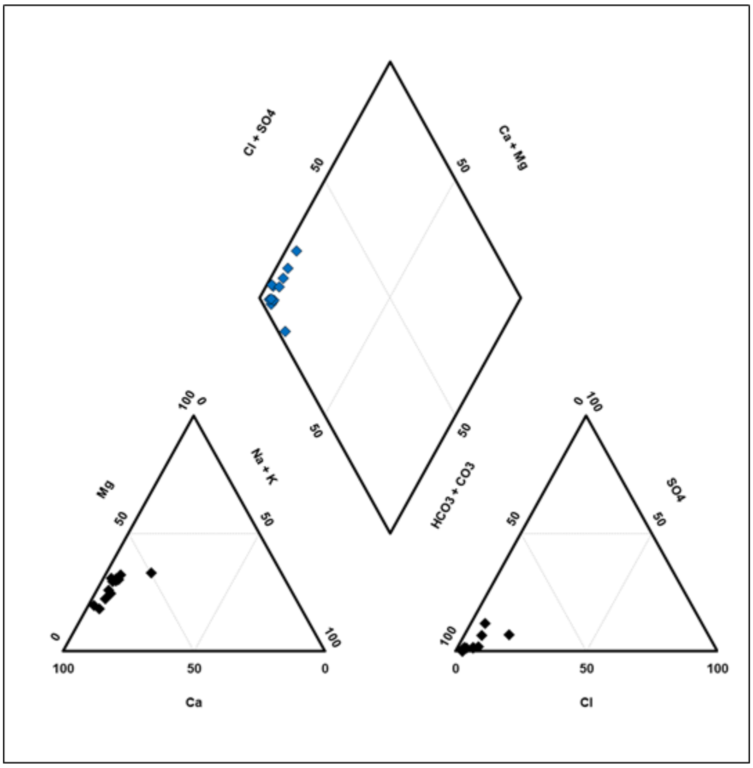

FIGURE 4 **Groundwater types in the Middle Devonian groundwater body in Koiva river basin district (No.25) shown on a Piper diagram. Groundwater types: Ca-HCO3 – blue; Ca-Cl – green; Na-HCO3 – yellow; Na-Cl – red**

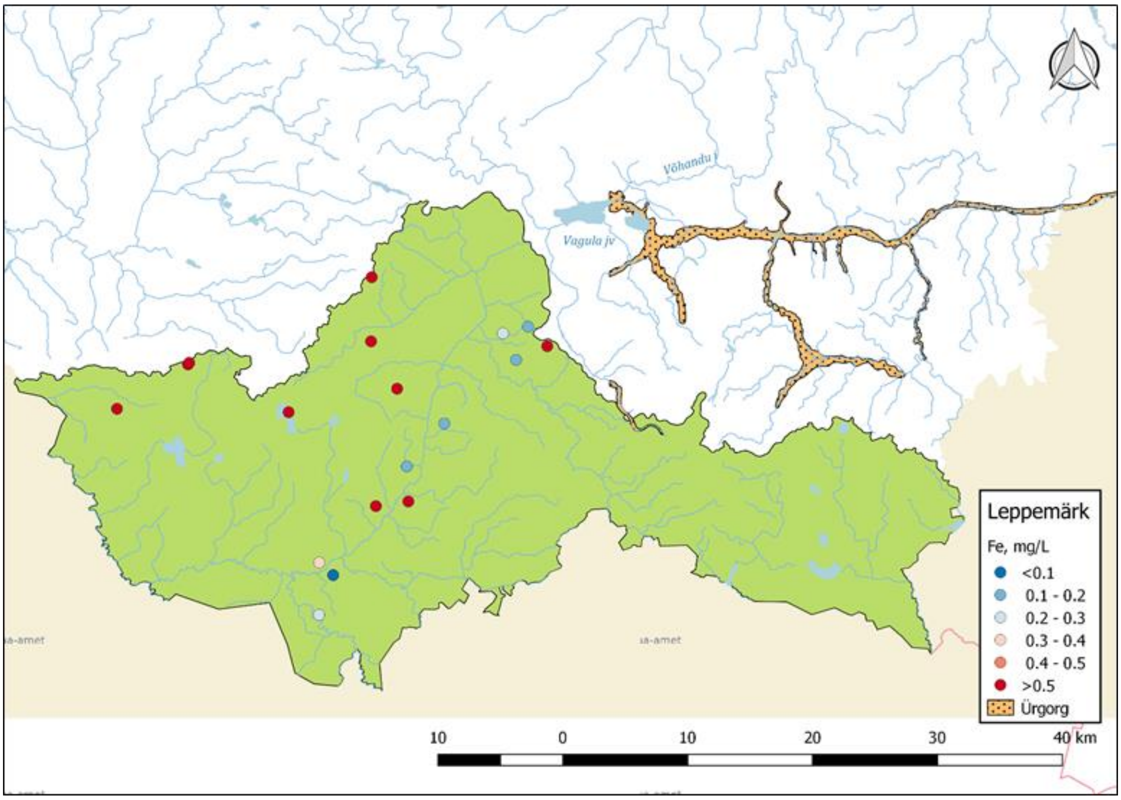

FIGURE 5 **Dissolved iron (Fe_{tot}, mg/l) concentrations in the Middle Devonian groundwater body in Koiva river basin district**

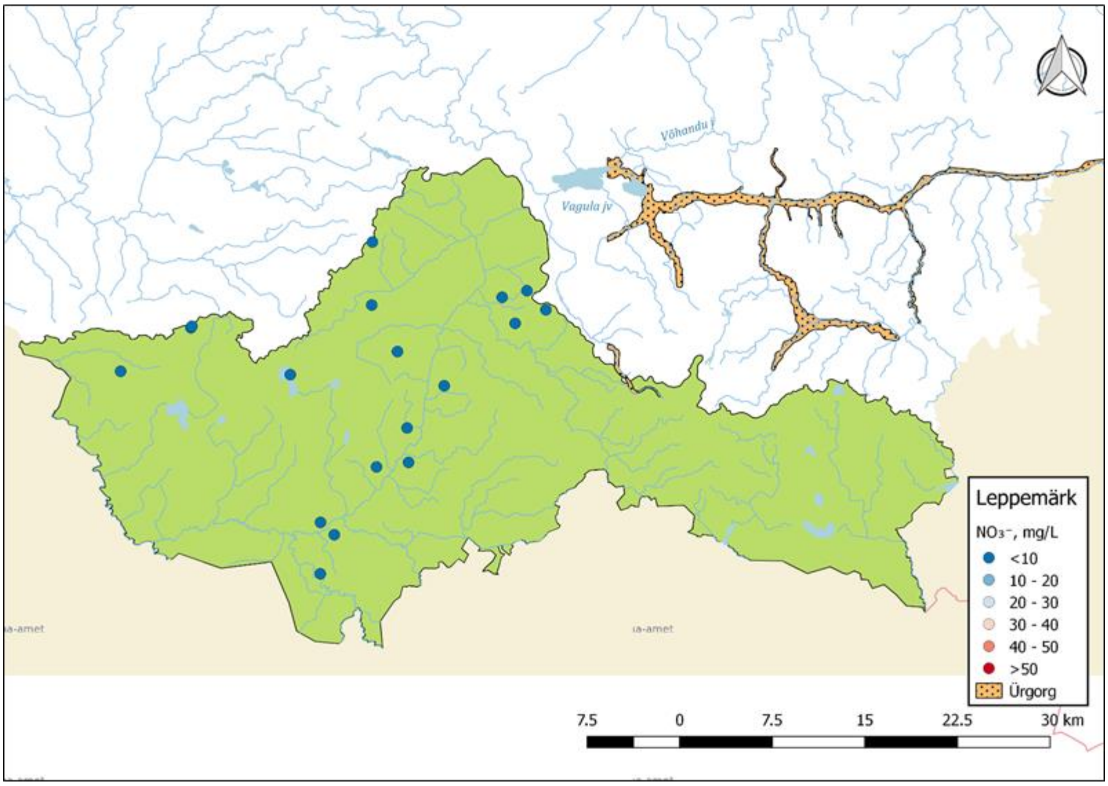

FIGURE 6 **Nitrate (NO_3^-) concentrations in the Middle Devonian groundwater body in Koiva river basin district**

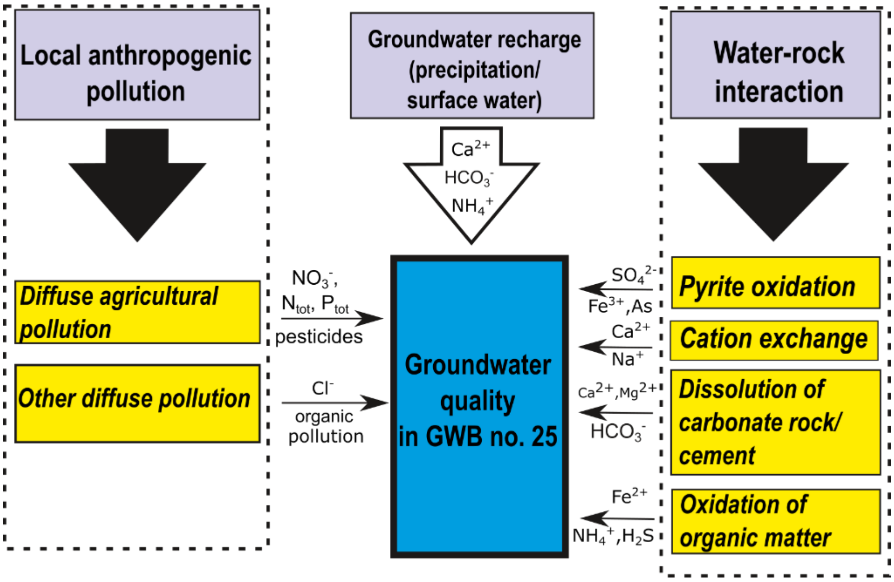

FIGURE 7 **Conceptual model of the hydrochemical evolution in the Middle Devonian groundwater body in the Koiva river basin district (D2) (No.5)**

Upper Devonian groundwater body in Koiva river basin district (no. 26)
(from Marandi et al. 2019)

No.	River basin district	Groundwater body group	Aquifer system	County	Area (km^2)
26	East Estonian/Koiva	Devonian	Quaternary, Upper-Devonian	Valga County, Võru County	726,1

			Figure No.
Hydrogeological description	*Lithological composition of the aquifer-forming rocks*	The lithological composition of the aquifer-forming rocks is quite homogenous. The aquifers are hosted by thick-bedded limestone and dolomitized limestone of the Upper Devonian Plavinas Stage and the overlying Quaternary sediments. The lower part of the formation consists of domerite and marl of the Snetnaja Gora Formation, which can be viewed as a local semi-permeable aquitard.	1 (at GWB No.25)
	Thickness of the groundwater body	The thickness of the bedrock aquifers is in the range of 30–40 m; the thickness of the overlying Quaternary deposits is mostly in the range of 5–10 m, locally up to 20 m.	1 (at GWB No.25)
	Overlying aquitard	The Quaternary sediments overlying the bedrock aquifers consist mainly of loamy till, which has a hydraulic conductivity of 0.1–1.0 m/d.	1 (at GWB No.25)
	Underlying aquitard	The domerite, marl and clay of the Snetnaja Gora Formation	1 (at GWB No.25)
	Groundwater levels and potentiometric surface	The aquifers are mostly phreatic. Groundwater level is usually about 20–30 m below ground surface. The absolute height of the groundwater level is in the range of 165–175 m.	2 (at GWB No.25)
Hydrodynamics	*Groundwater flow directions*	The most important groundwater divide in the area is the Haanja Heights, from where the groundwater flows to the south and west towards the edges of the height. Groundwater seeps out in the river valleys and a portion of its volume also infiltrates deeper into the Middle-Devonian aquifers.	2 (at GWB No.25)
	Hydraulic conductivity	The transmissivity of the aquifers forming the groundwater body is in the range of 30–300 m^2/d (Perens et al., 2012). The lateral flow velocity of groundwater is in the range of 1–10 m/d and can reach up to 50 m/d in karst aquifers (*Ibid.*).	1 (at GWB No.25)
	Recharge and seasonal variations in groundwater levels	The groundwater flows radially away from the Haanja Heights and the local hillocks towards topographically lower regions throughout the year. The amount of infiltrating water depends on the composition of local Quaternary cover. In areas with waterlogged soils or in areas underlain by clayey deposits the infiltration rate can be negligible.	1 (at GWB No.25)
Composition of groundwater	*Chemical composition*	Groundwater in the groundwater body is mainly of the Ca-HCO_3-type, with TDS concentrations ranging from 200 to 600 mg/L. The chloride concentrations are usually <15 mg/L. The concentrations of NO_3^- are also low and do not exceed 5 mg/L in most cases. In terms of drinking water quality, the most important characteristic of groundwater is its high natural Fe concentration (up to 3 mg/L; 1.8 mg/L on average). Locally high NH_4^+ concentrations are also observed (up to 2 mg/L; 0.2 mg/L on average). The natural background concentration of sulphates is low with concentrations <20 mg/L. Groundwater in the groundwater body is usually compliant with drinking water quality standards, except groundwater with higher iron or ammonium concentrations.	8, 9

	Conceptual model on the hydrochemical evolution of groundwater	The chemical composition of groundwater in the groundwater body has mainly evolved through the dissolution of carbonate minerals (mostly calcite) by infiltrating meteoric water. In deeper aquifers the dolomite dissolution causes an increase in Mg^{2+} concentrations. High Fe concentrations in groundwater indicate that aquifers associated with the groundwater body are under reducing conditions. The sulphate probably originates from pyrite oxidation.	7 (at GWB No.25)
Groundwater dependent ecosystems (TLÜ Institute of Ecology, 2015)	**Groundwater dependent river water bodies**	• Obinitsa river (Obinitsa; 1001900_1); • Piusa river up to Kivioja creek (Piusa_1; 1000200_1); • Piusa river from Kivioja creek to the river mouth (Piusa_2; 1000200_2); • Rõuge river (Rõuge; 1004100_1); • Pärlijõgi from Saarlase dam to the river mouth (Pärlijõgi_2; 1155700_2. The groundwater dependent river water bodies Piusa_1, Rõuge and Pärlijõgi_2 is in good status. The status of Obinitsa river water body is moderate and Piusa_2 body has been assessed to be in bad status. Historical data on the baseflow proportions to the total river discharge are the following: -50% in the Piusa_1 river water body (in the period 1931-1962); -50% in the Piusa_2 river water body (in the period 1931-1962).	
	Groundwater dependent lake water bodies	*Groundwater dependent lake water bodies related to the groundwater body:* • Kaussjärv (VEE2140200); • Liinjärv (VEE2140400); • Ratasjärv (Rõuge Ratasjärv; VEE2140100); • Suurjärv (Rõuge Suurjärv; VEE2140300); • Tõugjärv (VEE2140000); • Valgjärv (Rõuge Valgjärv; VEE2140500). • All groundwater dependent lake water bodies related to the groundwater body are in good status. *Important karst features related to the groundwater body:* • Tsiistre dolines; • Poksa ponor; • Palosland karst lake.	
	Groundwater dependent terrestrial ecosystems	Spring mires of the Rõuge buried valley. Groundwater dependent terrestrial ecosystems related to the groundwater body are relatively unaltered by the anthropogenic activities.	
Status assessment (Perens et al., 2015)	**Quantitative status**		Good
	Chemical status		Good
	Status		Good
Groundwater resources (m^3/d)	**Natural resources (NR)**		221586
	Allocated groundwater resources (AGR)		
	Groundwater abstraction (2017; GA)		65
	Free unallocated resources (AGR-GA)		
	Minimal estimate of the natural groundwater resource available for allocation (NR-AGR)		221586
	Minimal estimate of the natural groundwater resource available for abstraction (NR-GA)		221522

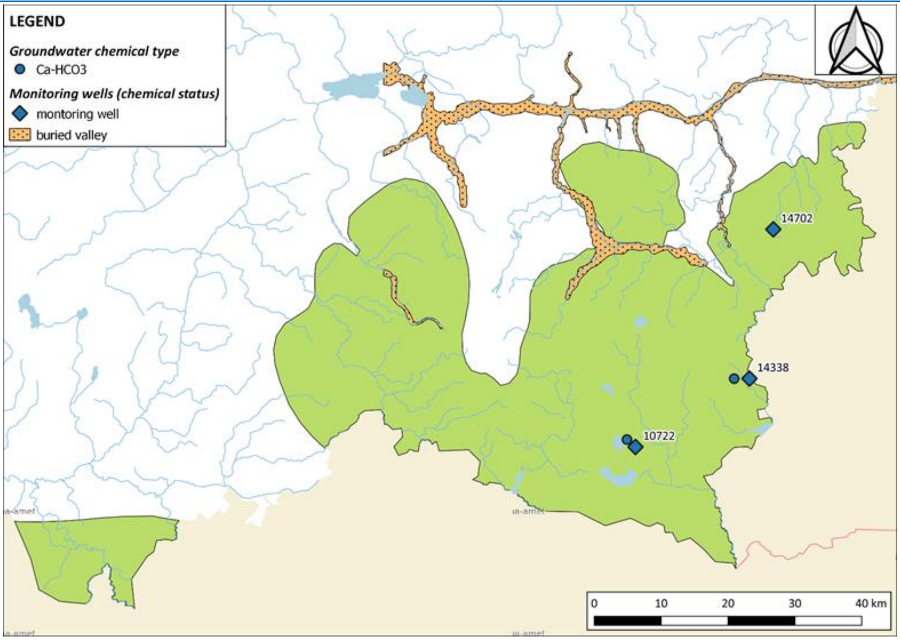

FIGURE 8 **Groundwater chemical types in the Upper Devonian groundwater body together with the positions of the monitoring wells for chemical status assessment (shown as diamonds)**

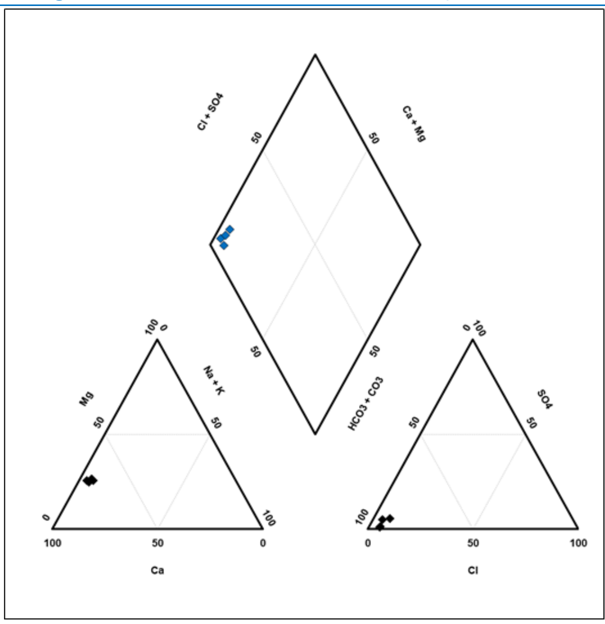

FIGURE 9 **Groundwater types in the Upper Devonian groundwater body shown on a Piper diagram.** ***Groundwater types:*** **Ca-HCO_3 – blue; Ca-Cl – green; Na-HCO_3 – yellow; Na-Cl – red**

Groundwater monitoring points in Gauja-Koiva river basin

Country	Monitoring station	Type	Aquifer system	GWB	Database number	Start of observation	X, m	Y, m	Aquifer	Filter interval, m	Quality	Quality Frequency*	Observation period	Quantity	Quantity Frequency*	Observation period
LAT	Carnikava	well	Middle-Upper Devonian multi-aquifer system	A8	1485	1979	332713	516300	D_2ar	184.0-202.6	Yes	1 x 4 year	1980-till today	Yes	730 x a year	1979-till today
LAT	Carnikava	well	Middle-Upper Devonian multi-aquifer system	A8	1486	1979	332716	516295	D_2br	133.0-154.0	Yes	1 x 4 year	1980-till today	Yes	730 x a year	1979-till today
LAT	Carnikava	well	Quaternary water-table aquifer	Q	1488	1979	332721	516290	m,l,lg Q_3ltvb- Q_4	17.4-23.0	Yes	1 x 4 year	1979-till today	Yes	730 x a year	1979-till today
LAT	Carnikava	well	Middle-Upper Devonian multi-aquifer system	A8	19140	1988	332735	516289	D_3gj_1	79.1-87.7	Yes	1 x 4 year	1988-till today	Yes	730 x a year	2000-till today
LAT	Dzērbene	well	Plavinas-Amula and Middle-Upper Devonian Multi-aquifer systems	D6	9642	1975	339448	601403	D_3am-slp	82.0-130.0	Yes	1 x 4 year	1975-till today	Yes	4 x a year	1976-till today
LAT	Dzērbene	well	Middle-Upper Devonian multi-aquifer system	A8	9641	1996	344772	596713	D_3gj+am	170.0-270.0	No	-	-	Yes	4 x a year	1996-till today
LAT	Dzērbene	well	Plavinas-Amula and Middle-Upper Devonian Multi-aquifer systems	D6	9643	2005	344767	596713	Q	2.1-2.6	No	-	-	Yes	4 x a year	2005-till today
LAT	Inčukalns	well	Middle-Upper Devonian multi-aquifer system	A8	1489	1978	331373	539036	D_2pr	346.0-356.0	Yes	1 x 4 year	1978-till today	Yes	730 x a year	1978-till today
LAT	Inčukalns	well	Middle-Upper Devonian multi-aquifer system	A8	1490	1978	331403	539022	D_2nr	248.0-258.0	Yes	1 x 4 year	1979-1997	Yes	730 x a year	1978-till today
LAT	Inčukalns	well	Middle-Upper Devonian multi-aquifer system	A8	1491	1978	331367	539036	D_2ar	214.0-221.0	Yes	1 x 4 year	1978-till today	Yes	730 x a year	1978-till today
LAT	Inčukalns	well	Middle-Upper Devonian multi-aquifer system	A8	1492	1978	331405	539021	D_2br	158.0-175.0	Yes	1 x 4 year	1978-till today	Yes	730 x a year	1978-till today
LAT	Inčukalns	well	Middle-Upper Devonian multi-aquifer system	A11	1493	1978	331402	539023	D_3gj_1	108.0-128.0	Yes	1 x 4 year	1978-till today	Yes	730 x a year	1978-till today
LAT	Inčukalns	well	Middle-Upper Devonian multi-aquifer system	A11	1494	1978	331406	539020	D_3gj_2	46.0-59.0	Yes	1 x 4 year	1978-till today	Yes	730 x a year	1978-till today
LAT	Inčukalns	well	Middle-Upper Devonian multi-aquifer system	A11	1495	1978	331408	539019	Q	6.0-12.0	Yes	2 x 4 year	1997-till today	Yes	730 x a year	1978-till today
LAT	Piukas	well	Middle-Upper Devonian multi-aquifer system	A8	1	1978	326110	527839	D_3gj	36.0-43.0	Yes	2 x 4 year	1978-till today	Yes	730 x a year	1979-till today
LAT	Piukas	well	Middle-Upper Devonian multi-aquifer system	A8	2	1978	326107	527836	D_2ar	152.0-162.0	Yes	1 x 4 year	1978-till today	Yes	730 x a year	1979-till today
LAT	Piukas	well	Middle-Upper Devonian multi-aquifer system	A8	3	1978	326102	527834	D_3gj_1	72.0-84.0	Yes	1 x 4 year	1978-till today	Yes	730 x a year	1979-till today
LAT	Piukas	well	Quaternary water-table aquifer	Q1	4	1978	326098	527834	aQ_4	6.0-11.0	Yes	2 x 4 year	1978-till today	Yes	730 x a year	1979-till today
LAT	Velēna	well	Plavinas-Amula and Middle-Upper Devonian Multi-aquifer systems	D6	9645	1961	346379	645575	D_3dg	64.0-75.1	Yes	1 x 4 year	1961-till today	Yes	730 x a year	2002-till today
LAT	Velēna	well	Plavinas-Amula and Middle-Upper Devonian Multi-aquifer systems	D6	9646	1967	346376	645572	Q	4.7-5.9	Yes	2 x 4 year	1967-till today	Yes	730 x a year	2002-till today
LAT	Virāne	well	Plavinas-Amula and Middle-Upper Devonian Multi-aquifer systems	D6	14445	2010	326612	645102	D_3dg	80.0-90.0	Yes	1 x 4 year	2010-till today	Yes	730 x a year	2010-till today
LAT	Virāne	well	Plavinas-Amula and Middle-Upper Devonian Multi-aquifer systems	D6	9647	1961	326614	645106	D_3kt+og	50.3-69.0	Yes	1 x 4 year	1961-till today	Yes	730 x a year	2002-till today
LAT	Virāne	well	Plavinas-Amula and Middle-Upper Devonian Multi-aquifer systems	D6	9648	1967	326616	645111	Q	13.0-17.2	Yes	2 x 4 year	1967-till today	Yes	730 x a year	2002-till today
LAT	Bānūžu avots	spring	Plavinas-Amula and Middle-Upper Devonian Multi-aquifer systems	D6	24550	2006	336069	595025	Q	-	Yes	2 x 4 year	2006-till today	No	-	-
LAT	Briņķu saltavots	spring	Middle-Upper Devonian multi-aquifer system	A8	24551	2006	368304	576272	D_3gj	-	Yes	2 x 4 year	2006-till today	No	-	-
LAT	Dāvida dzirnavu avots	spring	Plavinas-Amula and Middle-Upper Devonian Multi-aquifer systems	D6	24552	2004	348072	583708	D_3pl	-	Yes	2 x 4 year	2004-till today	No	-	-
LAT	Dukuļu avots	spring	Middle-Upper Devonian multi-aquifer system	A8	24553	2004	370071	583030	D_3gj	-	Yes	2 x 4 year	2004-till today	No	-	-
LAT	Ķērpju avots	spring	Middle-Upper Devonian multi-aquifer system	A8	24555	2006	355386	592411	D_3pl	-	Yes	2 x 4 year	2006-till today	No	-	-
LAT	Līdumnieku avots	spring	Middle-Upper Devonian multi-aquifer system	A8	24556	2004	341773	553087	D_3gj+am	-	Yes	2 x 4 year	2004-till today	No	-	-
LAT	Lielās Ellītes avots	spring	Middle-Upper Devonian multi-aquifer system	A8	24557	2006	361032	585789	D_3am	-	Yes	2 x 4 year	2006-till today	No	-	-
LAT	Mežmuižas avots	spring	Plavinas-Amula and Middle-Upper Devonian Multi-aquifer systems	D6	24558	2004	329024	548404	D_3pl	-	Yes	2 x 4 year	2004-till today	No	-	-
LAT	Rūcamavots	spring	Middle-Upper Devonian multi-aquifer system	A8	24559	2004	353920	573551	D_3gj	-	Yes	2 x 4 year	2004-till today	No	-	-
LAT	Saltavots	spring	Plavinas-Amula and Middle-Upper Devonian Multi-aquifer systems	D6	24560	2004	332290	551514	D_3pl	-	Yes	2 x 4 year	2004-till today	No	-	-
LAT	Vecstrautu avots	spring	Plavinas-Amula and Middle-Upper Devonian Multi-aquifer systems	D6	24562	2006	363481	617118	Q	-	Yes	2 x 4 year	2006-till today	No	-	-
LAT	Zīļu avots	spring	Plavinas-Amula and Middle-Upper Devonian Multi-aquifer systems	D6	24563	2006	379621	662194	D_3pl	-	Yes	2 x 4 year	2006-till today	No	-	-
EST	Lüllemäe	well	Middle-Devonian aquifer system	25	11890	2018	403306	641363	D_2tr	74.2-90.0	Yes	1 x year	2018-till today	No	-	-
EST	Varstu alevik	well	Middle-Devonian aquifer system	25	10890	2005	392320	658874	D_2gj-ar	83.0-123.0	Yes	1 x year	2005-till today	No	-	-
EST	Ruusmäe küla	well	Middle-Devonian aquifer system	25	10656	2018	392623	684137	D_2sv	153.1-189.3	No	-	-	Yes	12x a year	2018 till today
EST	Krabi põhikooli puurkaev	well	Quaternary and Middle-Devonian aqufer system	25**	13376	1997	388200	668863	f Q_3	9.6-15.5	Yes	1 x year	2014-till today	Yes	4 x year	1997-till today
EST	Kaubi küla	well	Upper-Devonian aqufer system	26	24521	2018	387283	690736	D_3pl	42-70	No	-	-	Yes	12 x a year	2018-till today
EST	Misso alevik	well	Upper-Devonian aqufer system	26	10722	2005	389030	693400	D_3pl	44-53	Yes	1 x year	2005-till today	No	-	-
EST	Lüta küla, Luhamaa piiripunkt	well	Upper-Devonian aqufer system	26	14338	2005	394026	701735	D_3pl	50-60	Yes	1 x year	2005-till today	No	-	-

*The frequency of observations is indicated by a long-term up-to-date monitoring program

**GWB old No.39

Map of groundwater monitoring points in Gauja-Koiva river basin