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Quickstart guide for groundwater studies in mining environments

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Photo: Kaisa Turunen (GTK); Bedrock groundwater sampling from a packer insulated interval at a mine site.

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Abstract
Water management is one of the key environmental and ecological questions for mines, and groundwater management is often part of that problem. In Finnish conditions there is an abundance of rainfall, and adding the groundwater in the equation, the amounts of water that need to be controlled can become a major issue environmentally, technically and economically.

Traditionally, groundwater studies at Finnish mines have been conducted in soft, Quaternary sediments, i.e. in porous medium, while in many cases, fractured bedrock has been observed to be more important for the water balance of the pit and the mine site. In many sites the aquifers in Quaternary sediments are shallow, discontinuous and have low hydraulic conductivity, thus making them less important than fractured bedrock aquifers and transport routes. Further, in Finland all quarries are excavated in hard, crystalline bedrock and the water balance in many cases is governed by the groundwater from the fractured aquifers and the direct rain to the pit and its catchment. Every mine site is different and must be evaluated thoroughly whether the hydrogeological research efforts should be concentrated on fractured bedrock or porous sediments and on which proportion.

Early stage groundwater studies are essential when building the most eco-efficient mine. The location of different processes (e.g. waste dumps) should be designed so that they do not pose a threat to the groundwater, even in case of accidents. Other factors such as slope stability and different water balances are also affected by groundwater and should be studied in an early stage. Hydrogeological studies cannot only provide answers to these issues, but they can be used as input data for flow and transport modelling studies. Modelling can provide invaluable information about effective water management methods and actions, but also forecast the fate of the possibly adverse substances in the future. Economically, early stage investigations and proper preparation for different water management scenarios can make a huge difference between a successful and failing mine.

This guidebook collects the methodologies and experiences GTK has received in groundwater studies in mining areas in the past few years. It does not cover all the aspects or methods of mining hydrogeology, but provides an outlook of macro scale studies performed in Finland. For a hydrogeology expert the guidebook is a checklist of common methodologies and equipment needed in those. For a non-hydrogeology expert, the guidebook gives an overview of different studies and available methodologies in relatively simple terms. It also shows that performing valid hydrogeological surveys at mine sites requires collaboration between multiple disciplines of geological- and other sciences.

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1 INTRODUCTION

Groundwater is one of the major factors to be counted for in almost every stage of a mines life. In Finnish conditions there is an abundance of rainfall, and adding the groundwater in the equation, the amounts of water that need to be controlled can become a big issue environmentally, technically and economically. Contrarily, in some parts of the world finding an adequate yield groundwater source can be a prerequisite for successful mining in otherwise arid environment.

Traditionally, groundwater studies in Finnish mines have been conducted in soft, Quaternary sediments, *i.e.* in porous medium. One of the reasons lies in our educational system, where hydrogeologists are graduated from Quaternary geology studies. Also, investigations in porous medium are much easier to conduct than in fractured bedrock. On the other hand, authorities have not demanded fractured hydrogeology studies, with a few exceptions, which is expected to change in the future.

At many mining sites in Finland, fractured bedrock has shown to be more important for the water balance of the pit and the mine site, than groundwater in Quaternary sediments. In many sites the aquifers in Quaternary sediments are shallow, discontinuous and have low hydraulic conductivity, thus making them less important than fractured bedrock aquifers and transport routes. On the other hand, at some sites, Quaternary sediment aquifers play a huge role in the mine’s water balance. Despite, in Finland, all quarries are excavated in hard, crystalline bedrock and the water balance in many cases is governed by the groundwater from the fractured aquifers and the direct rain to the pit and its catchment. Every mine site is different and must be evaluated thoroughly whether the hydrogeological research efforts should be concentrated on fractured bedrock or porous sediments and on which proportion.

Early stage groundwater studies are essential when building the most eco-efficient mine. Location of different processes (*e.g.* waste dumps) should be designed such that they do not pose a threat to the groundwater, even in case of accidents. Other factors such as slope stability and different water balances are also affected by groundwater and should be studied in an early stage. Hydrogeological studies cannot only provide answers to these issues, but they can be used as input data for flow and transport modelling studies. The modelling studies can provide invaluable information about effective water management methods and actions, but also forecast the fate of the possibly adverse substances in the future. Economically, early stage investigations and proper preparation for different water management scenarios can make a huge difference between a successful and failing mine.

This guidebook collects the methodologies and experiences GTK has received in bedrock groundwater studies in mining areas in the past few years. It does not cover all the aspects or methods of mining hydrogeology, but provides an outlook of macro scale studies performed in Finland. For a hydrogeology expert the guidebook is a checklist of common methodologies and equipment needed in those. For a non-hydrogeology expert the guidebook gives an overview of different studies and available methodologies in relatively simple terms. It also shows that performing valid hydrogeological surveys at mine sites requires collaboration between multiple disciplines of geological- and other sciences. The guidebook should also show that in many cases simple methodological approaches can only provide very broad answers. Several methodologies and their simultaneous interpretation is needed to provide acceptable results.

2 PRE-STUDIES AND DESIGN OF DRILLING PLAN

Before actual hydrogeological studies can be performed, drilling locations need to be designed and drillings need to be performed. In hydrogeological studies in bedrock, the main fracture zones and major fissures are the most probable pathways for water, whereas minor fissures and low permeability bedrock usually play a much smaller role. The resolution of the geophysical methodologies presented below are suitable for macro scale hydrogeological studies. In cases where the micro scale studies are needed, special methods, such as the Posiva Flow Log might be the only option (these advanced methods are discussed in further detail in Chapter 3.7).

The design of hydrogeological drilling locations in bedrock can be a tedious task, if only maps or digital terrain models (DTM) are available. Therefore, the design should be based on geophysical data, and most of the drillings should be located in most probable places with fracture zones containing possibly high discharge. Drillings in low permeability bedrock are also needed for comparison. To keep the drilling costs as low as possible, suitability of previous mineral exploration drill holes for hydrogeological studies, should be studied.

The design of ground geophysical investigations can also be difficult, as for example, measurement lines should cross the major hydrogeological features close to right angles. In most cases topographical maps, DTM's, geological maps and aeromagnetic data, if available, can be used to produce a crude lineation interpretation. This, together with planned or existing locations of mine processes, is usually enough for the design of ground geophysical survey.

The best results are normally received using different methods simultaneously, as different methods have their own strengths and weaknesses. Simultaneous interpretation of all data, *e.g.* geophysics, drilling, maps and DTM, is needed for high enough certainty.

2.1 Geophysical methods

In this guide, geophysical methods are discussed only briefly and the often complex theory behind the methods is mostly left out. There are a countless number of introductory books and publications where these commonly used methods, and their backgrounds, are discussed in great detail. To name a few, Telford (1990), Sharma (1997), Benson (2005), Reynolds (2011) and Parasnis (2012) are all commonly cited introductory books dealing with applied and/or environmental geophysics. In Finnish, Peltoniemi (1988) is still a relevant and excellent book, despite its age.

2.1.1 Electromagnetic methods

2.1.1.1 Ground penetrating radar (GPR)

Ground penetrating radar (GPR, cf. *e.g.* Annan & Davis, 1976, Daniels *et al.*, 1988, Davis & Annan, 1989 and Neal 2004) is a high-resolution, electromagnetic method for the study of shallow subsurface. The method is most commonly based on the transmission of high frequency (usually 25-1000 MHz) electromagnetic pulses to the subsurface and recording the amplitude and two-way travel time of the reflected signal, while towing the transmitter and receiver antennae on the survey line (common offset method). The final result is a high-resolution cross section of changes in electrical properties, which reflect the changes in the physical properties of the subsurface.

The GPR method is at its best in low electrical conductivity sediments, where depth penetration up to 50 meters with low frequency antennae can be received. The capability of GPR in bedrock hydrogeological studies depends in many cases on the thickness and electrical properties of the overburden. If the bedrock is reached with the GPR, the horizontal and sub-horizontal structures are easiest to detect. Vertical structures can be difficult for the common offset configuration, but the main hydrogeologically interesting structures, *i.e.* fracture zones can be interpreted with relative certainty.

2.1.1.2 GEM-2

GEM-2 is a lightweight low induction frequency domain electromagnetic system using the frequency band of 300 - 96000 Hz. Three to 10 frequencies can be measured at the same time. The equipment was introduced in 1996 (Won et al. 1996). Detailed brochures of the equipment can be found in Geophex Ltd. (2016). Field testing has been done by for example Lerssi et al. (2016), where GEM-2 was used for mapping the electrical conductivity of uppermost ground surface. The system has been found to be especially suitable for fast mapping, for example mapping contaminated water paths and plumes in shallow subsurface.

The unit uses the pulse-width modulation technique to transmit and receive any digitally-synthesized waveform (Won et al. 1996, Witten et al. 1997, Huang et al. 2003 and Geophex Ltd. 2016). GEM-2 has the separation of 1.66 m between the transmitter and receiver. Depth of exploration is about 10 m depending on ground conductivity, target volume and ambient electromagnetic noise. Station spacing along the line is about 10 cm at normal walking speed. Sensor is controlled and readings are stored into the handheld socket mobile computer (Somo 655) using Bluetooth. Similarly, location data can be transferred from a GPS-device wirelessly, minimizing the need for cables during the survey. Readings are exported as widely accepted comma-separated values (csv) -files, from which they can easily be imported to most processing and interpretation software, such as Oasis Montaj.

2.1.1.3 Very Low Frequency – Resistivity (VLF-R)

Very Low Frequency – Resistivity (VLF-R) is an electromagnetic (EM) method that uses a very low frequency electromagnetic field produced by a faraway radio transmitter or a more local portable transmitter to detect changes in resistivity of the ground. The method has been traditionally used to detect ore bodies, but have been applied successfully into finding groundwater containing fracture zones from uniform bedrock (see for example Lindsberg 2008). It can also be used to estimate the sediment thickness on top of fractured bedrock (Müllern & Eriksson 1982).

In case of stationary antennas the distance to the transmitter can be thousands of kilometres. The long distance makes the electromagnetic wave act like a plane wave, which makes the primary field in the survey area uniform and distortions, like edge effects, are rarely observed. VLF-R is a variant of the basic VLF method. In the variant, the quotient between the electric and the magnetic field is measured. Since the primary field is a plane wave, the quotient is proportional to the apparent resistivity and the phase between the fields (Cagniard 1953, Hjelt et al. 1990). Measurements can be done in either E-plane or H-plane. In the horizontal E-plane the electric field is parallel to the geological dip to which the magnetic field is perpendicular.

The results usually show good depth penetration. Depth that can be imaged is largely controlled by soil moisture, but on dry soil the depth can reach down to 100 meters. The method is also fairly affordable and the field measurements are fast to conduct, but require at least two persons. As an additional benefit, magnetic field can be measured by one of the operators simultaneously with the VLF-R measurements, with no extra costs or effort (Turunen 2008). Traditionally, Canadian Geonics Ltd.'s EM16R has been by far the most commonly used VLF receiver in Finnish geophysical studies (Hjelt et al. 1990), but commercial systems are also available from other manufacturers, such as IRIS Instruments. Further, Finnish measurements have typically used station DHO38 (23.4 kHz), located in northern Germany, as their source field. A two-layer inversion is routinely employed while interpreting the data in addition to traditional resistivity and phase contour maps.

As mentioned earlier, instead of using a VLF signal from a faraway radio station, the VLF signal can also be produced in-situ with a portable transmitter. This is in many ways advantageous, as the stationary transmitters that are operational (many of them do not transmit regularly or the signal might be weak) might not be orientated optimally (i.e. parallel to the geological strike) to study the EM anomalies (Mursu 1991). Also, a primary wave from a single transmitter can only be used to observe anomalies that are at maximum 45 degrees from the direction of the primary wave (Hayles & Sinha 1986, Mursu 1991). This means that to observe all anomalies with different strikes, at least two perpendicular VLF signals must be analysed. A portable VLF source antenna can be orientated optimally to the studied structures and the orientation can be changed if needed. Downsides to using portable transmitters include added workload, as the long and cumbersome transmitter has to be laid out, and added weakness to disturbances, such as phone- and power lines, which can affect the accuracy of the measurements (Tilsley 1976, Hayles & Sinha 1986). One commonly used VLF transmitter is the TX27 by Canadian Geonics Ltd.

2.1.2 Direct current resistivity methods

2.1.2.1 Electrical resistivity tomography (ERT)

Electrical resistivity tomography is a direct current (DC) method. The purpose of the measurements is to define conductivity distribution of the earth. The conductivity can be estimated on the basis of resistivity measurements. At minimum, the measurement device consists of a pair of current electrodes, used to induce electric current to the ground, and a pair of potential electrodes that are used to detect the current formed by the current electrodes (Fig. 1). The potential difference between the electrodes is affected by electrical properties of the subsurface and measurement geometry.

The electrodes can be set to different arrays, which have their own strengths and weaknesses. Common arrays include Wenner, Schlumberger and dipole-dipole (Figure 1). At GTK a multiple gradient electrode array is quite commonly used. For example Furman et al. (2003), Dahlin & Zhou (2004) and Martorana et al. (2017) compare different arrays extensively. Measurements can also be done downhole by suspending the electrodes into wells or boreholes.

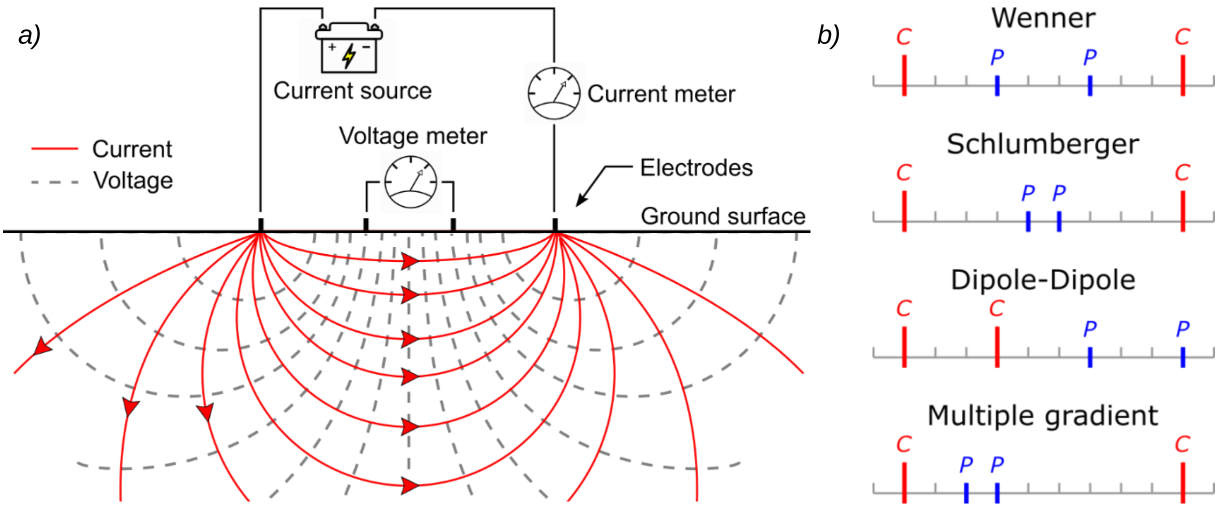


Figure 1. a) Basic working principle behind electrical resistivity tomography measurements. The array in which the electrodes are set is Wenner array, which consists of four equally spaced electrodes. Current is induced through the two outer electrodes and potential is measured between the two inner electrodes. Figure after Sharma (1997) and Muchingami et al. (2013). b) Comparison of different arrays used for ERT measurements. C = current, P = potential.

Depth penetration and measurement resolution can be controlled with electrode interval and length of the measurement section. The depth penetration of the measurements is roughly 1/5th of the total length of measurement section. For example with 56 electrodes and 5 meter electrode interval, the measurement section will be 275 m long. This will yield roughly 50 meters of depth penetration. With longer electrode intervals, deeper profiles with lower resolution can be made (Suomen vesiyhdistys 2005). However, it should be noted that the resolution of the method degrades fast with increasing depth, and that thin layers might not be detected if too coarse resolution is used. With modern automatic equipment, a two person team can measure between 500-1000m in a day (Suomen vesiyhdistys 2005).

One of the biggest advantages of ERT compared to electromagnetic methods like VLF-R is its good suitability for areas where also the top layer of soil consists of electrically well conducting sediment (such as clay). It is also less sensitive to disturbances such as power lines. Depending on the properties of the unlithified sediments, it might be even possible to estimate hydraulic conductivity of porous- and fractured aquifers from resistivity values, as generally the relationship between hydraulic conductivity and resistivity is directly proportional to grain size, yet, the method holds a lot of uncertainty and the results may vary grossly between aquifers (Mazac et al. 1990). Problems with the method are most commonly encountered if the contact between electrodes and ground is poor due to for example very dry or frozen soil (Benson 2005).

2.1.2.2 Resistivity fork

The resistivity fork is built in a Wenner electrode array and electrode spikes (Fig. 2). Resistivity fork gives resistivity estimate for the uppermost layer of the overburden, mainly for a depth range of 0 - 30 cm. The most reliable measurement range of apparatus is from 5 to 5000 ohmm (Puranen et al. 1999). The device is most useful when studying the resistivity of the most upper part of overburden, before actual ground geophysical surveys.

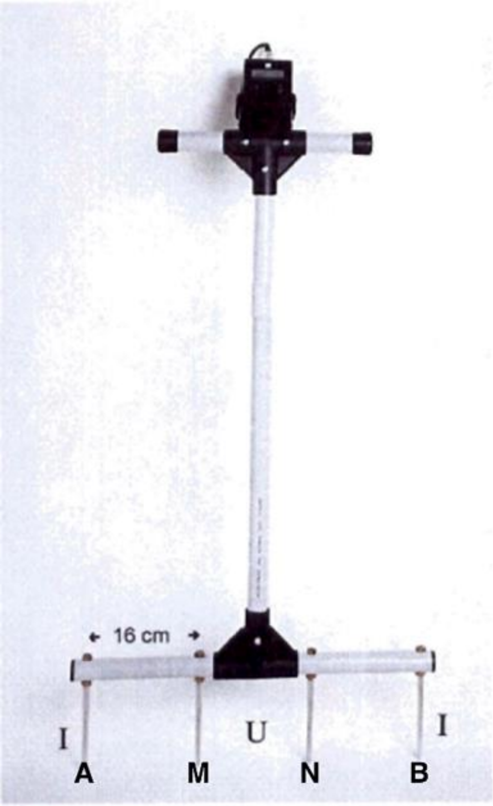


Figure 2. Resistivity fork. The array is 48 cm long, with 11 cm spikes. Note the equal spacing between the AMN and B electrodes, a key feature for the Wenner array. Current is injected through the outer electrodes while potential is measured between the inner electrodes. Operational principle of the device is essentially the same as shown in Figure 1.

2.1.3 Refraction seismic surveying

Many useful aquifer and subsoil properties can be defined with refraction seismic refraction surveys. These include groundwater level, the thickness and material of sediment layers and the depth and uniformity of the bedrock surface. The method is based on how induced (or natural) seismic waves refract from the boundaries of materials having different elastic properties. While encountering a boundary between two layers having different elastic properties part of the seismic waves are refracted back towards the ground surface (Fig. 3). This refraction follows the basic laws of optical refraction (known as the Snell's law). Depth of the different layers can be calculated from the travel times of the seismic waves and the structure of the subsoil can thus be imaged.

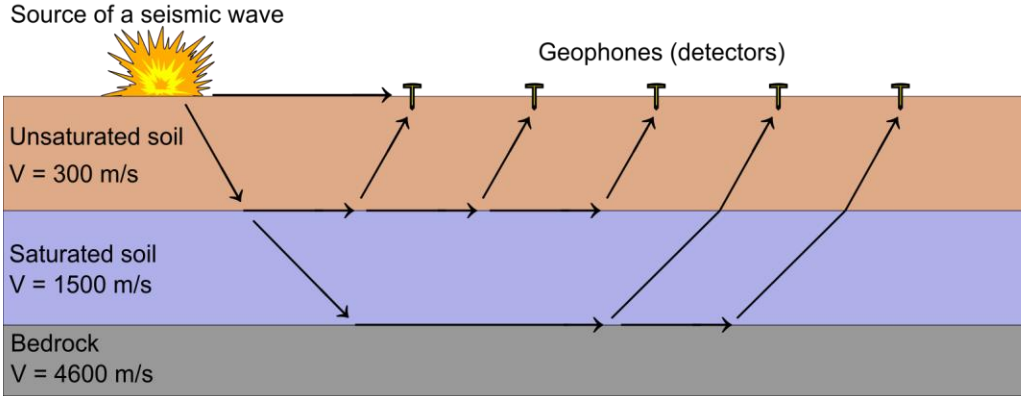


Figure 3. Basic principle of seismic refraction measurements.

In suitable conditions the method can be cost efficient, but at the same time the measurements can also be fairly time consuming as installing geophones can take a lot of time (Steeples & Miller 1990). Measurement speed typically varies between 400-1500 meters a day, depending on the size of the measurement team (usually 2 to 4 persons) and the chosen geophone interval (2.5 to 10 meters) (Suomen vesiyhdistys 2005). Additionally, if explosives are used to produce the signal (which is often the best choice for mapping deep layers and the bedrock surface, as explosives produce a strong and sharp signal), measurements and operators might require special permits or licenses. The method is also fairly weak to small ground vibrations, which might mean that at some sites getting an undisturbed signal might be difficult (e.g. due to machines at mine sites or waves at a shore) (Benson 2005). Still, the biggest pitfall of the method is seen with so called low velocity- and hidden layers. Typically, wave transmission velocities increase with depth (Fig. 3). If a low velocity layer exists below a higher velocity layer, the low velocity layers cannot be detected. As an example, if a sand aquifer underlies a compact clay unit, the aquifer will not be detected. Problems are also encountered with thin-, dipping- and discontinuous layers, which can be very hard to detect or correctly interpret (Steeples & Miller 1990). Error in the measurements is typically ±1 m when the total thickness of the overburden is less than 10 meters. The rather high error is partially explained by a higher risk of hidden and thin layers. If the overburden is more than 10 meters thick, the error margin is typically around 10% (Suomen vesiyhdistys 2005). Generally, also the need for special equipment and trained technicians has limited the use of the method in groundwater studies (Todd & Mays 2005).

2.1.4 Gravity method

Due to earth's stable gravitational field, a fairly universal gravitational acceleration of $6.67259 * 10^{-11}\ m^3kg^{-1}s^{-2}$ exists all around the world. This gravitational acceleration has been observed to vary by only very small amounts (100 ppm). While the local background value for the gravitational acceleration is known (by using absolute methods like the Pendulum or Weight drop, or relative methods like the Spring-mass method) inconsistencies, like changes in ground materials density, can be observed as fluctuations in the gravitational field.

The gravity method has been quite commonly used to locate groundwater resources located on top of bedrock depressions, as the large density difference between sediment and bedrock makes interpretation easy. The method cannot be used to identify individual sediment layers or the groundwater surface and is thus best suited for preliminary studies which are then enhanced with other methods (Suomen vesiyhdistys 2005).

Multiple factors like elevation above reference level, local topographic changes, earth tides and variations in the subsurface density between the reference and measurement points affect the results. Some of these unwanted effects can be compensated in post-processing, but still some assumptions about the composition of the subsurface have to be made. These assumptions and corrections can be the greatest drawback of the gravity method and can severely affect interpretation of the results if done incorrectly. For example, faulty corrections can easily have more impact on the gravity field than a buried hazardous waste dump (Vogelsang 2012).

The method isn't commonly used to locate groundwater resources from bedrock inconsistencies, yet Murty & Raghavan (2002) successfully used the method to locate groundwater bearing fractures in granitic bedrock. In environmental settings, use of the method is often limited by the high cost of the equipment and the skill and accuracy needed to detect very small gravity anomalies. Also, in order to locate the small anomalies, a dense measurement grid is needed, which further adds costs (Vogelsang 2012). On the upside, the measurements are fairly fast to conduct. In a day, a three person team can typically collect 3 kilometres of gravimeter data with a 20 meter measurement interval (Suomen vesiyhdistys 2005).

2.1.5 Interpretation of geophysical data in the context of bedrock hydrogeology

Before data from geophysical measurements can be interpreted a pre-processing and/or inverse modelling need to be performed. Pre-processing is a necessary step for all geophysical methods presented in this report, excluding the resistivity fork. Inverse modelling is performed for VLF-R, refraction seismic, ERT and gravity methods to create a cross section of subsurface. For GPR method, the different inversion methods are available in pre-processing stage, but to produce the cross section, inversion is not needed, only data enhancement.

In hydrogeological studies focusing on the bedrock, fracture zones are the most important macro-scale features to identify. Refraction seismic, ERT, GPR and VLF-R methods can be utilized directly to recognize the fracture zones. In refraction seismic method the direct recognition is noticed through lowered seismic velocity in bedrock. In GPR method the recognition can be made from the reflection patterns in the bedrock. In ERT and VLF-R method the vertical anomalies with usually higher electrical conductivity can be recognized. In fracture zones the higher electrical conductivity is caused by the groundwater.

All the above mentioned methods give also indication of overburden thickness and bedrock level. This combined with map and elevation data can also give hints of fracture zones. In recently glaciated areas the fracture zones are more easily eroded than unfractured bedrock causing bedrock valleys. If these valleys are filled with overburden, they cannot be identified from elevation data alone.

Combining different methodologies an interpretation of fracture zones can be made. Geophysical interpretation is always ambiguous, and several methods and interpretation techniques should be combined to make the results more precise. For example in ERT and VLF-R methods a similar modelling result, resembling fracture zone, can be achieved from many geological settings, such as bedrock lithology variations.

2.2 Investigating the condition of existing boreholes and observation wells

Practice has shown, that in many cases the state and even the location of boreholes and groundwater observation wells is poorly known. This is, of course, mostly a problem with boreholes and observation wells excluded from the active monitoring program of mines. In the worst case well logs (containing e.g. well location coordinates and depth) might not be available or have over time scattered to different operators (e.g. the mining company itself, consulting agencies, officials, etc.). The existing well logs might also be partially incomplete or contain information that is no longer valid. Thus, if possible, the condition of wells and boreholes of which reliable information cannot be acquired, should always be checked onsite. Neglected wells and especially old boreholes can become clogged over time or even get completely filled and buried. Water level can also be quickly measured during the same check-up. It might also be beneficial to check for obstructions and the total well depth with a cheap dummy slug (e.g. a sand filled PVC cylinder hanging from a wire), especially if large and expensive downhole equipment (e.g. packer- and pumping test) are planned to be used on the well (Fig. 4). On the other hand, this dummy-slugging will unwelcomely disturb water in the well or borehole and so it could be done later if the plan is to start from a simple electrical conductivity profiling or slug tests.

Of course, pre-investigating the condition of wells and boreholes is not always possible or viable because of for example time constrains or long distances to the study area. However, background information gathered during this step can save a lot of time during the actual field campaign as the sites are well known and the focus can be on the measurements themselves, not on preparatory steps or hauling equipment ineffectively from one borehole to other.

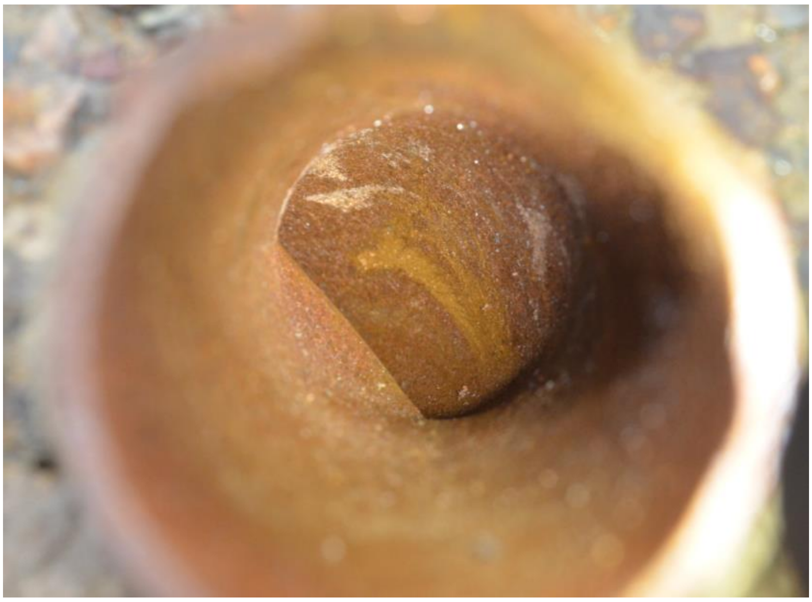


Figure 4. The metal casing of an old borehole has twisted due to land recontouring. The kink prevented using large downhole equipment, but allowed for the use of smaller devices such as CTD -sondes.

2.2.1 Hydrogeological core logging

After the core locations are designed, there are, in principle, two different methods for creating a hole in the bedrock, drilling and boring. Boring is many times preferred due to lower costs, but in boring lots of precious hydrogeological data is lost. Many times, the cost savings in boring in comparison to drilling are misplaced if more elaborate hydrogeological methods are needed at a later stage. Thus, drilling is the suggested method to make an observation hole in the ground, because the drill core can be preserved and logged. Also it need to be bear in mind that existing drill holes and their cores, made for exploration purposes, can also be used in hydrogeological interpretation.

Core logging can give invaluable information of the hydrogeological conditions. The logs can give information of cracks, joints and dykes that may have importance on the groundwater flow. The cores cannot be used for hydrogeological analyses, but at the minimum they can direct the rather expensive hydraulic testing methods to most probable positions. The most used method to describe the fracturing in rock is the Rock Quality Designation (RQD, Deere, 1964).

3 HYDRAULIC TESTING

Understanding the subsoil, its properties and the water moving through it, is always a challenging task. The wells and boreholes, through which we have to observe groundwater, are merely pathways to its study. In mining environments, gaining knowledge of the performance (e.g. yield) of the well/borehole itself, is often only a secondary objective.

Drilling and installing purpose built groundwater observation wells are expensive. However, existing mines always have exploration boreholes that have originally been drilled to delineate and study the extent and grade of the ore resource. These boreholes are a huge and mostly underused resource in groundwater studies. Unfortunately, utilizing them for hydrogeological studies is not an easy task. Unlike purpose built observation wells, boreholes tend to be dipping, curving and usually do not contain casings after the hole has reached the bedrock, making the walls in some cases very rugged and even out-of-round. In Finland, boreholes and observation wells are also generally quite narrow compared to for example North-America. This generally makes methods, and especially those that require multiple downhole components, even harder to use.

Screened wells, and especially open boreholes, are also huge pathways for vertical groundwater flow. Aperture of a natural fracture might be in the range of few millimetres and groundwater flow through them might account to just a meter or two a year. Boreholes, cutting through this system, have their diameters measured in centimetres and provide an unrestricted vertical flow path for groundwater. Even tiny vertical hydraulic gradients can induce vertical groundwater flow, which can affect all measurements done in a borehole. In the worst case this effect can be large enough to render all measurements and samples almost useless (Elci et al. 2001). In larger diameter wells and higher water temperatures, also convection can affect the measurement results. Drudy et al. (1984) draw critical gradients for convection inside boreholes, which are based on well diameter, water temperature and temperature gradient of the earth. Convection has been found to be especially prominent in wells with long open- or screened sections (Reilly et al. 1989). In Finland, convection inside the boreholes can be expected to be negligible due to low groundwater temperatures and lower than average temperature gradient of ground (Järvimäki & Puranen 1979). The negative impacts of vertical groundwater flow can be minimized by sealing of individual sections of a borehole (see Chapter 3.6), but use of such methods substantially increases complexity of any downhole measurements.

Different methods available for a groundwater study could be ordered and presented in countless different ways. In this report, we start with relatively simple and quick methods and move towards more advanced and often also more challenging ones. Also, in a short guidebook all methods and their variations cannot possibly be presented. Instead, the focus is on methods on which GTK already previously had knowledge on, and especially on methods that were tested during the KaivosVV –project. This means techniques that have reasonable initial and operational costs, do not require months of time to produce usable data and are relatively portable. Emphasis will be on the up- and downsides of each method and on the most common pitfalls that the user might face, without wondering too far into background theory.

Many of the same methods are used to estimate properties of shallow porous medium aquifers and some, like fluid electrical conductivity logging introduced in Chapter 3.2, are even used with surface water bodies. Still, in the following sub-chapters we try to focus especially on methods that are suitable for groundwater measurements in fractured bedrock. Essentially we try to answer why a mine operator should choose one test over another and in which order the tests should be conducted to provide actually useful data (Fig. 5).

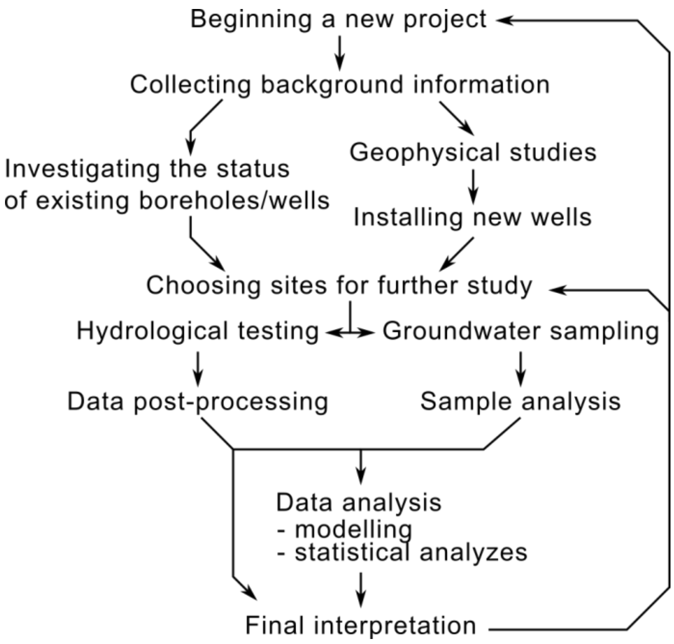


Figure 5. A very indicative flow-diagram for groundwater studies in mining environments.

3.1 Manual monitoring of hydraulic head

What for?

Checking and monitoring changes in hydraulic head (i.e. “groundwater level”) by manual measurements.

Pros

- ✓ Quick, cheap & easy.
- ✓ Can be used to monitor big changes, such as dam and lining breaches at tailings ponds.
- ✓ Useful in general interpretation of groundwater flow direction

Cons

- ✗ Without other measurements, uses are limited.
- ✗ Electric water level meters are sometimes prone to alarm from wet pipe walls etc. Care needs to be taken that the actual groundwater table is being measured.

How to?

Measurement steps for manual measurements using a water level meter

1. Turn on the instrument and lower the measurement tape into the well or borehole.
2. Lower the instrument until groundwater surface is reached (a sound or a light comes on).
3. Record the head and retrieve the instrument.

Checklist for the field

- ☐ Groundwater level meter or a small heavy slug hanging from a string.
- ☐ Pen and notebook.

Measuring the hydraulic- or piezometric head is the most common preliminary step before any other measurements or collecting water samples. However, on active long term monitoring, more and more operators opt for automatic head measurements using autonomous systems (discussed in Chapter 5.2), as manual measurements are very labour intensive and can quickly become more expensive than autonomous systems. Manual monitoring also lacks the capability to produce real-time results or continuous records with short measurement intervals.

In unconfined aquifers the hydraulic head is equal with the level of the groundwater table in atmospheric pressure. In confined aquifers, where the top of the aquifer is not open to the atmosphere and is blocked by semi-permeable layers such as clays or tills, the hydraulic head doesn’t necessarily match the groundwater level (called also potentiometric head). In these cases the head can raise above the ground surface to form an artesian well.

3.2 Fluid electrical conductivity and temperature logging

What for?

Quickly and affordably assessing the chemical quality and/or flow regime in a well or borehole. Narrowing down interesting sites for further study.

Pros

- ✓ Quick measurements.
- ✓ Reasonable costs.
- ✓ Simple equipment and operation.
- ✓ Small instrument size (especially with CDT-sondes) makes equipment jams less likely in boreholes and transporting the equipment more manageable.

Cons

- ✗ Measures the properties of the water in the well/borehole, which might differ from the groundwater in the surrounding medium.
- ✗ Can be used to indicate groundwater flow, but cannot be used to measure it.
- ✗ Current equipment is mostly designed for monitoring, not profiling.

How to?

Measurement steps

1. Calibrate the instrument if needed.
2. Measure groundwater head.
3. Connect the sonde to a laptop or central unit and start the measurement.
4. Lower the sonde into the well or borehole with a slow and even pace (~2m/min) until the well bottom or predefined depth is reached.
5. Repeat the measurement from bottom to top or end the measurement and retrieve the instrument.

Checklist for the field

- ☐ Groundwater level measurement device.
- ☐ Multiparameter- or CDT-sonde
- ☐ Calibration liquids & equipment (if calibration is not performed in the laboratory).
- ☐ Laptop or central unit.
- ☐ Connection cable to connect the sonde to laptop or central unit.
- ☐ Spool with suitable length of wire or cable.
- ☐ Free-wheel to ease the deployment.

Fluid electrical conductivity (FEC) logging is a simple method that can be quickly used to produce data from wells or boreholes. In mining or other environments with multiple wells and boreholes, it is most useful as a tool for narrowing down the sites for more advanced studies such as pumping tests or flowing fluid electric conductivity logging. The method is gaining popularity worldwide in all kinds of groundwater studies (In-Situ Inc. 2017).

In the test, a conductivity sensor is used to make a vertical conductivity profile through the length of the well or borehole. From this profile, horizons of high or low EC can be seen. These horizons correlate with discontinuities of the bedrock that transport water into the well with different ion content compared to the surrounding well water. Simultaneously with the EC profile, a temperature profile is created. Temperature logging has a fairly long history of use in characterization of groundwater flow into boreholes (e.g. Drudy et al. 1984 and Conaway 1987), and the results have also been used for many other purposes, even mineral exploration (see Mwenifumbo 1993). However, in shallow boreholes typical for Finnish mines (< 300m), we observed EC to be a more useful parameter as water temperature variation along the boreholes was in many cases very small.

Unlike many other measurement techniques, the aim during FEC -logging is to disturb groundwater inside the well as little as possible. Because of this, the test should be conducted before anything else is done on the well. Thus, the most important and accurate logging is the first one moving downwards (Keys 1990). To further avoid mixing water and different horizons in the borehole, the probe should be lowered at a slow and constant velocity, even 1-2 m/min, but depending on the borehole diameter higher speeds can also be used. For example Foote et al. (1998) conducted an FEC -logging in an existing water supply well at a speed of 5.5 m/min. In a measuring campaign that only includes few shallow wells, logging speed doesn’t have huge importance as even with slow speeds the measurements can be conducted usually in a few hours. However, if the plan is to log tens of boreholes, logging speed starts to have a significant impact on the time required for field work. Negative impacts of higher logging velocity were tested in a single 250 m long, slowly overflowing borehole, which was logged at about 4.5 m/min. The sensor (In-Situ Aqua TROLL® 200) was set to its minimum 2 second measurement interval. In the produced FEC-log, larger changes in EC are still clearly visible, but the EC-graph seems to have minor (<0.5 mS/m) stuttering possibly due to the higher logging speed. Based on the single test it can be concluded that higher (>2 m/min) logging speeds are viable in cases where many boreholes need to be quickly narrowed down for major changes in EC. In-Situ Inc. (2017) also reported that some of their clients log wells with well-known EC-profiles at higher speeds, where the EC has before been observed to be stable, but switch to lower speeds at zones where the EC varies or changes.

There are some devices on the market, specifically designed for EC and temperature logging in boreholes (e.g. Solinst Model 107 TLC Meter, In Situ Inc. Rugged Conductivity/Level/Temperature Meter and OTT Hydromet KL010 TCM Contact Gauge). In this project, we tried making the profiles with more common CDT -probes originally intended for long term autonomous water conductivity and temperature monitoring. Unfortunately, most CDT -probes are not meant to be used in vertical profiling, which sets additional challenges. For example, on the devices the conductivity sensor is commonly fitted vertically and the connectors in the probes might not be intended for constant attachments and disconnects. Profiles can also be made with multiparameter sondes that are capable of measuring many other variables simultaneously (e.g. pH, dissolved oxygen and redox potential). These devices are also more suited for single-event measurements. The biggest downside of using a multiparameter sonde is perhaps their larger size, which means that they might not be able to fit all boreholes and might disturb groundwater more. The extra sensors (if used) also add need for additional calibration and maintenance, thus adding costs.

3.3 Slug test

What for?

Quickly and affordably estimating hydraulic properties in a well or a borehole. Good option especially for shallow aquifers and narrowing down the sites for more comprehensive pumping- and packer tests.

Pros

- ✓ No added or removed water.
- ✓ Relatively quick measurements (often less than an hour, compared to +24h needed for pumping tests).
- ✓ Affordable.
- ✓ Simple equipment and operation.
- ✓ Relatively small amount of light weight equipment, making transportation manageable.

Cons

- ✗ Quantifies the whole borehole/screened section of the well. Cannot be used for characterizing individual zones or layers.
- ✗ Essentially a method for characterizing the properties of a single well/borehole. Vertical reach is very limited outside the immediate surroundings of the well, making reliable interpolation between wells difficult.
- ✗ Interpreting the results and choosing the right mathematical model requires some expertise.

How to?

Measurement steps

1. Measure groundwater head.
2. Lower a water lever sensor into the well, while making sure it is deep enough not to intervene with the submerged slug later on.
3. Connect the sonde to a laptop or central unit and start recording the water level.
4. Drop a heavy slug into the well and make sure it is completely submerged. Observe and wait as the groundwater level recovers (falling head).
5. As quickly as you can, pull the submerged slug above the water table and then remove the slug from the well. Observe and wait as the groundwater level recovers (rising head).
6. Repeat measurements 3 times.
7. Stop the water level recording and remove the sonde from the well.

Checklist for the field

- ☐ Groundwater level measurement device.
- ☐ Multiparameter-, water level- or CDT-sonde.
- ☐ Laptop or central unit which can show changes in groundwater head.
- ☐ Connection cable to connect the sonde to laptop or central unit.
- ☐ Spool with wire or a cable to suspend the water level measurement device.
- ☐ Spare batteries.
- ☐ A heavy, solid slug with known volume or a bailer.
- ☐ Wire or rope to suspend the slug from.

A slug test is the most commonly used method for in-situ estimation of hydraulic conductivity (Butler 1997). The test can be conducted in many different ways. Most commonly, a heavy slug with a known volume is either dropped into (falling head), or pulled from (rising head) the groundwater inside an observation well (Fig. 6). The water level in the well instantly rises or falls to accommodate the volume of the slug. After the change, the water table starts to return towards equilibrium and the time it takes for the head to reach equilibrium is measured. For example, Hvorslev (1951), Cooper et al. (1967) and Bouwer & Rice (1976) have published commonly used formulas to calculate hydraulic properties (such as hydraulic conductivity and permeability) from slug test results. Both rising- and falling head methods should provide similar interpretations, yet some (see for example Weight 2008) have argued that rising head method should be preferred, as the testing happens completely in the phreatic zone (i.e. water level is not raised to layers that are normally dry).

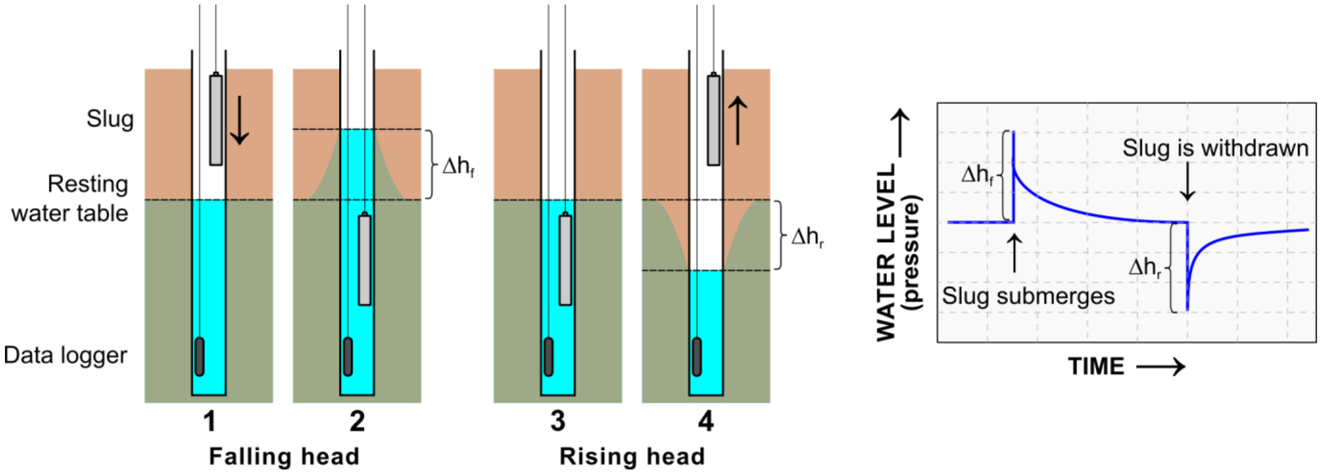


Figure 6. The basic principle of falling- and rising head slug tests. In the falling head test, a slug with a known volume is dropped below the water table (1). This causes the water table to rise according to the volume of the slug (2). After the slug submerges the water table starts to stabilize again. The time it takes for the water table to reach an equilibrium is measured. In the rising head test (3 & 4), the same process is simply reversed.

Slug test has some advantages over the more comprehensive pumping test, mainly its simplicity, speed (minutes instead of hours, days or months needed for pumping tests) and the fact that it can be conducted without adding or removing any water (highly advantageous if dealing with contaminated groundwater). At sites where the transmissivity of the aquifer is too low for a proper pumping test (like at bedrock monitoring wells or aquitards), a slug test might be the easiest option to collect data. On the other hand, a slug test realistically evaluates only a small portion of the aquifer adjacent to the well. Thus also the results can be easily compromised if, for example, the screen of the well is blocked by fine sediment. The method also only gives a general picture of the hydraulic conductivity in the whole well, and cannot be used to identify different zones of hydraulic conductivity. Butler & Healey (1998) also found that K values obtained from slug tests are on average lower than those from pumping test. They estimated this to be mainly caused by slug tests being more impacted by poor well development (e.g. remnant drilling fluids and debris that disturb measurements), and to a lesser degree, vertical anisotropy and variance in hydraulic conductivity.

Equipment wise, the test is very simple and versatile. The most common option is to use a heavy solid slug e.g. a metal cylinder or a sand filled plastic pipe. Optionally, the test can also be done by adding or removing water from the well. This can be easily achieved with a tube bailer. If a single bailer isn't large enough to cause a sufficient drawdown, the bailer can also be retrofitted with extendable pipe to grow its internal volume. The empty pipe is easy to carry in the field, compared to a heavy solid slug. Alternatively, water can also be added or removed using pumps or the water level can be driven down using pneumatics. However, in most cases simple solutions are preferred, as removing water from a well quickly enough with pumps is hard and the pneumatic systems often add unwelcome complexity. Regardless to the slug choice, water level is most commonly measured by monitoring changes in water pressure, as this is much easier than directly measuring the water level during the test. Butler et al. (1996) have formed a testing protocol for slug tests. When followed, the protocol should help novice users in the field and reduce errors in the result estimations. Also for example Butler et al. (1998) and Weight (2008) provide a lot of general tips for conducting slug tests.

3.4 Pumping tests

What for?

Comprehensively defining the hydraulic properties of an aquifer.

Pros

- ✓ Can be used to accurately estimate most hydraulic properties of a formation.
- ✓ Reasonable equipment costs in shallow depths.
- ✓ Fairly simple equipment and operation.

Cons

- ✗ Testing can last from hours to days to months, usually lasting at least for 24 hours.
- ✗ Quite a lot of equipment is needed in the field and the pumping requires electricity.
- ✗ Water from the well needs to be discarded far enough from the aquifer to avoid re-infiltration. Contaminated water needs to be treated.
- ✗ Water is drawn from the whole length of a borehole or from the screened section of a well. Thus, estimating vertical variation in K is not usually possible without packers.
- ✗ Well screens can become clogged during testing, which can lower K-values compared to reality.

How to?

Measurement steps (Constant head test)

1. Measure groundwater head in the test- and surrounding observation wells/boreholes.
2. Install water level sensors to surrounding wells/boreholes and optimally also into the test well/borehole itself.
3. Connect all required hoses and cables and lower the pump into planned depth.
4. Begin pumping. Monitor the pumped water volumes and the depression cone seen in the adjacent water level logs. Adjust pumping rate until a steady state is reached.
5. Hold the steady state for at least 10 hours. Monitor water level periodically.
6. End pumping and monitor water level transgression.
7. Retrieve equipment.

Checklist for the field

- ☐ Pump with sufficient capacity for required depths and water volumes.
- ☐ Hose reaching from the pipe to ground surface.
- ☐ Pump control unit.
- ☐ Power source (mains, a battery or a generator).
- ☐ Laptop or central unit.
- ☐ Water level sensor.
- ☐ Connection cable to connect the sensor to a laptop or central unit.
- ☐ Spool with suitable length of wire or cable for the sensor.
- ☐ Water flowmeter or a bucket to measure water volumes.
- ☐ Step watch.

Pumping tests (also referred to as *aquifer tests*) are probably the most comprehensive of the commonly used methods for estimating basic hydraulic properties and dimensions of an aquifer. In the tests, the aquifer is stressed by pumping water out from a test well, while simultaneously water level is being monitored in surrounding observation wells, and optimally also in the test well itself. The tests can be conducted in several different ways, most common of which are:

- Constant-Rate test
  - The most common form of pumping test. The control well is pumped at a constant rate while drawdown is measured in one or more surrounding observation wells. This multi-well test can be used to estimate multiple hydraulic properties of an aquifer including transmissivity, hydraulic conductivity and storage coefficient in between the wells (i.e. on a relatively large area).

- Step-Drawdown test
  - During the test, water discharge rate is increased in steps from smaller to higher. Each step usually has the same duration (~ 30-120 min) (Kruseman et al. 1994). The test is mostly suited for single-well study and can be especially useful if well performance (e.g. well loss and well efficiency) is of particular interest.

- Constant-Head test
  - Water is pumped with a constant rate and the pumping rate is only slightly adjusted to reach a steady state where the groundwater surface stays constant. Hydraulic properties are calculated based on achieved drawdown (m), time and Q (pumping rate). The method is well suited for multi-well approach, but a good, stable drawdown might be difficult to achieve in poor permeability layers. More rarely used than Constant-Rate test.

- Recovery test
  - Conducted at the end of any of the other pumping tests, but often interpreted separately. Water level recovery is monitored in the surrounding wells and optimally also in the control well itself. Can be used to provide additional data for estimating aquifer properties.

Aquifer properties are estimated from the pumping test results by fitting drawdown data from observation wells to mathematical models by curve matching. A model is chosen based on the aquifer and test settings. Well known and commonly used models include those by Theis (1935), Cooper & Jacob (1946) and Hantush & Jacob (1955). In addition to the drawdown data the pumped water should be systematically sampled and field analysed to see whether the pumping causes changes in the chemical composition of the groundwater.

Pumping tests benefit highly from a multi-well approach where drawdown is monitored in observation wells surrounding the test well. The multi-well approach allows to estimate hydraulic properties in-between the wells, and can so allow reliable extrapolation of hydraulic properties for a larger radius around the test well. However, naturally a multi-well approach requires an existing network of close-by groundwater observation wells. This is rare, and so testing is often limited to the single-well used both for monitoring and pumping, which makes estimating the hydraulic variables hard outside the immediate surroundings of the well.

One downside to pumping tests is that they generally take a lot of time to conduct, and so can severely affect groundwater conditions at the site. Before starting the test it needs to be considered, for example, how lowering the groundwater surface will affect nearby structures or municipal water wells. Other thing to consider is how the large amounts of pumped water are disposed. This might prove problematic especially if dealing with contaminated groundwater. Even with un-contaminated waters, it needs to be made sure that the pumped water doesn't enter back into the same groundwater system as this would make the test results inaccurate. This might mean pumping or transporting the water to a nearby lake or a stream. Further, pump in the test well requires electricity. Producing electricity with a generator is not favourable due to fuel and maintenance costs, along with environmental impacts, but might be the only option at remote sites (Suomen vesiyhdistys 2005).

3.5 Flowing fluid electric conductivity logging

What for?

Relatively affordable and quick method for defining hydraulic properties along the vertical profile of a borehole. An alternative worth considering where packers or advanced flow logging equipment aren't available/viable, and handling large water volumes is not an issue.

Pros

- ✓ Can be used to characterize vertical variations in hydraulic conductivity.
- ✓ Quick measurements.
- ✓ Reasonable costs.
- ✓ Simple equipment and operation.
- ✓ Equipment jams are unlikely.

Cons

- ✗ Requires large volumes (hundreds of litres) of low-EC water.
- ✗ Produces very large volumes of water that needs to be treated/discarded.
- ✗ Interpreting the results and choosing the right mathematical model requires some expertise.
- ✗ Large capacity pumps might be difficult to acquire.
- ✗ Current logging equipment is mostly designed for monitoring, not profiling.

How to?

Measurement steps

1. Lower one pipe or a large diameter hose to the bottom of the borehole and another only slightly below the water level.
2. Start injecting low-EC water to the bottom of the borehole using a high-capacity pump (e.g. ~100 l/min). Remove water from the top of the borehole using another pump operating at the same pumping rate.
3. Monitor EC of the water removed from the well. When the EC-stabilizes, halt the water injection.
4. Begin pumping water from the borehole in a stable pace (e.g. ~10 l/min)
5. Make multiple complete up-down EC –logs along the borehole using a steady pace (~10 m/min)
6. Halt pumping and retrieve all equipment

Checklist for the field

- ☐ Groundwater level measurement device.
- ☐ Multiple well-volumes of low-EC water (e.g. purified water or water of known EC from a nearby surface water body) and a way to transport this water to the test site.
- ☐ Two large capacity pumps.
- ☐ Control unit and electricity source for the pump (portable generator)
- ☐ A long pipe/hose that reaches to the bottom of the borehole.
- ☐ Multiparameter- or CDT-sonde for logging and test monitoring.
- ☐ Calibration liquids & equipment for the EC-sonde.
- ☐ Laptop or central unit for the EC-sonde.
- ☐ Connection cable to connect the sonde to laptop or central unit.
- ☐ Spool with suitable length of wire or cable for the EC-sonde.
- ☐ Free-wheel to ease deployment of hoses and sondes.

The Flowing fluid electric conductivity (FFEC) logging method first introduced by Tsang et al. (1990) and further developed by e.g. Tsang & Doughty (2003) could be considered a more advanced variant of the regular borehole FEC-logging (introduced in Chapter 3.2). At the beginning of the test, water in the borehole is first replaced with low-EC water (e.g. deionized water or clean surface water) by injecting fresh water to the bottom of the borehole and simultaneously removing water from the top of the borehole at same rate as the injection. The borehole flushing is monitored by measuring EC of outflowing water to ensure that all water in the hole is replaced. When this has been achieved, the water injection is halted and multiple electrical conductivity profiles are made over a time period varying from few hours to a couple of days until a steady state in the conductivity has been reached (Fig. 7). During this, the borehole is pumped at slow, constant rate (few litres per minute), and the pumped water volumes as well as the groundwater level from a water level sensor are monitored. The produced logs can be used to evaluate transmissivity and electric conductivity (salinity) along the well profile (Tsang & Doughty 2003). Aquifer properties are estimated from the FFEC test results either by fitting the data to mathematical model such as BORE II -code (Doughty & Tsang 2000) or by inversion mathematics (e.g. Moir et al. 2014).

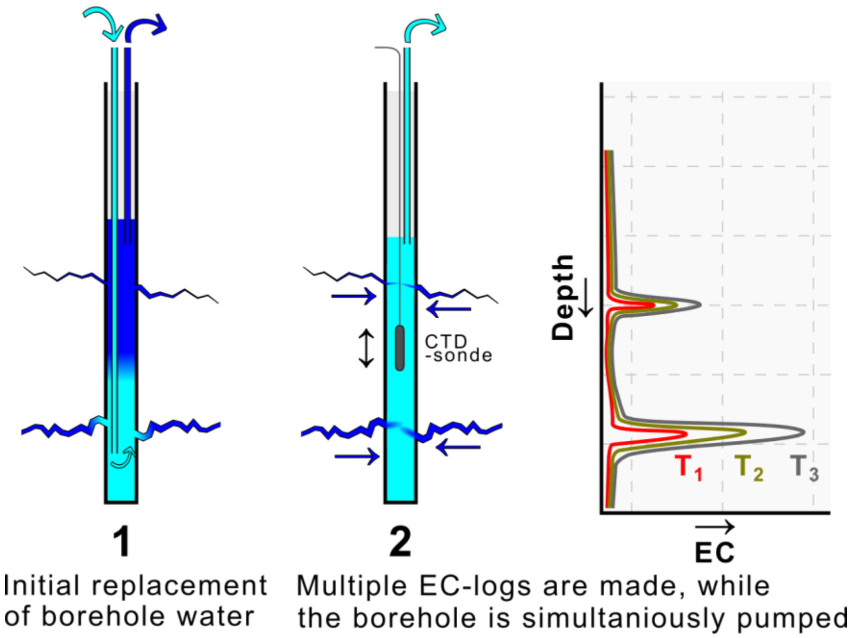


Figure 7. The basic principle of flowing fluid electric conductivity logging (FFEC). First (1), borehole water is replaced with low-EC water by injecting it into the bottom of the well. Simultaneously, borehole water is removed from near the groundwater surface. After the borehole water has been completely replaced, the injection is halted. In the second step (2), the borehole is slowly pumped, while multiple EC-logs are done. Figure modified Tsang et al. (2016).

Overall, the method is very promising and is being actively developed. Tsang & Doughty (2003) recommend it as a much less laborious and time consuming option for pumping tests in packer-isolated intervals. Further to its advantage, the test can be conducted using relatively affordable and readily available equipment. One of the biggest difficulties with the method is the need for large volumes of fresh or even deionized water, which in most cases needs to be transported on site with water tankers. This might be difficult to accomplish, especially in more remote areas. Also, the test produces a lot of excess water from the initial flushing phase and from the active pumping during logging, which might be very problematic especially if dealing with contaminated groundwater. This can be countered by treating the removed water through e.g.

filtering and ion exchange and then re-injecting the water (see Doughty et al. 2005), which on the other hand requires complex and expensive water treatment equipment.

3.6 Packers in hydraulic testing

According to Bishop et al. (1992), significant vertical groundwater flow happens in basically all boreholes. This flow mixes groundwater from different layers or fracture systems and, in worst case, can transfer contaminants from one system to another. To prevent vertical flow or to limit the zone where hydraulic testing is conducted, packers can be used. Packers are essentially expandable rubber plugs, which can be fitted into wells or boreholes. Empty packers are lowered into the well and expanded by water, inert gas or air. This seals the space between the packer and the borehole wall and thus prevents vertical flow through the borehole. In literature, the term *Packer test* is commonly used. In most cases, the term seems to be used to refer to some variation of a pumping test, done in a borehole/well section isolated by packers. However, it is possible to do many kind of hydraulic tests in-between packers – even pressure driven slug tests.

The results can be analysed via the same formulas used to analyse regular pumping- and slug tests. However, choosing the right formula to estimate the hydraulic conductivity in the short, isolated, test section can be difficult and the hydrogeological properties of the study site must be well understood. For example Bliss & Rushton (1984) recommends using Barker's formula (Barker 1981) at sites where there are major fractures, but suggests trying Hvorslev's anisotropic formula (Hvorslev 1951) at sites with only minor fissures or layers of different hydraulic conductivity. Unfortunately, in their simulation Bliss & Rushton (1984) also observed that during tests, water inflow was collected almost entirely from the first 10 meters of a fracture. Thus they concluded that fissure length cannot be reliably estimated based on packer tests as inflow could be similar from fissures of 10 meters or 3 kilometres long.

In mobile test setups, a dual packer, also called straddle packer, configuration is most commonly used. The system consists of two packers with the test section in between them (Fig. 8). Length of the test section can be adjusted by adjusting the distance between the packers. The ratio of distance between the packers and borehole diameter should always be at least 5:1, if common horizontal flow equations are intended to be used for interpretation (Sevee 2006).

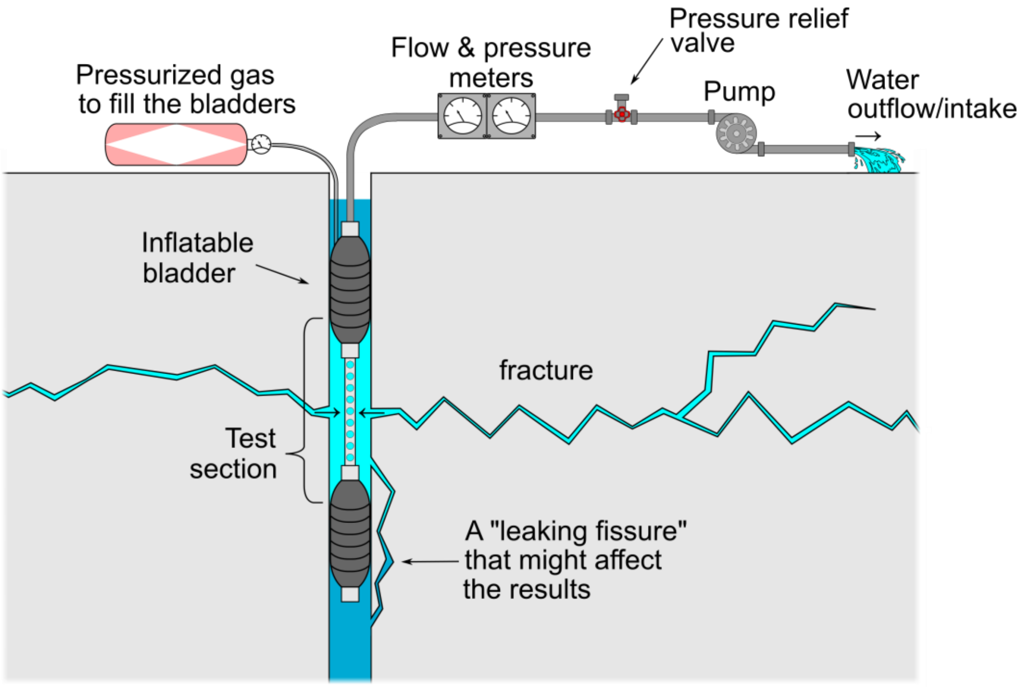


Figure 8. Example of a typical double-packer test set up.

In addition to the obvious extra hassle brought on by large number of equipment, hoses, suspender cables etc., and the formidable challenge of conducting tests in between the two sealed packers, one of the biggest risks with packers, especially in uncased boreholes, is the risk of them getting permanently stuck. This can be caused by for example loose debris, which gets wedged between the packer and the borehole wall (Rautio et al. 2004). The amount of debris can be minimized by cleaning the borehole walls before any downhole equipment is installed. The cleaning can be done using a “dummy slug”, fitted with wire brushes, or using high pressure water injection. It needs to be noted, however, that the borehole will need a long time to recover after any cleaning. Sevee (2006) notes that testing which starts from the bottom of the borehole and moves upwards also reduces the risk of equipment entrapments, as the testing itself can reduce the stability of the borehole walls. One method for loosening stuck packers is to assemble a long, extendable rods that can be used to push the packers downwards, hopefully loosening them. Other option is to try and wedge a narrow pipe, like a sleeve, around the stuck packer. Third method, and often a last resort due to the risk of the suspender cable snapping, is to simply try to pull the packer forcefully upwards using electric winches, car tow hooks etc. If all else fails, the last option is to destroy the packer using a drill rig. This is naturally very expensive and should only be considered if no other methods for removing the packer remain and the drillhole must absolutely be kept open. Report by Rautio et al. (2004) is a very good starting point for removing any stuck downhole equipment, and discusses removal methods and the numerous challenges related to the process in further detail.

One other common problem with packers in open boreholes, is the possibly poor seal between the packer and the borehole wall, which allows water to pass from the isolated section to other parts of the borehole. Similar problems can also be caused by vertical fractures, which transport water from one part of the borehole to another (illustrated in Figure 8). Also, multiple fractures on a single test section can be problematic to distinguish, and can also cause poor sealing. Poorly sealed packers are often difficult to detect during hydraulic testing. One way is to monitor water level above the packer system. If the groundwater head begins to change above the packers, water is likely drawn past the upper packer. Physically, rubber sections of packers are quite susceptible to damage, especially if the borehole contains jagged edges, the packer is moved before it has completely deflated or the packer is over-inflated (Graber et al. 2002).

3.6.1 Standard Lugeon test

What for?

Very accurately defining hydraulic properties of a single formation or fracture system by isolating the test section with packers.

Pros

- ✓ Allows assessing hydraulic conductivity in different layers and zones (both high and low flow) throughout the well or borehole.
- ✓ Can be used to estimate properties of single fractures.
- ✓ Suited for single-event studies.

Cons

- ✗ Large amount of (specialized) equipment is needed, resulting in a complicated test set-up.
- ✗ Generator is usually required to spin the pump and run a compressor.
- ✗ Water from the test needs to be discarded.
- ✗ In boreholes a real risk of equipment jams (and thus in worst case also equipment losses) exists.

How to?

Measurement steps

1. Assemble the straddle packer system with the pressure sensor in between the packers. Connect hoses running to packers and test section. Attach suspender cable.
2. Lower the assembled packer system into the testing depth and inflate the packers (usually 4-8 bar depending on the packers). Verify that the pressure stays constant and doesn't drop.
3. Start the generator, pump, compressor and test control unit.
4. Go through the test stages one by one (*Table 1*).
5. Turn off the equipment and deflate the packers. Make sure that the system is being suspended by steel wire, not hoses, and so does not collapse into the well.
6. Retrieve the equipment.

Checklist for the field

- ☐ Pump with sufficient capacity for required depths and water volumes.
- ☐ Packer system
- ☐ Test control unit and air compressor
- ☐ Multiple spools of tubing to connect the pump into the packers, and the pump and the packer test section into the control unit.
- ☐ Tube fittings, tools etc.
- ☐ Strong steel wire to support the packer system.
- ☐ Clamps to hold the steel wire and the packer system in place if the packers are not inflated.
- ☐ Power source (mains, a battery or a generator).
- ☐ Water flowmeter or a bucket to measure water discharge.
- ☐ Step watch.

The Lugeon test, named after Maurice Lugeon, a Swiss geologist who formulated the method in 1933, is a test where water is injected into a segment of borehole under steady pressure. If simplified, the Lugeon test is basically a constant head permeability test carried out in a packer isolated part of a borehole, and allows to characterize hydraulic properties of even single fractures with very high accuracy. Unfortunately, Lugeon tests are fairly difficult and time consuming to conduct, especially at sites with very low hydraulic conductivities.

During the test, pressure needs to be monitored in the section between the packers, which adds complexity and requires feed through for the pressure sensor. Before the test a maximum test pressure (P_{MAX}) for injecting water needs to be defined. The maximum test pressure should never exceed confinement stress to prevent hydraulic fracturing or jacking (Quiñones-Rozo 2010). In reality, the maximum pressure is often very roughly estimated or given a safe, arbitrary value without much pre-existing information. The test itself is conducted in five stages. In every stage the water pressure is kept constant for 10 minutes. Water inject pressures for the five test stages are determined based on the P_{MAX}, the methodology of which is shown in Table 1.

Table 1. Pressures needed for the different stages of a Lugeon test. The pressure is first increased in steps from Stage 1 to 3 and then decreased in the same steps in Stages 4 and 5.

Test stage	**Description**	**Pressure**
1st Stage	Low	$0.5 * P_{MAX}$
2nd Stage	Medium	$0.75 * P_{MAX}$
3rd Stage	Maximum	P_{MAX}
4th Stage	Medium	$0.75 * P_{MAX}$
5th Stage	Low	$0.5 * P_{MAX}$

Average values from each stage are then used to calculate so called Lugeon values. The Lugeon value is dimensionless and empirically defined as the hydraulic conductivity required to achieve a flow rate of 1 litre/minute per meter of test section under a water pressure of 1 MPa. Change in Lugeon value between different test stages can be used to characterize fissures and their reaction to the testing (e.g. fissure dilation and flushing of fillings) using a method developed by Houlsby (1976). More modern versions for interpretation have been presented by for example Quiñones-Rozo (2010), which should better suit fast and modern automatic measurement devices. The interpretation methods help to choose one final Lugeon value, which can be used to estimate hydraulic conductivity and the state of the rock mass discontinuities at the test section (Table 2).

Table 2. How to define hydraulic conductivity and state of the rock mass discontinuities based on the Lugeon value (Quiñones-Rozo 2010).

Lugeon value	**Classification**	**Hydraulic conductivity range (cm/s)**	**State of rock mass discontinuities (e.g. fractures)**	**Reporting precision (Lugeons)**
< 1	Very low	$< 1 * 10^{-5}$	Very tight	< 1
1-5	Low	$1 * 10^{-5} - 6 * 10^{-5}$	Tight	± 0
5-15	Moderate	$6 * 10^{-5} - 2 * 10^{-4}$	Few partly open	± 1
15-50	Medium	$2 * 10^{-4} - 6 * 10^{-4}$	Some open	± 5
50-100	High	$6 * 10^{-4} - 1 * 10^{-3}$	Many open	± 10
100 <	Very high	$1 * 10^{-3} <$	Many open and closely spaced or voids	100 <

3.7 Groundwater flowmeters

In surface water studies focusing on rivers or streams, the use of flowmeters is widespread. However, the relatively low flow velocities of groundwater, and the difficulty of study through wells and boreholes, have made groundwater flow measurements difficult to achieve. Despite of these challenges, and to address the difficulties of packer tests and dye tracers, several types of groundwater flowmeters have been developed over the years.

Spinner flowmeters are perhaps the simplest flowmeter design. These devices measure fluid flow based on the speed of rotation of a spinner. The spinner can be helical, i.e. like a screw, or more similar to a fan blade. With both designs, the speed of rotation is measured and related to the effective velocity of the fluid. Problem with spinner flowmeters is, that due to their mechanical nature, they are not sensitive enough to measure most natural groundwater flow (according to USGS (2016) they are typically limited to about 0.03 m/s). However, the devices can be used in combination with pumping to create pumping flow profiles, which in suitable conditions can sometimes be used to identify transmissive fracture zones.

Other, more advanced flowmeter designs also exists. Construction and operation principle of these devices varies. Some utilize physical phenomenon like heat pulses or electromagnetism to track groundwater movement. Others observe particles in groundwater by lasers or other optical methods. Because groundwater flow velocities can be very low, these equipment must be very sensitive, yet also robust enough to operate hundreds of meters below ground surface inside boreholes. Because of this, the systems initial and operational costs tend to be high and they generally require a high level of expertise to operate. On the other hand, the devices can give results with unparalleled accuracy and provide information of groundwater flow direction of individual micro-scale fractures. For example, the transverse flowmeter described by Öhberg & Rouhiainen (2000) has sensitivity better than 1 ml/h for flow across a borehole, which corresponds to a groundwater flux value of about $< 1 * 10^{-10}$ m/s. The methods have been originally developed mainly for deep geological repository studies, oil and gas exploration and quantifying major groundwater reserves. However, some of these methods, like the Posiva Flow Log, have also been used with success on mining environments.

4 GROUNDWATER SAMPLING

Sampling methods and their choice largely depends the purpose of the study. The simple, fast and easy sampling equipment, such as bailers and pumps, are favoured in studies requiring large amounts of general data of groundwater quality. These methods produce composite samples that reflect water from all the water bearing zones in the well and can be used for example during early baseline studies or in monitoring when a mine is operating normally. More advanced sampling tools and methods require considerably more time, effort and equipment, but can provide depth accurate groundwater samples from different geological formations. This kind of information can become vital for example in cases where the goal is to find out extents of a groundwater pollutant plume. In that kind of situation, more accurate knowledge can also bring sizable cost savings for the operators in the form of more definitive groundwater remediation goals. Over the years, a very large number of groundwater sampling tools have been developed. Focus on this chapter will be on the most commonly used methods, such as bailers, along with few other less common methods, like the tube sampler, which were evaluated during the KaivosVV -project. Also, when discussing the ups and downs of each method, we will try to highlight the methods' suitability for mining environmental water sampling and the variables that are typically monitored on mine sites. This means that the methods' suitability to, for example, collect volatile organic compounds (VOC), bacterial samples or dissolved gasses, is not focused on in the following chapters.

4.1 Bailers

What for?

Quickly and easily collecting groundwater samples from observation wells or vertical boreholes.

Pros

- ✓ Quick and simple.
- ✓ Disposable bailers suited for shallow depths are very affordable and can be discarded after each use, eliminating the need for cleaning or maintenance, and thus also minimizing the risk of sample contamination.
- ✓ Suited for vertical sample profiles or sampling from specific depth (with limited accuracy).

Cons

- ✗ Has a tendency to aerate samples, increase turbidity and increase the amount of colloid particles in samples – questionable sample integrity with many parameters including for example some trace-elements and dissolved oxygen.
- ✗ Requires purging which produces excess water.
- ✗ Cheap plastic bailers are not suited for deep groundwater sampling or dipping boreholes and their capability to maintain sample integrity in these conditions has been found to be questionable.
- ✗ Disposable bailers can produce a lot of plastic waste.

How to?

Sampling steps (Disposable bailer)

1. Remove the bailer from its plastic wrapping and attach a line to the upper end of the device. Check that the ball inside the device is moving freely.
2. Lower the bailer into the well.
3. Purge the well by removing 3 – 5 well volumes of water.
4. Lower the bailer into the intended sampling depth.
5. Retrieve the sampler with a smooth and steady motion.
6. Collect the sample from the bailer either by pouring water off the top or through the valve by a slow emptying device included with each bailer.

Checklist for the field

- ☐ Suitable number of disposable bailers or another type of grab sampler.
- ☐ Rope or wire.
- ☐ Suitable number of sample containers.

Bailers are very simple grab-sampling devices that in their most basic form consist only of a pipe (i.e. the sample chamber) and a bottom check valve. Sampling is done by attaching the bailer to a line and then lowering the device, usually manually, into water until the desired sampling depth has been reached. As the bailer is lowered, a bottom check valve allows water to flow through the bailer. When the bailer is being retrieved, the weight of the water inside the bailer closes the valve, keeping the sample inside.

The length, diameter and construction of the devices varies greatly. Plastics are by far the most common materials, but samplers made of for example biodegradable plastics, stainless steel or Teflon are also available. Due to the low cost of plastic bailers, they are often disposed after single use, which eliminates the need to clean up the device, but produces a lot of plastic waste. Additionally, the plastic bailers can be fitted with an extra check valve on top, or be fitted with additional weight to make deployment of the light weight samplers easier.

Open, disposable bailers are very commonly used while sampling observation wells in shallow aquifers. However, according to our tests, they probably should not be used in deeper depths and dipping boreholes (Fig. 9). More complex bailer designs have been developed for this purpose. These commonly incorporate for example a metal vessel that is evacuated or filled with over-pressured air or inert gas and inlet valves which can be opened and closed at a specific depth to collect samples. Commercial solutions are available from many different manufacturers and include for example the Discrete Interval Sampler, Kabis Water Sampler and Pneumo-Bailer. As an example, the Discrete Interval Sampler from Solinst Canada Ltd. is first pressurized with air at the ground surface, which closes a check valve in the sampler and prevents water from entering. The pressure is released when the sampler is at the target depth. This opens the valve and allows groundwater from the target depth to flow into the sampler. The sampler is then re-pressurized, preventing the introduction of groundwater from other depths into the sampler. The biggest downsides of these more advanced samplers are naturally their higher initial cost, cleaning requirements and often also more complex operation compared to their disposable counterparts.

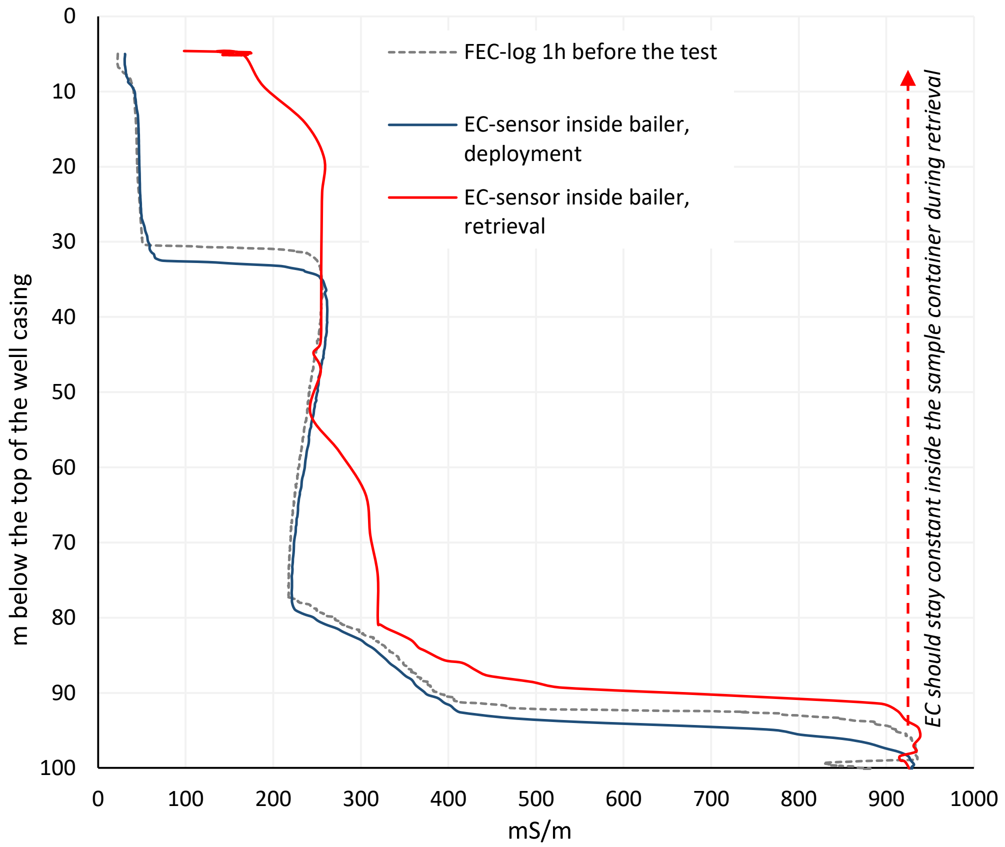


Figure 9. In our tests, a simple disposable plastic bailer with a single bottom (ball) check valve, was not able to maintain sample integrity while being used in an almost vertical (85°) uncased borehole. The test was done by suspending a CDT -sonde inside the bailer and then monitoring changes in electrical conductivity during a regular sampling event. Variation in the EC indicates mixing, which should not be happening as sampler is being retrieved and the check valve should be closed. Note how the EC curve during sampler retrieval resembles the EC profile seen inside the bailer while it was lowered.

Bailers are often a very tempting option due to their low price, and quick and easy operation. However, they are not suited for sampling without purging, which means quickly removing from 3 – 5 well volumes of water before sampling. The purging aims to remove standing water from the well and ensure that only fresh groundwater is sampled, as bailers themselves do not have the ability to draw in water from the formation surrounding the well. Purging is problematic in many ways as it causes an unnatural drawdown of groundwater surface and causes an in-rush of fresh groundwater, increasing turbulence (Puls & Barcelona 1996), which has been found to be especially harmful if samples for metal analyses are required (Parker & Clark 2004). On one hand, lowering the devices will also mix groundwater. Because of this, e.g. Sukop (2000) recommends using bailers with remotely operated valves and purging the well after the device has been lowered into the intended sampling depth. Similarly, disposable open bailers also have a tendency to mix groundwater, as the inlet valve in most bailers is smaller than the diameter of the sample chamber itself. This causes them to carry down water from upper parts of the borehole as they are descended, which can result in samples that do not match the maximum depth the sampler has reached (EON Products Inc. 2015).

The efficiency of bailers to collect representative samples has also been questioned by many others. Puls et al. (1992) found that, along with elevated As and Cr concentrations, turbidity in samples collected with bailers was almost ten times higher (200 NTUs) compared to samples collected by slow pumping (25 NTUs). In further differences to pumped samples, Pohlmann et al. (1994) noted that bailers regularly collected samples with over 20 times the amount of solid particles compared to pumped samples. They also observed that the particle distribution in bailed samples was strongly skewed towards large (> 5 µm) particles, whereas in pumped samples the particle distribution was more normal. Despite all the criticism, bailers are likely going to remain as a popular sampling method in the near future. Still, the users of the method should be more aware of its numerous limitations. Thus, from a sample accuracy viewpoint, pumped samples are more recommendable in most cases.

4.2 Tube sampler

What for?

Accurately sampling the whole vertical profile of a borehole. Depth accurate sampling of in-well water. Especially suited for dipping and deep boreholes.

Pros

- ✓ Reasonable cost, and relatively simple and commonly available components.
- ✓ Suited for accurate vertical sample profiles or sampling from specific depths also in non-vertical, dipping boreholes.
- ✓ Does not require gases or electricity.

Cons

- ✗ Without additional purging, samples represent in-well water, not fresh groundwater surrounding the well. Sampling sections need to be long enough to provide necessary sample volumes (depends on the hose diameter, but often longer than 1m sections are needed)
- ✗ Arduous method compared to e.g. bailers. Cleaning long hose sections might also be challenging.
- ✗ Relatively high amount of equipment is needed especially for deeper boreholes.

How to?

Sampling steps

1. Assemble the sampling hose according to the sampling depths. Attach check valve to the bottommost hose-section. Attach valves to bottom and top of each sampling interval.
2. Lower the assembled sampling hose slowly and evenly into the sampling well until the intended depth is reached.
3. Pull the hose back up from the well with a smooth and steady motion. Close the valves as they are retrieved from the well.
4. Disassemble the hose. Samples can be temporarily stored inside the hose enclosed between the valves.
5. Mark the hose sections according to sampling depth and empty the hoses into appropriate sample containers as soon as possible.

Checklist for the field

- ☐ Hose
- ☐ Check valve for the bottom of the hose.
- ☐ Valves for the bottom and top of each sampling section.
- ☐ Hose connectors
- ☐ Tools (e.g. measuring tape, wrenches, thread seal tape)

The tube sampler (in Finnish: *letkunäytteenotin*), is a sampling device capable of collecting continuous water profiles from deep boreholes. First introduced by Nurmi & Kukkonen (1986), the simple device consists of a bottom check valve fitted to the end of a long and flexible plastic tube. The tube can be cut into individual sampling sections connected by shutoff valves. Samples are collected by slowly lowering the hose into the borehole. While the hose is retrieved again, the bottom check valve closes and a practically undisturbed continuous water profile of the borehole is retrieved inside the hose (Fig. 10) (Nurmi & Kukkonen 1986).

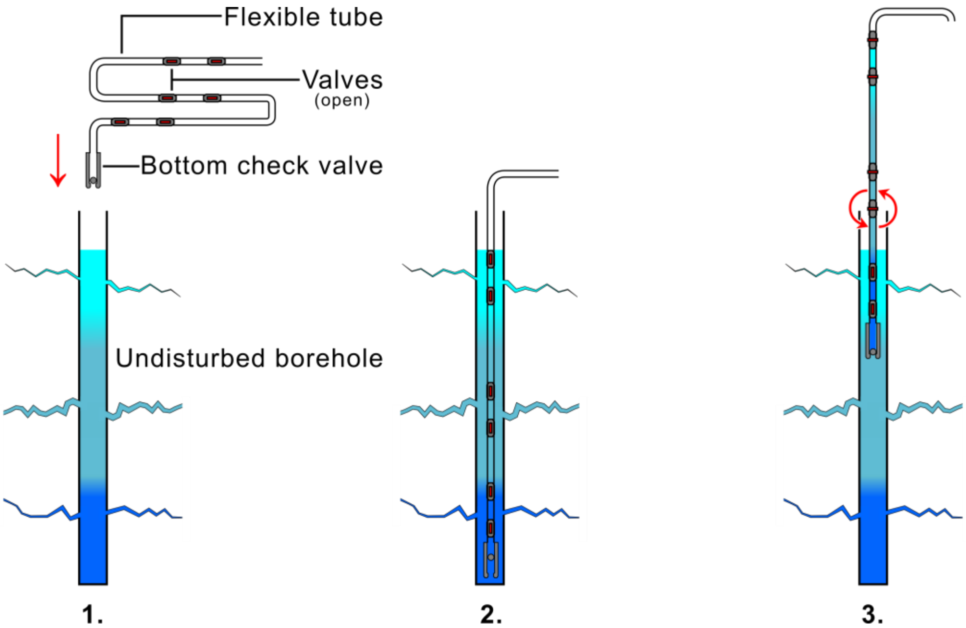


Figure 10. Sampling event using the tube sampler. 1.) A long flexible tube is fitted with a bottom check valve and valves surrounding each intended sampling section. 2.) The tube is slowly lowered into an undisturbed borehole while the bottom check valve lets water fill in the tube. 3.) The sampler is evenly pulled up, which seats the check valve and prevents water from escaping. The valves surrounding the sampling sections are closed as the tube is pulled up and the water samples are stored inside the sampling sections.

The device has been used with success to collect water and gas samples from drill holes reaching over one kilometre in depth (see for example Kietäväinen et al. 2013). During the KaivosVV project, EC from collected samples was matched against results from electrical conductivity profiling. The profile sampler seemed to preserve the original EC of the water samples at both tested boreholes. However, in another case where the sampling sections had a larger inner diameter (to provide larger sample volumes) compared to the sections and valves between them, sample integrity was questionable. Thus, even if not completely verified to be the reason for mixed samples, it can be strongly recommended that the hose should have a constant diameter to make sure that the hose is filled with minimal turbulence. Nurmi & Kukkonen (1986) also emphasize that the lowering speed of the hose needs to be lower than the filling rate of the tube, and that the filling rate gradually decreases with sampling depth. They suggest monitoring the filling rate by putting the upper end of the tube into a water container, such as bucket, and then observing the bubbles coming from the end of the tube.

Perhaps the biggest downside to the method is that, with short sampling sections and narrow tubes, the sample size can be very limited. For example, a 1 m long sampling section, with an inner tube diameter of 1 cm produces only 0.8 dl of water (i.e. too little for most lab analyses). Naturally, this can be countered with larger diameter hose or longer sampling sections, neither of which are desirable due to increased bulk, downhole volume and more difficult operation. Further, the hose, which in mining environments can realistically reach few hundred meters in length, is fairly difficult to handle and the setup (connecting fittings, cutting tube for borehole specific sampling sections, etc.) takes time, making the method slightly cumbersome. The hose is also fairly difficult to properly empty and clean between sites and having individual hoses for each site is not realistic in single-event sampling. Thus, the hose should always be moved from lower to higher concentrations and made sure that it has been thoroughly emptied between sites. An air compressor can be tremendously helpful in the emptying process. In sensitive sampling scenarios, one might also want to consider the possibility of sample contamination due to affordable and commonly available brass valves and fittings (contaminants include metals such as copper, zinc and lead), and consider more expensive stainless steel valves instead.

Still, overall, the method was found to function well also in mining environments for level-accurate in-well water sampling, as it maintained sample integrity even in dipping boreholes.

4.3 Pumped samples

What for?

Quickly, easily and affordably collecting bulk groundwater samples from relatively shallow wells/boreholes. The traditional purging method is also suited for wells with low hydraulic conductivity.

Pros

- ✓ Reasonable equipment costs. More expensive than disposable bailers, but re-usable. Very affordable options exist for shallow depths.
- ✓ Fast and easy sample collection.
- ✓ Water flow rates are usually easy to adjust
- ✓ Suited for dipping wells and boreholes

Cons

- ✗ Some pumps might be difficult to clean and contain materials (i.e. metals) that could compromise sample integrity.
- ✗ Purging can cause turbulence, which has been observed to affect sample quality.
- ✗ Purging can cause excessive drawdown in aquifers, affecting their local flow regime during sampling.
- ✗ Many pumps struggle in deeper depths (> 100m); especially affordable and easily obtainable submersible pumps.
- ✗ Usually requires electricity (although hand operated pumps do exist).

How to?

Sampling steps (Traditional purging method)

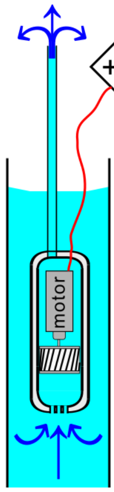
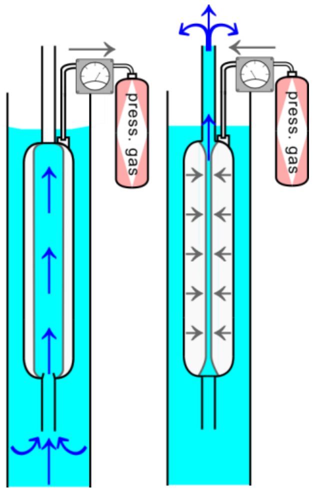
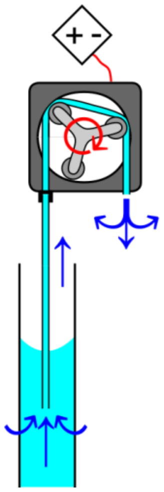
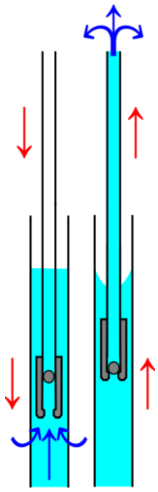
1. Attach the pump to the sampling hose and sustainer cable.
2. Lower the pump and attached hose in to the well. Avoid the near bottom of the well (due to loose sediment).
3. Connect pump main to a power source (generator or battery).
4. Remove 3-5 well volumes of water. Monitor water quality with a multiparameter sonde or EC-meter.
5. Check that the pumped water has reached a constant physico-chemical quality (e.g. temp, EC and pH have stabilized). Collect samples.
6. Turn of the pump and retrieve the equipment.

Checklist for the field

- ☐ Sample hose
- ☐ Submersible pump (or other type)
- ☐ Pump control unit
- ☐ Generator or battery to power the pump.
- ☐ Support cable for the pump.
- ☐ Tools (e.g. measuring tape, wrenches, thread seal tape)

One of the most common ways to acquire a water sample from a well or a borehole is by using a pump. However, even such a simple sounding procedure can be conducted in multiple ways, which are varyingly suited for different sites and sampling goals. Also different type of pumps are varyingly suited for different types of sampling. Some commonly used pump types, along with few of their key strengths and weaknesses, are shown in Table 3.

Table 3. Comparison of a few commonly used pump types

Type	Submersible pumps	Bladder pumps	Suction pumps (e.g. peristaltic pumps)	Inertia pumps
				
Pros	✓ Can provide high or low flow rates and the rate can be easily adjusted ✓ Usually very reliable ✓ Very affordable options available for shallow depths	✓ Provides un-aerated samples ✓ Very suitable for low-flow sampling ✓ Easy to clean and maintain ✓ Operation is simple and can be fine-tuned ✓ Usually relatively good and sufficient reach of depth (e.g. 300m)	✓ Relatively affordable ✓ Highly portable ✓ Doesn't need to be lowered into the well. ✓ Simple construction. No valves. ✓ Easy to clean between sites. Only inner tubing is in contact with the sample.	✓ Very affordable ✓ Extremely simple operation and construction ✓ Can be operated by hand in shallow depths ✓ Good for purging if motor operated
Cons	✗ Often difficult to clean. Risk of sample contamination ✗ Usually not viable for deep boreholes ✗ Large downhole volume	✗ High initial costs compared to other options ✗ Often restricted flow rates ✗ Requires compressed gas or a compressor	✗ Very often limited by maximum operational depth (<10m) ✗ Restricted flow rates ✗ Vacuum based operation can cause loss of dissolved gasses and volatile organics	✗ Very limited flow rate (especially if operated by hand) ✗ Up-down motion mixes water

4.3.1 Multi-volume purging

The traditional way to collect groundwater samples is via a purging step, where usually from 3-5 casing volumes of water are removed before sampling. During the purging, changes in for example EC, temperature, oxygen content and water colour can be monitored for additional data, and samples should only be collected if those variables have reached a steady state. After the well has been fully purged, the depth of the pump intake should not have any effect on sample water quality, as the water is drawn in from all water bearing zones along the borehole and represents a flow-weighted average of those waters (Varljen et al. 2006).

Compared to low-flow sampling (introduced in the following chapter), the traditional purging method has been noted to be easy to implement for novice users and the results have been observed to contain less variation between field technicians (U.S. Environmental Protection Agency (EPA) 2017). The purging step also draws in water further from the well compared to low flow sampling. However, purging has been questioned from a sample integrity viewpoint for a long time. The purging step induces a heavy local drawdown, which will impact the local flow regime during sampling. According to Puls & Barcelona (1996) purging has a negative impact on sample integrity especially due to high turbidity. Turbidity can cause otherwise immobile metal and hydrophobic organic compounds to mobilize, which will lead to concentration overestimates in sample analysis. This problem is sometimes countered by filtering, which can have further adverse impacts on sample integrity (Puls & Barcelona 1996). Problems are seen especially in heterogeneous aquifers with spatially varying concentrations of chemical constituents. Purging also takes time and will produce large volumes of water that needs to be discarded and/or treated.

Even if the method has some advantages over low flow sampling, it should only be considered if the use of low-flow methods is not possible due to e.g. well properties or equipment constrains.

4.3.2 Low-flow sampling

To address some of the numerous problems associated with the purging process (as discussed in Chapters 4.1 and 4.3.1), alternative sampling methods have been developed. Probably the most common of these is the so called low-flow or minimal-drawdown purge method. In this variant water is pumped from the well in low volumes for extended periods of time, which results in smaller groundwater drawdown and smaller risk of sample contamination due to turbulence (Puls & Barcelona 1996). Another big advantage of the low-flow method is that it usually produces only few litres of purge water. With the regular purging method this amount can easily be ten times more (Durham Geo Slope Indicator 2003). This difference can be substantial both cost and labour wise, especially if dealing with contaminated groundwater which has to be treated.

Low flow-sampling can be monitored using two different approaches. First, and most commonly, stabilization of physico-chemical parameters (e.g. pH, temperature and EC) in pumped water can be used to indicate when the casing water has been removed and formation water is drawn in (Puls & Barcelona 1996). In another approach, results from low-flow samples are compared to results from a conventional purging to proof that the two methods produce comparable data. The second approach is typically used only in very low yield wells that don’t produce sufficient water volumes for continuous water chemistry measurements, but where repeated purging could be very intrusive and could quickly result in for example clogged well screens (Kaminski 2003).

Installing pumps or any other downhole equipment should be avoided before sampling as such disturbances can substantially increase turbidity in the collected samples (Kearl et al 1992, Puls & Powell 1992, Nielsen & Nielsen 2006). Thus, the use of well dedicated sampling equipment is preferable (e.g. Kaminski 2003) and allows for example easier and faster sampling events that can be conducted by a single field technician (Vance 2008).

Low-flow sampling has been found to perform best in groundwater observation wells with short screens (less than 1m would be optimal) (Puls & Barcelona 1996). With longer screens, zones other than those intended may be sampled. Further still, some portion of the sampled water will always likely be acquired from the borehole casing due to vertical inflow. Martin-Hayden (2000) observed in their laboratory tests, that despite low pumping rates (i.e. 0.22 L/min), 5% of the sample water still represented pre-purge water, even after purging the equivalent of five well volumes.

Pumping rates for low-flow sampling vary according to well- and site characteristics. According to Puls & Barcelona (1996), rates varying between 0.1 – 0.5 L/min are typical for most cases, but even rates up to 1 L/min have been successfully used on sites with high hydraulic conductivity. Maximum pumping rate for a well can be defined by monitoring changes in water level, while the pumping rate is slowly increased.

Downsides of the method are fairly few and include higher initial costs, longer set-up time in the field, need to transport additional equipment from/to the field and increased training requirements for operators (Puls & Barcelona 1996). Still, due to its advantages such as minimal purge water volumes and better sample integrity, low flow sampling should be the favoured pumping method in most cases.

4.3.3 No-purge sampling

Samples can also be collected from wells without any purging. In the so called no-purge sampling method, the well/borehole is expected to be in a constantly flushed naturally due to groundwater flow. This sets one of the biggest restrictions on the use of the method, as it is only acceptable for sites where the hydraulic conductivity is already known to be high (~ K > 10^{-5} cm/sec) (EPA 2013). If used, samples should be collected using very low pumping rates or passive samplers such as passive diffusion bags (PDB). Sampling equipment should be well-dedicated or deployed long before actual sampling to avoid mixing groundwater. Thus, due to the very time consuming sampling events and restrictions on site geology, no-purge method can be considered poorly suited for most groundwater sampling events on mine sites. On the other hand, on longer monitoring programs at suitable sites, passive sampling methods have been found to be cost effective compared to low-flow sampling as they minimize labour hours and completely remove the need to treat any purged water (Stroo et al. 2014).

4.4 Level-determined sampling

Sometimes, conventional sampling equipment and approaches do not give results that are accurate enough for modern hydrogeological studies (e.g. Einarson 2006). This is mostly because in open boreholes or observation wells fitted with longer screens, traditional sampling techniques have a tendency to diminish variation due to in-well mixing caused by vertical groundwater flow. As an example, Robbins (1989) calculated that in contaminated vertically heterogeneous aquifers, in-well blending could lower maximum concentrations by ten times. In similar settings, Gibs et al. (1993) observed that the contaminant concentration in a vertically averaged sample would be only 28% of the maximum concentration in the aquifer. Even miniscule vertical hydraulic gradients will almost certainly cause ambient vertical flow of groundwater in monitoring wells and boreholes, which can lead to non-representative samples and even contaminant migration from one part of the well to another (e.g. Church & Granado 1996, Einarson 2006).

To prevent the effects of in-well mixing on sample quality, so called level-determined or level-discrete sampling methods have been developed. In some studies, the term level-determined is used for all methods capable of collecting water samples from specific depth (such as bailers, see for example Lerner & Teutch 1995). However, in this report the term is used exclusively for methods capable of reliably collecting samples from the formation itself, on the intended depth, usually by isolating the sampling section from the rest of the well or borehole by some means.

One option to achieve this, is to use well dedicated sampling systems capable of collecting samples from many depths. On the downside, dedicated sampling systems tend to be expensive to install, hard or impossible to maintain or repair and prevent using the wells for most other purposes (Lerner & Teutch 1995). This often makes the systems unsought from the site operator’s viewpoint (Einarson 2006). However, a lot of development has happened in the systems over the last decade. For example, most of the commercially available dedicated systems can nowadays be removed from the well and allow multi-level sampling up to seven depths. Still, while these systems are a good choice for groundwater quality monitoring and have even become quite affordable and readily available, they are not a viable option for short term deployment or -sampling campaigns, which are the focus in this guide.

4.4.1 Portable devices

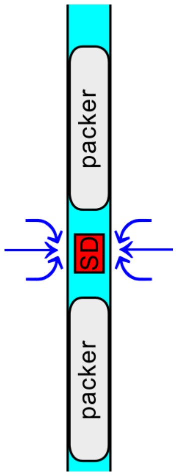
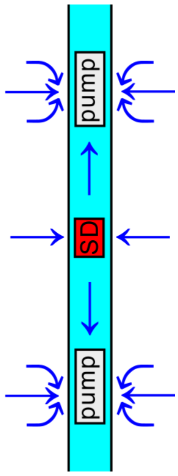
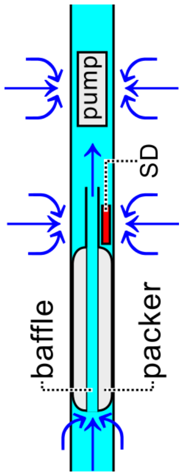
Options for portable non-permanent methods capable of doing deep groundwater level-determined sampling in a reasonable timeframe are far more limited compared to dedicated systems. One common method uses a similar packer setup as described in Chapter 3.6. Water samples can be collected from the isolated section of a well or a borehole from in-between two packers or above or below a single packer.

Bailers can be used for depth discrete sampling if the well is being pumped simultaneously to sampling. This way, the device will collect samples which represent all the water bearing zones located above or below the pump intake (Gossell et al. 1999). In addition to a pump, this method requires a bailer with valves that can be remotely opened or closed.

Level-determined sampling can also be conducted using only pumps. In the so called separation pumping, water is drawn from above and below the sampling zone. This creates a water divide where flow should be completely horizontal. It is recommended to install pumps to the top and bottom of the borehole and then adjust their rates in order to move the divide up or down. Location of the divide can be identified with a horizontal flow logger as the zone with the highest horizontal flow should be at the divide. Samples themselves can be acquired either by a third, lower rate sample pump or a bailer lowered into the divide.

On top of the previously mentioned methods, other devices have been developed for level-determined sampling, but have failed to enter wide scale use. One example is the baffle system (Table 4). Briefly described, the baffle system incorporates a packer with a large diameter tube (baffle) running through it. The tube reaches above the packer leaving a static zone between the pipe and the borehole wall. Two pumps are then used. A more powerful pump is used above the baffle to induce flow through the baffle and through borehole walls. A second, lower capacity sampling pump is installed between the baffle and the well to draw water in only from the static zone. Lerner & Teutch (1995) describes the method in further detail and also compares its performance against the other more commonly used level-determined sampling methods.

Table 4. Comparison of methods capable of depth-discrete groundwater sampling, but are also removable and transportable between sites. Additionally, some modern commercial multi-level systems can be removed after longer deployments and use packers to isolate small diameter sampling pipes/channels from each other, but aren't generally suited or designed for single-event sampling.

<table>
<tr><th>Type</th><th>(Dual/straddle) packer</th><th>Dual-pump</th><th>Baffle system</th></tr>
<tr><td></td><td>
</td><td>
</td><td>
</td></tr>
<tr><td></td><td colspan="3">SD = "sampling device". Usually a pump operating at a low flow rate, but it is possible to collect samples using for example bailers with remotely operated valves (see for example Foote et al. (1998), who used such a device to collect samples from an 5ft long, straddle packer isolated well section)</td></tr>
<tr><td>Pros</td><td>✓ Proven track record
✓ Multi-purpose. Suited from single-event sampling to long term deployments and permanent installations
✓ Some commercial solutions available</td><td>✓ Some scientific validation exists
✓ Simple equipment – no packers
✓ Moving between sampling depths is easier than with packers</td><td>✓ Only one packer</td></tr>
<tr><td>Cons</td><td>✗ High initial costs
✗ Prone to jamming in uncased wells/boreholes
✗ Moving between sampling depths can be cumbersome</td><td>✗ A lot of water that needs to be discharged
✗ Three pumps in one well is likely difficult to manage
✗ Sampling device location must be exactly at the water divide between the two pumps</td><td>✗ Very little information exists. Not commonly used
✗ Requires a packer of special construction (a baffle through the packer)</td></tr>
</table>

Naturally, portable level-determined sampling methods have their own drawbacks. They are usually quite cumbersome to use and the sampling takes considerably more time, equipment and effort compared to bailers or pumped samples. The large amount of equipment is hard to transport and clean between sites. This is especially a problem at mine sites where concentrations of effluents can vary vastly between boreholes. Thus, it is not recommendable to use equipment which has been deployed on a site with high concentrations, at a site with lower ones without thorough, off-field cleaning. Moving between sites with similar concentrations of effluents, should be manageable at the field (Lerner & Teutch 1995). Because of the technical challenges, portable level-discrete sampling options can mostly be considered viable in cases where the hydrogeological properties of a mine site are already well known and high resolution, high accuracy water quality samples are needed for special purposes such as refining an existing hydrogeological model or mapping out the extents of a contaminant plume.

4.5 Sample pre-treatment and laboratory analysis

Since sample pre-treatment methodologies as well as laboratory methodologies have been discussed widely in published literature and elsewhere (e.g. Van Heerden 1986, Ward et al. 1990, Mäkelä et al. 1992, Pulles et al. 1995, 1996, Salminen et al. 1998, Suomen vesiyhdistys 2005, Heikkinen et al. 2008, Kauppila et al. 2013, Mine Closure wiki 2015 and Gard guide 2017) they will be presented only briefly in this report, where the focus has been on the sampling techniques and hydraulic testing.

Nowadays it is very common to use outside contractors for water sample analysis. As laboratory practices and procedures vary even between well-established laboratories, it is always advisable to discuss with the laboratory for detailed description on analysing, preservation, and identification of the samples. The laboratories may also help choosing or even provide proper sample containers and adequate sampling equipment if needed. These are dictated by for example analytical method requirements, sample pre-treatment requirements, sample matrix and the contaminant content of the sample.

Due to the reactive nature of waters at mine sites, samples tend to change during transport to laboratory unless adequately pre-treated. For example, without acid conservation, iron tends to precipitate through oxidation, which decreases pH and leads to decreased concentrations in dissolved metals. To hinder any possible chemical transformations, water samples should be pre-treated accordingly to the laboratory analysis by acid conservation and filtering immediately after sampling. During transportation samples should be preserved in dark and in low temperatures, while avoiding freezing. The pre-treatment procedures for water samples carried out currently in mine environmental studies of the Geological Survey of Finland are presented in Table 5.

Table 5. Analysis methods, sample volumes and pre-treatments for water samples used in mine environment studies of GTK (modified from Kauppila et al. 2013 and Räisänen 2013).

Analysis	Sample volume	Pre-treatment
Multielement (dissolved), ICP-OES/MS-ICP	100ml	filtering with 0.45 µm GD/XP-filter or 0.2 µm vacuum filter, conserving with suprapur® HNO_3, 0.5 ml/100 ml
Multielement (total), ICP-OES/MS-ICP	100ml	conserving with suprapur® HNO_3, 0.5 ml/100 ml
Fe^{2+}, spectrophotometric	100ml	filtering with 0.45 µm GD/XP-filter or 0.2 µm vacuum filter, conserving with HCl 4 ml/100 ml
TOC, CHN-analyser	100ml	conserving with H_3PO_4 1 ml/100 ml
DOC, CHN-analyser	100ml	filtering with 0.45 µm GD/XP-filter or 0.2 µm vacuum filter, conserving with H_3PO_4 1 ml/100 ml
tot-N, ammonium	100ml	no pre-treatment
Anions (SO_4, Cl, F), suspended solids, pH, Ion-chromatographic	1l	no pre-treatment
Phosphate, alkalinity, NO_3, NO_4, Spectrophotometric	500ml	no pre-treatment

In mining environments, ground water samples should be analysed at least for total and soluble metal and metalloid concentrations and anions. In addition, dissolved organic carbon (DOC), total organic carbon (TOC) and ferrous iron (Fe^{2+}) can be significant for estimation of water quality. The dissolved and total concentrations of cations in water can be measured by using Inductively Coupled Plasma-Atomic Emission Spectrometry (ICP-AES), Inductively Coupled Plasma Optical Emission Spectroscopy (ICP-OES), Inductively Coupled Plasma Mass Spectrometry (ICP-MS) or Atomic Adsorption Spectrometry (AAS), whereas anions are determined using ion chromatography. Organic contaminants are generally determined with gas- or liquid chromatographic methods, carbon through CHN-analyser and phosphate and Fe^{2+} by spectrometric methods (Heikkinen et al. 2008 and Kauppila et al. 2013).

Many laboratories offer pre-constructed analysis packages for groundwater analysis, which include a pre-determined set of water quality variables. At least in Finland these packages are commonly constructed with drinking water analysis in mind, and could include or exclude variables relevant to mining environmental sample analysis. Variables that can be deemed irrelevant in many cases, include for example odour and colour. From important variables, for example alkalinity is sometimes omitted, as it has no limit values on drinking water legislation, but is often crucial in mining environments. As analysis costs often form a big portion of total study or monitoring costs, notable cost savings can sometimes be achieved by modifying analysis packages to better suit analysis goals. This must be done with upmost caution and good knowledge of the site's general groundwater geochemistry. Often a good and safe approach to cost savings is to vary between comprehensive and more minimal analysis packages. A minimal package can include only the most important variables, and is used regularly to track changes in these variables. A comprehensive package is used to identify important variables and to monitor possible changes in variables outside the minimal package, and can be done less regularly (e.g. once a year) once the local groundwater characteristics are mostly known and observed to be stable.

To overcome any uncertainty of the sampling procedure, all monitoring and sampling should be undertaken through common quality assurance measures and coupled with a statistically based sampling plan to maintain the integrity of the samples prior to analysis. In addition to the actual samples, duplicate samples should be taken at the same place in conjunction with actual sampling (e.g. every fifth sample). The duplicate samples reflect the precision and repeatability of the sampling procedures. Field blanks, which are usually deionised water, are taken, treated and measured parallel with the actual samples to demonstrate the accuracy and for identifying of possible contamination during and/or due to sampling and analysis procedures.

To test the precision of the laboratory measurements also the laboratories should split certain amount of the samples into subsamples called as lab replicates. It is also common procedure for the laboratories to undertake calibration blank samples to zero the measuring instrument and to check the measuring instrument periodically for a possible "drift". The lab blanks can also be compared to the field blank to pinpoint where the possible contamination might have occurred. The lab replicates and blanks are again analysed as the actual samples (Heikkinen et al. 2008, Kauppila et al. 2013 and Räisänen 2013).

5 FIELD MEASUREMENTS AND MONITORING

5.1 Measurements commonly conducted on the field

As laboratory analysis are often rather expensive and time consuming, onsite field measurements act as a quick, simple and low-cost way to provide an overview of the environmental conditions at a mine site. Field measurements also provide means of evaluating the performance of water management and treatment measures and can be used to guide sample site selection for additional analyses. As discussed in Chapter 4.5, due to the reactive nature of mine waters, water sample quality might change during transport. Onsite measurements of physico-chemical parameters should be performed with portable equipment in conjunction with sampling as results from these measurements can be compared with values obtained from the laboratory. This process is also often reversed and the laboratory measurements are used to validate results gained with field equipment.

Parameters measured in the field and the technology and methods used to measure them depend on site specific conditions. Most commonly the parameters of interest include water level, temperature (T), pH, electrical conductivity (EC), redox potential (Eh), dissolved oxygen (DO) and alkalinity. Many water quality parameters (e.g. T, DO, pH, Redox, EC, NO_3, turbidity, water level) can be measured by portable multi-parameter devices that are provided by several different companies (Fig. 11). The devices are quick and versatile and are often suited for both individual measurements and long term deployments, and have become standard issue equipment for most mine water related studies. However, these devices still come with some drawbacks. Accuracy and features of the devices vary. As an example, some manufacturers report, that from the factory, their sensors are within 1 standard deviation from the actual value, which in more common language means only 68.27% measurement accuracy. Initial equipment costs are usually quite high, but maintenance costs (calibration liquids, spare parts, sensors, etc.) can in the long run form even bigger chunk of the total costs, especially with devices that are used to monitor multiple parameters. For example, the lifespan of a pH sensor is commonly around a year, after which it needs to be replaced (In-Situ Inc. 2017).

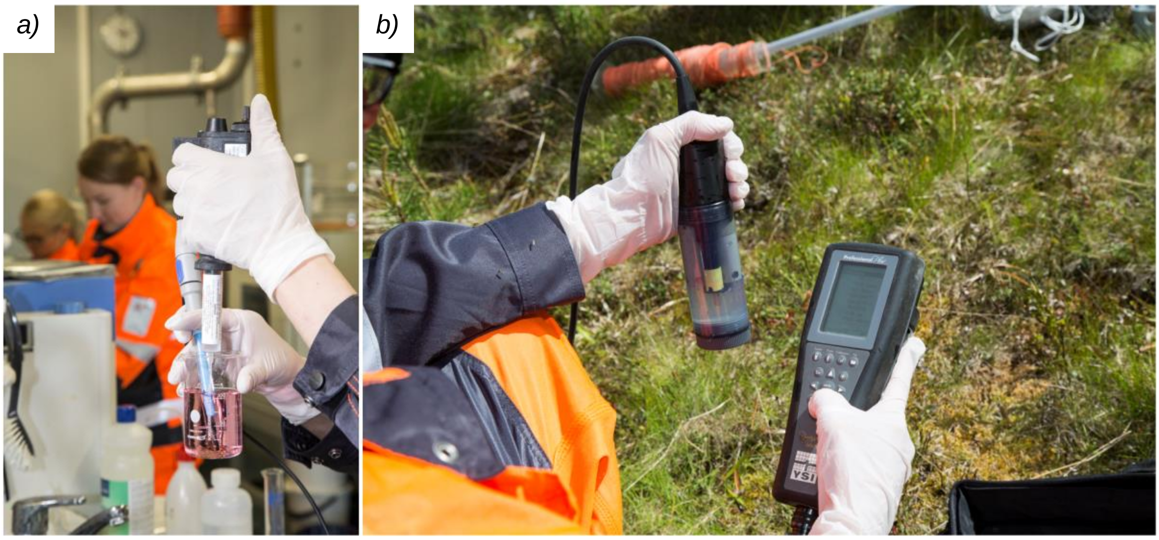


Figure 11. Water quality data collection images. a) determining alkalinity titrimetrically (sulphuric acid titration), b) measuring the physico-chemical parameters by multi-parameter sonde. (Photos © GTK)

In the following small sub-chapters, we briefly discuss individual field measurements of some of the most common physic-chemical parameters.

pH

pH in water ranges from 0 (most acidic) to 14 (most basic) and is determined by the concentration of H^+ and OH^- in a solution. At pH 7, the solution has an equal number of these ions, meaning that the solution is neutral. Since most reactions of mine waters are pH dependent and it is rather easy and cheap to measure, pH may be the most common parameter measured at mine sites.

Electrical conductivity

Conductivity is a measure of the ability of water to pass an electrical current, and so its value is mainly dependent on the ion content of water. Since natural waters usually have a rather low and constant range of EC, it can be used as a baseline to compare and detect influence of contaminant sources. As mine waters commonly exhibit elevated ion concentrations, EC is a useful parameter for monitoring the mixing of fresh water with mine influenced waters. However, reliable measurements are needed to understand contaminant sources and seasonal variation in mixing and dilution processes controlling the conductivity changes in the hydrological systems. For example, EC fluctuates depending on temperature and so the measured EC values at different temperatures need to be corrected corresponding to a standard temperature (Hayashi 2004). This temperature is usually either 20 or 25 °C and should be checked before comparing the results with others. Furthermore, conductivity results should always be compared with other water quality parameters since the results do not represent individual contaminants nor the overall state of the water (Tremblay & Hogan 2001).

Oxidation-redox potential

The reactiveness of mine waters is largely regulated by oxidation-redox potential (ORP). Because redox conditions control the mobility and reactivity of elements such as iron, sulphur, nitrogen, carbon and many metallic elements, ORP acts as a significant control of contaminant release from mine sites. Redox reactions describe the exchange of electrons between reduced and oxidized species (Pourbaix 1966). The oxidation state of many solutes influences not only their mobility in solution, but also their toxicity (e.g. As, Sb, Cr). Furthermore, due to its role in solute mobility, redox potential is a key variable in selecting suitable water treatment technologies, and evaluating their performance.

ORP is determined potentiometrically using an inert indicator electrode, usually made from gold or platinum, and a reference electrode (calomel or silver/silver chloride) (Nordström 2005). It is important to keep in mind that redox potential indicates only processes occurring in solution, not individual contaminants nor the overall state of the water.

Alkalinity

Alkalinity is a property of water that tells of its acid-neutralizing capacity. It is the sum effect of all bases (e.g. bicarbonates, carbonates and hydroxides) present in water. Without this acid-neutralizing capacity, any acidic pollution derived from mine site would cause an immediate change in the pH of the recipient water body. Thus alkalinity is one of the best measures of a water body's sensitivity to acid inputs (EPA 2013).

The most common method for measuring alkalinity is the potentiometric titration technique, in which acid of defined concentration is continuously added to a water sample of known volume until the pH of the water reaches a specified endpoint (Fig. 11) (Tremblay & Hogan 2001). The dominant species of alkalinity can be defined by titrating the sample to different endpoints (pH 8.3 for Phenolphthalein Alkalinity and pH 4.5 for Total Alkalinity). However, as in the Finnish groundwater conditions (pH < 8) alkalinity is usually strongly defined by bicarbonate, many times only total alkalinity is defined to save time and effort. Indicators such as Bromcresol Green-Methyl Red can be used to determine when the titration end point has been reached, but due to their subjectivity it is recommendable to monitor pH with a properly calibrated pH meter.

The reagent (acid) can be added either with a burette or a digital titrator. Burettes are arguably more accurate, but titrators are usually a more convenient option on the field as they are less fragile (Rounds 2012). Ready-made alkalinity titration kits exist on the market. Their use is often convenient as the kits can come with for example pre-packed reagent cartridges, a suitable digital titrator and clear user instructions (Fig. 11).

Very detailed step-by-step instructions for alkalinity titration are given by e.g. Rounds (2012), but in many cases research organizations have their own standard operational procedures (SOP) regarding alkalinity titration. The Finnish Standards Association also offers a standard *SFS 3005 Veden alkaliteetin ja asiditeetin määritys. Potentiometrinen titraus* for defining alkalinity from water samples.

Due to the titration process, alkalinity is perhaps the most tedious variable to measure out of all the variables introduced on this chapter. An experienced field technician can titrate a sample in a minute or two, but for less experienced personnel, measurements might require more time, concentration and a little bit of practise.

Dissolved oxygen

Dissolved oxygen is the amount of gaseous dissolved oxygen in a body of water. Its measurement has traditionally been neglected in groundwater studies as oxygen has been expected to be absent below water table. However, dissolved oxygen can actually have a substantial impact on groundwater quality as it can regulate the valence state of trace metals and affect bacterial metabolism (Rose & Long 1988). With modern field equipment and multi-parameter devices, DO is measured optically. This makes the measurements easy to conduct and rough device calibration fairly quick and manageable even on the field (e.g. water saturated air or air saturated water – no special calibration liquids, little bit of clean water, and takes a minute or two to conduct). Thus, while using a multiparameter device, there is often not a real reason not to measure DO while other parameters are being measured.

5.2 Continuous measurements and online monitoring

Developments in wireless data transfer have made it possible to install autonomous data loggers with passive sensors that are constantly submerged in groundwater. These systems can commonly operate for months or even years without human intervention and are powered by batteries or solar energy. There are many different manufacturers for automatic data loggers with slightly different specs, including In-Situ Inc., YSI Inc., Solinst Canada Ltd. and Heron Instruments Ltd. Commonly, the systems measure at least hydraulic head and water temperature, but more complex instruments can also monitor other parameters such as dissolved oxygen content, electrical conductivity, salinity, pH and oxidation reduction potential. Even mobile devices capable of measuring dissolved metals are being developed and the first products are currently hitting the market. Overall, the devices are developing very rapidly and newer iterations are more compact (e.g. some devices can fit completely inside a groundwater observation well without any need for external equipment), can operate reliably for longer periods without human intervention (thanks to for example different anti-fouling systems) and are overall more user friendly than ever before.

Currently the biggest strength of autonomous systems lies in real-time online monitoring, which is very hard to implement any other way. In other words, the data loggers can be used to monitor 24/7 for example dam integrity by looking for sudden changes in groundwater head or conductivity. When a breach or a similar event is recorded as a rapidly rising head or conductivity, the devices can automatically send an alert (e.g. SMS) to mine operators, notifying them of the potential danger. With the traditional labour intensive manual measurements, it would very likely take hours or even days before the breach would be noted and mine operators could take necessary action.

Long-term monitoring might seem extremely tempting to operators in the first glance as it could lower the cost related to sampling and sample analysis and potentially even lower personnel costs. However, the approach still comes with drawbacks. Accuracy and features of the devices vary, and experience is required from the end user in order to acquire the best system for each site as well as the knowledge how, when and where to use and maintain them. Initial equipment costs are usually quite high, and so, repayment periods can be long and vary a lot between applications. Real time online monitoring systems are also often related to manufacturer specific computer programs and/or cloud services, which can add cost in the form of software licenses. Outside the mine premises these expensive devices are also exposed to theft and misdeed. Further, in regards to measurement accuracy, as the groundwater in the well is never circulated (i.e. pumped), the sensors constantly measure the water that is sitting in the observation well, close to the sensors. This water might actually be substantially different from the groundwater surrounding the well, especially in cases where groundwater flow is weak (see for example Smith & Granato 1998). The problem is even bigger at sites where the well collects water from multiple aquifers in different depths, because the water in the well is actually a mixture of water from different aquifers and does not accurately represent any single aquifer. Thus, active systems that include a pump to circulate the groundwater before the measurements have superior accuracy compared to passive data loggers, but are less popular due to their complexity and higher maintenance requirements (Smith & Granato 1998).

6 METHODS FOR INTERPRETATION

Purpose of this chapter is to introduce some methods for using and interpreting the data that can be gained by hydraulic testing and groundwater sample analysis, which are the main subjects of this guide. Both statistical methods and groundwater modelling have a long history of use and good introductory books are available for both subjects. For statistics good starting points include for example Kitanidis (1997), Reimann et al. (2011), Ranta et al. (2012) [in Finnish] and Brown (2012). For groundwater modelling Appelo & Postma (2004), Seppälä & Tuominen (2005) [in Finnish], Baalousha (2011) and Fetter (2018) can be recommended, among many others. Thus, both of these concepts are introduced here only very briefly and hopefully in an easily understandable form. In real life, both of these methods are quite error prone for novice users and require user experience, good quality data and site-specific background information to be used successfully and accurately.

6.1 Statistical methods

Different statistical methods can be used to for example to simplify, characterize, analyse, classify and group different analysis results, other quantifiable site characteristics or sampling sites themselves. Thus, choosing the right statistical methods comes down largely to what the user wants to achieve. In contrast, application of statistical methods into groundwater data is often somewhat complicated and requires careful consideration. For instance, removal of outliers, an almost everyday data pre-treatment step in regular statistics, is often very undesirable from geochemical viewpoint, as different outliers often form the most interesting part of a geochemical dataset. Another possible limitation of the statistical approaches is the inability to capture deterministic principles, and hence such methods lack predictive capability. Due to the limitations inherent in hydrogeochemical datasets, robust statistics are often preferred. Robust statistics, as the name already implies, are better at dealing with small departures from model assumptions (such as non-treated outliers and non-normal distributions). Some common data pre-treatment steps for geochemical data, along with two fairly robust multivariate statistical methods that are commonly used with hydrogeochemical datasets, are briefly introduced below.

6.1.1 Data pre-treatment

Univariate normal distribution is one of the most common requirements of parametric statistics. This requirement is often neglected which can lead into potentially erroneous results, especially with hydrogeochemical data which is rarely normally distributed (Reimann & Filzmoser 2000). Both graphical and numerical methods exist for analysing data normality. Commonly used graphical methods include Q-Q plots, histograms, box plots and stem-and-leaf plots. Numerical methods include Shapiro-Wilk, Kolmogorov-Smirnov (K-S), Anderson-Darling and Cramer-von Mises –tests. Simple calculations of skewness and kurtosis can also tell a lot about the distribution without more complex tests. According to George & Mallery (2010), values for skewness and kurtosis between -2 and +2 are considered acceptable in order to prove normal univariate distribution needed for multivariate analysis. It is to be noted, however, that different programs use different formulas to calculate these values and for example skewness calculated with SPSS will differ from that calculated using R.

If the distribution is found to be non-normal, the data is often treated some way. One of the common methods is a simple log_{10} -transformation, which helps especially with right-skewness and high kurtosis often characteristic for geochemical datasets (e.g. Miesch 1976, Reimann et al. 2011). Transformations, however, cannot be done totally without a debate, as some consider them “cheating”, arguing that they’ll make the results non-comparable with results acquired from original data (see for example Changyong et al. 2014). New and more accurate methods such as generalized estimating equations (GEE) have risen to replace traditional, problematic data transformation techniques, but the older simple methods such as the log_{10} -transformation are still widely being used.

Hydrogeochemical datasets also often include so called censored values (Güler et al. 2002). These can most typically be for example laboratory results that have been below or over detection limits. In results such values are often presented simply as values less than the detection limit. In statistical analysis, such values will produce errors and need to be transformed into unqualified values (e.g. Farnham et al. 2002, Güler et al. 2002). The crudest option is to simply discard the variables with censored values. This is not a bad option in some cases, as the values with censored values are often trace elements seen in very small (< ppb) concentrations. When discarding these variables is not a viable option, many other alternative approaches exist. These include for example multiplying the values below detection by 3/4 and values above detection limits by 4/3 (VanTrump & Miesch 1977), or using values that are one-half of the lower detection limit. These simple procedures are usually only valid if only a small portion of data (<10%) consists of censored values (Sanford et al. 1993). In cases where a bigger portion of the data contains these values, the values can be imputed by using for example maximum likelihood estimates (MLE) (e.g. Sanford et al. 1993 and Güler et al. 2002).

Even further, values that are altogether missing can also cause problems in geochemical datasets. Statistical programs tend to automatically either replace these missing values with means of the variables or remove the data points with missing values case-wise, neither of which is usually desirable (Güler et al. 2002). Luckily, if the variable with missing values holds a strong correlation with another variable (e.g. Na and Cl), the missing values can often be estimated via a simple linear regression model (e.g. MacDonald & Zucchini 1997 and Güler et al. 2002). Some values, like missing alkalinity, can also be estimated based on other variables, such as ionic balance.

6.1.2 Principle component analysis

Principal component analysis (PCA) is a dimension reduction method, which in practice can be used to simplify large datasets into fewer comprehensive principal components (i.e. factors or “groups of variables”). This will make it much easier to interpret and draw conclusions from large datasets. The first principal component (PC) explains as much of the variation in the original data as possible, the second tries to explain all the variation that the first PC could not explain and so forth until all input variables can be explained by the components.

Modern computers and software have made PCA easily accessible. It is to be noted however, that PCA is an error-prone procedure even with large datasets and optimal data. The accuracy and viability of PCA compared to the similar, but even more complex factor analysis is often debated (Costello & Osborne 2005). However, PCA is a more robust method (Ranta et al. 2012) and has less input variables, which makes it somewhat better suited for novice users. On top of this, results from PCA and true factor analysis are often strikingly similar (Costello & Osborne 2005).

6.1.3 Hierarchical cluster analysis

Hierarchical cluster analysis compares variables of individual samples in pairs and forms clusters of samples possessing the least dissimilar values (Bridges 1966). The method combines similar samples into smaller number of groups that are easier to handle than large quantities of data. The method is best suited for datasets having less than 200 samples. The method is largely used in different applications of statistical analysis, being also a common tool in water sample analysis (see for example Vega et al. 1998, Suk & Lee 1999, Alberto et al. 2001 and Shrestha & Kazama 2007).

6.2 Groundwater modelling

Groundwater modelling can be an efficient tool for groundwater management and remediation. Models act as a simplification of reality and can be used to investigate certain phenomena or to predict future behaviour. The challenge is to simplify reality in a way that does not adversely influence the accuracy and ability of the model output to meet its goals. Generally speaking, groundwater models can be used for one or more of these general purposes:

1) To generate a hypothetical system where problems can be studied and solved
2) To describe and help understand groundwater systems
3) To predict or forecast changes in these systems (either artificial or natural).

As an example, estimating water inflow into a future mining excavation is an important part of mine feasibility studies. Knowledge about the quantity and sources of water inflow helps mine operators to estimate the need, scale and costs for dewatering systems. Sufficient dewatering is crucial in order to keep mine operations safe and continuous (e.g. Fernandez-Rubio 1978, Singh et al. 1985). Most components contributing to the total water inflow into a mine pit are easy to accurately estimate or measure (like surface water inflow or precipitation). However, quantity and characteristics of groundwater inflow, that often contributes a significant portion of the total inflow, is harder to measure, but can often be estimated by groundwater models. At mining environments, modelling is also commonly used to predict e.g. changes caused by the termination of mining activities and impacts of mining to water resources during mining operations (Hentinen 2015).

Groundwater models can be classified into three basic categories: physical, analogue and mathematical models (Baalousha 2011).

Physical models, as the name suggests, are actual physical models in the form of for example sand tanks. They are useful in many cases (e.g. estimating chemical quality of drainage from tailings or waste rock piles in a smaller scale) and are relatively easy to set-up. Unfortunately, they cannot handle very complicated problems, and so their use in groundwater context is often limited (Baalousha 2011).

Analogue models utilize other, more understandable or analysable systems to model groundwater flow and transport. Baalousha (2011) gives an example of this in the form of Ohm's law of electric current flow, which is similar to Darcy's law of groundwater movement. As electric current moves from high voltage to lower voltage, groundwater moves from higher to lower head. The use of analogue models has largely ended due to modern computers (Bear et al. 1992).

The third category, mathematical models, often in the form of computer code, are the most common option when modelling groundwater movement and reactions. They can be further divided into simple, but limited analytical- and complex, yet accurate numerical models. The following sub-chapters briefly discuss the use of these types of solutions in groundwater modelling.

6.2.1 Analytical models

Analytical models are only available for simplified groundwater and contaminant transport problems and were developed before the use of numerical models (e.g. Baalousha 2011). They are basically closed-form solutions of groundwater flow equations under specific assumptions and boundary conditions. These models are usually relatively simple, computationally inexpensive, and can be a reasonable option especially in the early stages of a mine project when the geology and hydraulic properties of the mine site are not yet well known (Marinelli & Niccoli 2000).

Essentially all groundwater modelling is based on Darcy's law, which describes the flow of fluid through porous medium as a function of pressure gradient. At the most basic level, groundwater flow in an aquifer can be presented as 1-dimensional flow-through system using two groundwater level observation points (Fig. 12). However, to estimate groundwater flow in mine environment, more advanced methods, taking into account a broad range of variables in nature, are required.

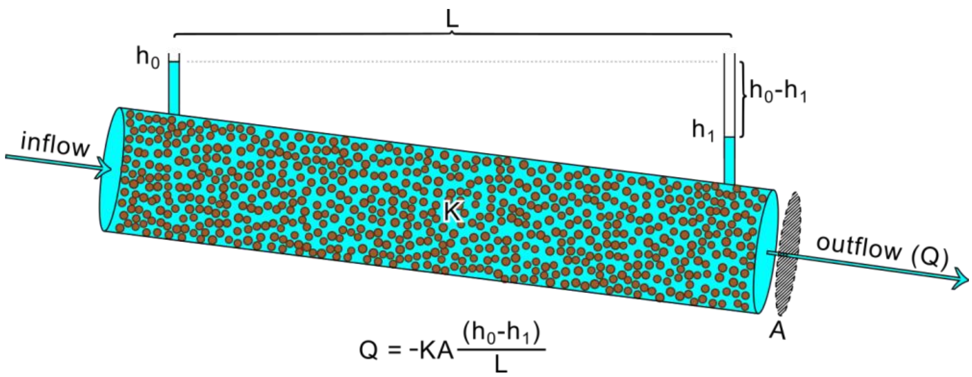


Figure 12. One-dimensional groundwater flow through porous medium. Where Q = total discharge, K = permeability of the medium, A = cross sectional area of flow, h_0-h_1 = pressure difference between two observation points, and L = length over which the pressure drop is taking place.

Numerous analytical models have been developed over the past decades for different kinds of mining- and groundwater related applications (see for example Theis (1935), Jacob & Lohman

(1952), Hantush & Jacob (1955), Hantush (1959), Pérez-Franco (1982), Singh & Atkins (1984, 1985), Singh et al. (1985), Singh & Reed (1988), Hanna et al. (1994), Marinelli & Niccoli (2000), Ardejani et al. (2003) and Neville (2017)). Some of these models, focusing on estimating flow into open mine pits and excavations, are presented in Table 6. These models have been designed specifically for the use in mines and often try to take the special properties of mine pits into account. For example, Marinelli & Niccoli (2000) conceptualized the mine pit by two specific zones that are modelled separately (Table 6). The analytical model for the upper zone estimates the groundwater flow conditions above the bottom of the pit, whereas the model for the lower zone estimates flow conditions below the mine pit. This approach allows the model to estimate water flow through the bottom of the mine pit, while models originally developed for wells would consider such flow negligible. Neville (2017) has collected and solved discharge equations for basic hydrogeological settings and pit geometries, and even provides these solutions in an easy-to-use Excel form, presenting an excellent and up-to-date starting point for anyone interested in trying out these models.

Table 6. Some analytical models suitable for estimating groundwater flow into open mine pits. A generalized, combined nomenclature is given following the equations. For actual usage, the reader should check the appropriate reference for more specific variable descriptions and -conditions as they vary between models.

Conditions	**Equation**	**Reference**
Linear flow into the sides of an excavation in an unconfined aquifer	$Q = -K\frac{(h_0{}^2 - h_p{}^2)}{A}L$	Neville (2017)
Radial flow into the sides of a circular excavation in an unconfined aquifer	$Q = -\pi K\frac{(h_0{}^2 - h_p{}^2)}{\ln\left\{\frac{R}{R_p}\right\}}$	Neville (2017)
Steady-state flow into a vertical mineshaft fully penetrating an aquifer	$Q = \frac{4\pi K \Delta h h_a}{W(u)} \quad where\ u = \frac{R^2 S}{4Tt}$	Theis (1935)
Linear, steady state flow from an unconfined aquifer into a mine pit	$Q = \frac{KL(h_0{}^2 - h_x{}^2)}{R}$	Singh & Reed (1988)
Steady state flow into a mine pit partially penetrating an aquifer.	*Flow through pit walls:* $h_0 = \sqrt{h_g{}^2 + \frac{W}{K_h}\left[R^2 \ln\left\{\frac{R}{R_p}\right\} - \frac{(R^2 - R_p{}^2)}{2}\right]}$ $After\ R\ is\ defined\ from\ the\ above\ equation\ trough\ iteration:$ $Q = W\pi(R^2 - R_p{}^2)$ *Flow through pit bottom:* $Q = 4R_p\left(\frac{K_h}{m_2}\right)h_d \quad where\ m_2 = \sqrt{\frac{K_h}{K_v}}$	Marinelli & Niccoli (2000)
Lateral, radial flow to a circular pit fully penetrating an unconfined aquifer.	$Q = \frac{4\pi T \Delta h}{\ln\left\{\frac{2{,}25Tt}{R_p{}^2 S}\right\}}$	Kruseman & De Riddler (1979), Hanna et al. (1994)

Nomenclature:

h_0	*Original head of water / original thickness of saturated zone (conditions and units vary)*
h_a	*Thickness of a formation being dewatered (m)*
h_p	*Head at the excavation (m)*
h_g	*Thickness of the saturated zone above the pit bottom (m)*
h_d	*Hydraulic drawdown along pit bottom (m)*
h_x	*Dynamic water table at distance x from the pit wall (m)*
Δh	*Drawdown / lowering of piezometric surface or water table from the original head h_0 (m)*
Q	*Quantity of flow / flow rate / pumping rate (units vary)*
R	*Radius of the depression cone at the mine boundary / Radius of the required drawdown (m)*
R_p	*Radius of the well, excavation or pit (m)*
S	*Storability of the aquifer*
t	*Time (units vary)*
T	*Transmissivity (units vary)*
K	*Hydraulic conductivity (m/s)*
K_h	*Steady-state horizontal hydraulic conductivity (m/s)*
K_v	*Steady-state vertical hydraulic conductivity (m/s)*
L	*Pit wall length (m)*
W	*Distributed recharge flux*
$W(u)$	*Theis' well function*
m_2	*Anisotropy parameter*

Despite their original purpose, all analytical methods grossly simplify the actual groundwater conditions. These assumptions are hardly ever met in nature which adds uncertainty on the modelling results.

Some of the most common assumptions related to analytical models are listed below:

- *Aquifer has infinite or strictly defined finite area and it is homogeneous, isotropic and has uniform thickness at least on the area of the potential mine*
- *Groundwater surface is expected to be horizontal at a natural state*
- *Aquifer is pumped at a constant discharge rate*
- *Water flows into wells through the whole thickness of the aquifer*
- *Well diameter remains constant through boreholes*

Methods focusing on transient conditions commonly make these further assumptions:

- *Water stored in wells is ignored*
- *Discharged water causes instant drop in groundwater level (head)*
- *Drop in groundwater level (head) does not affect groundwater flow*

Due to the aforementioned and other possible assumptions, analytical methods commonly have difficulties taking into account factors such as:

- *Sloping water tables*
- *Complex geometry, geology (e.g. material anisotropy, faults or dykes), and flow topology*
- *Aquifers on different depths separated by aquitards.*
- *Variation of storage coefficient in relation to time.*
- *Compression of aquifer due to variation in permeability, stress and depth.*
- *Boundaries and connections with surface water systems*.

6.2.2 Numerical models

Numerical models rely on the numerical solution techniques of partial differential equations describing groundwater flow. Generally, they can be considered to be more versatile and not limited by as many assumptions and standard boundary conditions as analytical models. This makes them more accurate and better suited for complex mining or groundwater environments with heterogeneities (Singh & Atkins 1983, Ardejani et al. 2003). On the down side, numerical models are often very complex and require accurate variables (i.e. a lot of expensive field research), while still possessing their own set of limitations and sources of difficulties (e.g. problems modelling saturated/un-saturated flow, confined/unconfined flow or flow in non-uniform hydraulic conditions (Ardejani et al. 2003)).

A lot of baseline data and expertise are required for accurate numerical modelling. With groundwater flow models, at least accurate geological and geographical information along with knowledge of the anthropogenic inputs are needed. This includes, for example, knowledge of different sedimentary structures and shape of the bedrock surface, accurate elevation data, and sources and drains of water (e.g. extent of rain, possible existence of wells, drains, tunnels, mining pits and surface linings like asphalt, location of rivers and their flow rates and the location of major bedrock fractures). Some of this information might be hard, time consuming, expensive or sometimes even impossible to gather.

Groundwater flow modelling is the simplest, and is most commonly done, on saturated sediment layers where groundwater flow is mostly linear. Accurately estimating flow through the unsaturated zone or especially through bedrock fractures is considerably more difficult and less common. Still, even in relatively simple hydrogeological settings, errors that could compromise the accuracy of the model are commonly found. Usually these errors are caused by inaccurate definition of geological- or flow related parameters, or induced by small geometrical errors like inaccurate borehole measurements and poorly modelled ground surface topographies (Lehikoinen 2012).

The first step in any modelling process involves the construction of a conceptual model (Fig. 13). This model consists of the assumptions that verbally describe the groundwater system, transport processes that take place in it, the mechanisms that govern them, and the relevant medium properties. The conceptual model is envisioned by the modeller for the purpose of providing information for constructing the actual model (Bear et al. 1992).

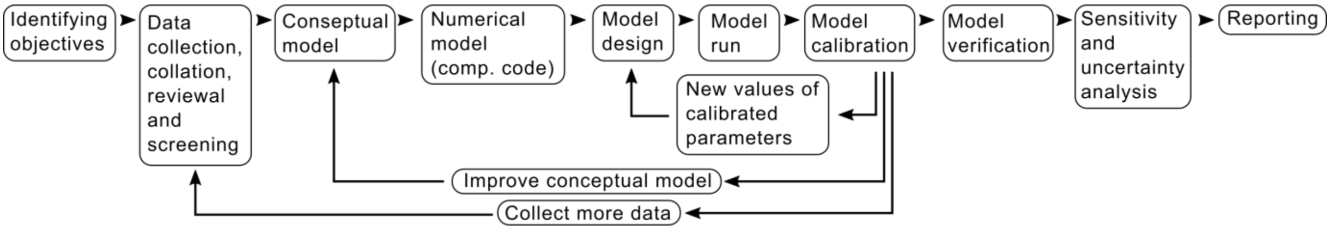
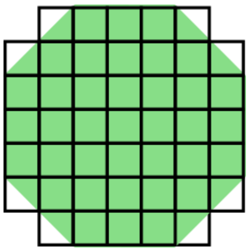
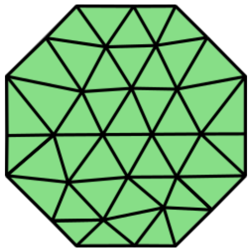
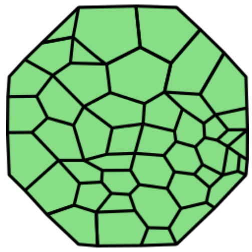


Figure 13. Stepwise methodology of groundwater modelling (Modified from Bear et al. 1992 and Baalousha 2011).

The next step is to express the conceptual model as a mathematical model which is then further expressed by computers as a code in the form of a numerical model. The most widely used numerical models use either finite difference, finite element or finite volume method (Table 7). The models can be one-, two-, semi-three- or three-dimensional.

Table 7. Comparison of finite difference, finite element and finite volume methods. Finite difference and finite element after Baalousha (2011).

Type	Finite difference	Finite element	Finite volume
			
	Example 2D model grids made over an octagon.		
Cell geometry	Rectangular	Triangular	Almost anything (both structured and unstructured)
Documentation	Well documented. Numerous extensions to add functionality and address weaknesses.	Good documentation.	Relatively new in groundwater modelling.
Model boundary	Difficulties in incorporating irregular and curved boundaries	Can incorporate irregular and curved boundaries.	Very good at incorporating any type of boundary
Nodes	At the centre of a cell	At vertices and flux boundary	At the centre of a cell
Mesh/grid building	Easy	Difficult and time consuming	From easy (structured) to difficult (unstructured grids)
Anisotropy	Difficult to incorporate	Easy to incorporate	Very easy to incorporate. Unlike others, allows grid size to vary between vertical layers.
Computation time	Can be long especially with 3D grids	Acceptable	Faster especially in environments where some layers can be simplified
Well known computational codes	MODFLOW	FEFLOW	MODFLOW-USG

By far, the most common code for groundwater flow modelling is MODFLOW, which is a modular finite-difference flow model. The model has been developed by U.S. Geological Survey (USGS) since the early 1980s and is available free of charge for all uses and applications. Integration of graphical user interfaces (GUI) to codes like MODFLOW, has made numerical modelling considerably easier to approach for novice users. Many of these GUIs also expand the capabilities of the original codes by adding for example solute transport modelling capabilities or compatibility with other software like ArcGIS. Most professionally used flow modelling software have commercial licenses. Commonly used software include GMS, Leapfrog Hydro, Visual MODFLOW and Groundwater Vistas. Non-commercial software include ModelMuse, MODFLOW- GUI and iMOD. All these tools have very similar features and in practice, the choice of a code and software comes largely down to what software licenses are available and what are the modeller's personal preferences (Dixon 2013).

Along with groundwater flow modelling, it is also possible to model geochemical reactions occurring in groundwater systems. Commonly done modelling types include: (1) speciation modelling, which is used for calculation of free ions, complexes, and saturation index values, (2) inverse geochemical modelling, which can be used to identify reactions causing changes in water chemistry between hydraulically connected points, (3) direct geochemical modelling which is used to predict the chemical composition of water resulting from pre-determined reactions, and (4) reactive transport modelling, which can be used to estimate how solutes (often some type of contaminants) move and react in groundwater over time by combining flow and transport models with geochemical models.

By far, the most commonly used standard program, which includes all types of modelling indicated above, is PHREEQC developed by the USGS. However, PHREEQC can only be used to model 1D (one-dimensional) transport through saturated porous medium. Therefore, other software packages (such as PHT3D and PHAST) have been developed to combine PHREEQC's geochemical capability to 3D flow (e.g., MODFLOW) and transport models (such as MT3DMS). In mining environments, three dimensional transport models allow to predict the movement of contaminants and changes in groundwater quality for decades into the future. This can be extremely useful while for example carrying out environmental impact assessments or planning mine closure (Hentinen 2015).

6.2.3 Estimating groundwater inflow through fractured bedrock

Characterizing water flow through fractured rock can be seen as one of the biggest challenges of modern hydrogeology (Neuman 2005, Faybishenko & Benson 2000). Flow through fractured bedrock (consisting of crystalline rock) is hardly ever uniform and is characterized by erratic heterogeneity, non-horizontal orientations and turbulent flow conditions making modelling difficult (Neuman 2005). Flow through bedrock happens mostly through unconformities like fractures, faults and dykes, but also very slowly through the rock matrix itself. Flow along the unconformities commonly dominates and defines flow in the system and so the slow flow or seepage through the rock matrix is usually discarded. In order for these unconformities to conduct water, they need to have sufficient aperture to allow water to flow and they need to be connected to other unconformities to form continuous flow paths. In case of fractures, large fractures usually have higher aperture and thus usually also higher transmissivity than smaller fractures. On the other hand, smaller fractures are usually more numerous than large fractures, and when combined, small fractures can have a large impact on the total flow budget (Golder Associates 2018). Even if suitable methods and models (e.g. FEFLOW, NETFLO, FRACTRAN, FRAC3DVS, SWIFT, etc.) for analysing flow in fractured bedrock exist, it is still rare to apply them to mine environments due to complexity of the models and fairly hard-to-get input parameters, like accurate structural information and groundwater flow estimates (Rapantova et al. 2007).

7 CONCLUSIONS

Hydrogeological studies in mining environments, and especially in fractured bedrock, can be performed with higher quality than has been done previously. The methodologies to measure and interpret the data exist, but the main problem is still the lack of experienced people and teams that can perform all the necessary steps from pre-investigations to final solutions, e.g. modelling of water evolution in a mine pit. To perform a successful mine water study a multidisciplinary approach is needed involving geophysicists, hydrogeologists, geochemists, field personnel and modellers, to name a few.

Even though hydrogeology of fractured bedrock is deemed the most difficult branch of hydrogeology, the benefit of performing such studies in mining areas can lead to cost savings and faster permitting processes in the long run. Study of previously unstudied bedrock hydrogeology can also reduce uncertainties in mine planning, permitting and operation.

This guidebook presents some of the methodologies used at the Geological Survey of Finland in mining hydrogeology studies in fractured bedrock. Most of the methods presented are applicable for the porous sediments, but in Finland, focus at most mine sites should be in the fractured bedrock.

Method selection is the most important part of hydrogeological study, as many of the methods can give similar results. Often, adequate data can be obtained with fairly inexpensive methods, while more detailed studies might require equipment that can be relatively expensive and complicated to use. On the other hand many of the methods presented in this guidebook produce unambiguous data, and the use of several methodologies is needed to reduce result uncertainty.

This guidebook gives a short introduction on how hydrogeological study in bedrock can be performed efficiently and can produce valuable results. The guidebook does not go in to the details and theory of different methodologies, but gives an overview, a checklist for the field work and tries to emphasize the importance of proper method selection.

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