Master´s thesis

NTNU
Norwegian University of
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& Technology
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Resources Engineering

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Sunndalsøra groundwater works

- Water quality assessments

Master's thesis in geology

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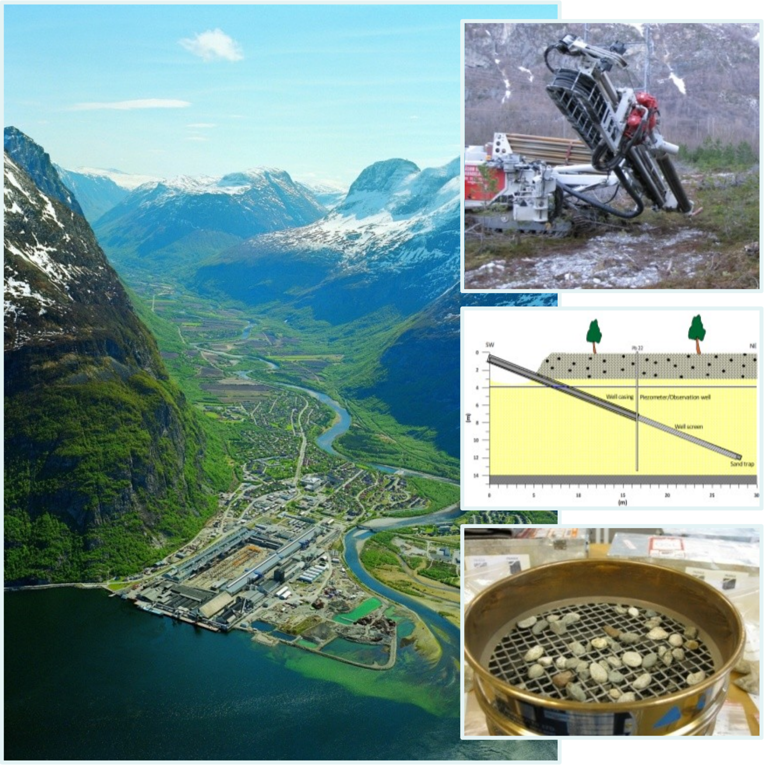

NTNU

DESCRIPTIONS OF THE ASSIGNMENT (NOR/ENG)

NTNU
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MASTER'S THESIS
MASTEROPPGAVEN

The candidate's name/Kandidatens navn: Mari Vestland

Title of thesis/Tittel på masteroppgaven: Sunndalsøra groundwater works. Water quality assessments./Sunndalsøra grunnvannsanlegg. Vurderinger av vannkvalitet.

Extended text/Utfyllende tekst:

Hovedformålet med oppgaven er å vurdere vannkvaliteten ved Sunndalsøra grunnvannsanlegg. Vurderingene skal bygge på prøvepumping av en nyetablert brønn og prøvetaking av eksisterende brønner og infiltrert elvevann. Det skal dokumenteres hvilken effekt infiltrasjon av elvevann har på grunnvannskvaliteten og om grunnvannskvaliteten endres som følge av stopp i uttak fra brønn B. I tillegg skal grunnvannskvaliteten beskrives og vurderes ut fra:

- Løsmassenes mineralsammensetning
- Løsmassenes innhold av organisk materiale
- Grunnvannets oppholdstid og strømningsmønster

Grunnvannskvaliteten skal hovedsakelig vurderes ut fra fysisk-kjemiske parametere, men mikrobiologisk kvalitet skal også vektlegges. Det bør gis en vurdering av hvordan grunnvannets kvalitet endres fra elv, via infiltrasjonsbasseng og til brønnene. Det skal særlig legges vekt på grunnvannets jerninnhold og de parametere som har betydning for jerninnholdet i grunnvannet. På grunnlag av vurdering av grunnvannskvalitet og kapasitet på ny brønn skal det foreslås andre aktuelle tiltak for ytterligere forbedringer i vannkvaliteten. Dette inkluderer også tiltak rettet mot sikring av vannkilden. Det skal også vurderes i hvilken grad den nyetablerte brønnen løser anleggets problemer med jern.

Oppgaven skal gjennomføres med feltarbeid (kapasitetstesting av ny brønn, vannprøvetaking, grunnvannstandsmålinger) og laboratorieundersøkelser av sedimentprøver (XRD, XRF, glødetap, kornfordelingsanalyser). Et litteraturstudium skal også utføres, og bør gi en kort sammenstilling av grunnvannskjemi relatert til løseligheten av jern i grunnvann. Problemer med jern i grunnvann til drikkevannsforsyning, både nasjonalt og internasjonalt, samt en kort presentasjon av hvilke metoder som benyttes for å redusere/fjerne jerninnholdet skal gjennomgås. Her bør både in situ-metoder og konvensjonelle rensemetoder omtales.

Specialization/Studieretning: Miljø- og geoteknologi

Main study profile/Hovedprofil: Hydrogeologi

Time interval/Tidsrom: 20.08.2008-10.06.2010

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Teacher/Faglærer

PREFACE

A great number of unconsolidated aquifers are distributed throughout Norway that represents potential sources for drinking water. Still, only some 15 % of the Norwegian population receives groundwater as potable water. This is a low percentage by comparison with other countries on the continent, e.g. Denmark and Germany. In Norway on the other hand, there is a strong tradition for using surface water for household purposes as result of the relatively pure water contained in easy accessible river courses. Groundwater generally holds a much better natural microbiological quality than surface water. This is due to the degree of natural protection of most groundwater basins. The pressure on groundwater reservoirs in Norway is also quite low with respect to conflicting area utilization and issues of pollution.

However, problems related to poor groundwater quality can occur as a response of certain natural constituents that are present at too high concentration. Problems related to use is often experienced when groundwater with high values of dissolved iron comes into contact with air or oxygen-rich water and subsequently forms rust. This problem is recognized at groundwater works all over the world. Iron concentrations are usually too low to amount for a level representing a health threat. Instead, iron may cause economic load on the water works in question related to recurring problems with e.g. precipitation in well systems and transmission lines, as well as staining and discoloration at the consumer. Water is often treated with conventional and/or in-situ methods, which require specific operational and maintenance routines. Problems with high values of iron have been an issue at Sunndalsøra groundwater works since the first establishment of wells in the mid 1980's, and after the relocation of the groundwater works in the beginning of the 1990's. In this assignment, the situation at the groundwater works is highlighted in an attempt to solve problems caused by iron.

The assignment, *Sunndalsøra groundwater works. Water quality assessments,* has taken its form in parallel with an ongoing project at Asplan Viak, Trondheim. The project aims at improving Sunndal water works with emphasize on both improving the water quality and to secure a more stable water supply. The assignment has been made in cooperation with Asplan Viak, Sunndal municipality and NTNU.

Bernt Olav Hilmo, hydrogeologist at Asplan Viak, has been my supervisor and essential discussion partner throughout the process, as well as a source of inspiration for future work on the topics of this assignment. Bjørge Brattli, my adviser at NTNU, has contributed with discussions and advises. In addition to thanking my supervisors, I would like to use this opportunity to return thanks to Eilif Lervik, Bjørn Røkkum and Kurt Amundbø at the technical department in Sunndal municipality for good collaboration, hydro/environmental geologist at Asplan Viak, Rolf Forbord, for being an important source of inspiration and ph.d.-student Hanne Kvitsand at NTNU for cooperative field work in March 2009.

Trondheim, 10-Jun-2010

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Mari Vestland

ABSTRACT

Comprehending factors controlling groundwater quality is of great importance in the context of exploiting and supplying groundwater of good quality for public use. Groundwater quality is conditioned by aspects of geological and hydrological character, as well as involving physical, chemical and biological processes. Groundwater quality also depends on factors controlled by well performances and characteristics, as well as the operational routines at the waterworks in question. This research combines disciplines of hydrogeological, geological and geochemical character in order to describe and understand the water quality at the public water supply at Sunndalsøra in the county of Møre og Romsdal, Norway.

The research is based on field and laboratory investigations in relation to the establishment of an additional production well in an aquifer exploited for public water supply in Sunndalen. Sediment samples have been collected in relation to the drilling of the well. Water sampling and groundwater level measurements have been conducted as part of an aquifer test and during the test pumping of the well. The test pumping also includes field analyses of groundwater, as well as water sampling for characterization of physical-chemical parameters of all wells, the infiltration basin and the river Driva. Collected sediment samples were analyzed in the laboratory at Department of Geology and Mineral Resources Engineering (IGB) using X-ray analyses, grains size analyses and loss on ignition. Water samples have been analyzed by the Geological Survey of Norway (NGU) combining tools of chromotography, mass spectrometry, nephelometry and titrations. In addition, water samples have been collected for microbiological analyses in relation to an ongoing ph.d-project at NTNU and by Sunndal municipality.

A fluvial aquifer at Sjølandsøra is exploited for public drinking water supply, supplying the community of Sunndalsøra and surrounding areas. Problems related to terms of use result from too high values of iron in one of the production wells (well B). Elevated levels of iron have been a persisting problem affecting well constructions and transmission lines, as well as causing discoloration and staining experienced by the consumers. This problem and its concerns define the starting point of a series of improving actions, including the work related to this assignment. A new production well has been established with the aim of replacing well B. In order for any replacement to take place, the new well has to meet all requirements for good drinking water as stated in *the drinking water regulations*, as well as demands for well capacity.

The mineralogical composition of the fluvial sediments comprising the aquifer, contribute with ferrous iron through chemical dissolution processes in a zone where the groundwater is sufficiently poor in oxygen and acidic in character. This mineral-groundwater interaction may explain the problems experienced with well B, which has the most deeply situated well screen. Relatively high concentrations of carbon dioxide in groundwater extracted from well B suggest that microbial decomposition is an important process in the aquifer sediments in vicinity of this well. This process of decomposition sets the stage for an anoxic groundwater evolution in the deeper and finer grained (fine sand/silt) parts of the aquifer, leading to efficient iron dissolution. Groundwater flow patterns and residence times gives an additional insight to factors influencing the geochemistry of the groundwater basin. It is documented that dissolved iron concentrations increase with reduced aquifer recharge. This is to a great extent linked to a decrease in infiltration of river water deriving

from Driva. Studies and observations related to different operational situations at the works document the close link between functionality of different components of the works and water quality, as well as the relationships between quality and capacity aspects. The closing of well B resulted in an increase in dissolved iron concentrations in the overlying well A as this well now started to withdraw the deeper and more reduced groundwater. The infiltration basin also has an influence on water quality by means of contributing to aquifer recharge in vicinity of well D.

Results and observations from the test pumping period prove that improving actions should be implemented at the groundwater works. The new production well meets all requirements in terms of capacity and quality. With this aspect accounted for, actions aiming at improving water quality with respect to the reduction of high values of iron are preferably implemented in the following order:

1. Establish a water supply from the newly established well (D), which then replaces well B as a production well of drinking water.
2. Well B should still be operative such that the withdrawal of groundwater equals an amount necessary to avoid extraction of reduced groundwater into the overlying well (A). Groundwater from well B (c. 10-15 l/s) could be pumped by means of diversion to Driva or function as an additional water supply for the infiltration basin. The latter option is feasible in that will contribute to an increase in pH and alkalinity from re-infiltration of groundwater relatively rich in CO_2. Iron hydroxide is expected to precipitate in/on the filter mass of the basin, and the establishment of a second infiltration basin should be considered.
3. Improve the river intake by means of establishing a new tilted well beneath the river bed and/or extending the already existing river intake into the deeper parts of the river. This will secure a more predictable water supply on an annual basis.
4. Increases in surface area of the basin should be viewed in connection with the establishment of a second infiltration basin (basin 2). This will enhance water supply during periods of dry spells through an increase in aquifer recharge.

Suggestions for actions contributing to an additional improvement of water quality:

5. It is strongly recommended that the degree of groundwater protection is increased, to better protect the aquifer from future activities and land use. This action should be accomplished with the definition of four protection zones, and with an extension of the protection zone south of well D. The well area and the infiltration basin must be secured with a lockable fence (zone 0).
6. Consider possible water reserves, as well as an improvement of the Kalken water works, to ensure a stable water supply on an annual basis.
7. Reduce the amount of bacterial counts in well D by means of disinfection with e.g. chlorine. Regeneration (disinfection) should also be conducted in relation to all wells and transmission lines.
8. Regenerate (refill) the filter mass of the infiltration basin with the aid of crushed limestone/marble. This action will increase the groundwater alkalinity, pH and calcium. If considered feasible, this step might replace the present use of $Ca(OH)_2$ added at the treatment facility.

SAMMENDRAG (NOR)

Forståelsen av hvilke faktorer som innvirker på grunnvannskvalitet er viktig ved bruk av grunnvann som tas ut i forbindelse med en kommunal vannforsyning. Grunnvannskvalitet betinges av geologiske og hydrologiske faktorer, så vel som fysiske, kjemiske og biologiske prosesser. I tillegg kontrolleres vannkvaliteten av forhold som knytter seg til brønnfunksjonalitet og konstruksjon, samt drifts- og vedlikeholdsrutiner hos det enkelte vannverk. Dette studiet kombinerer disipliner av hydrogeologisk, geologisk og geokjemisk karakter, for å kunne beskrive og forstå vannkvaliteten ved Sunndalsøra grunnvannsanlegg i Møre og Romsdal, Norge.

Studiet baserer seg på felt og laboratorieundersøkelser knyttet til etableringen av en ny produksjonsbrønn i en akvifer i Sunndalen. Sedimentprøver ble samlet inn i forbindelse med boringen av den nye brønnen. Vannprøver for analyse av fysisk-kjemiske parametere samlet inn fra infiltrasjonsbasseng og brønner, samt registrering av grunnvannsnivåer, ble utført under prøvepumpingen. Prøvetaking for bakteriologiske analyser av grunnvannet ble gjort av Sunndal kommune. Utvalgte fysisk-kjemiske parametere ble analysert i felt. Sedimentprøvene ble analysert i Ingeniørgeologi-laboratoriet ved Institutt for geologi og bergteknikk, NTNU, med bruk av metoder for analyse av kornfordeling, røntgenanalyser og glødetapsanalyser. Norges geologiske undersøkelse i Trondheim analyserte vannprøvene med metoder som kromatografi, massespektrometri, nefelometri og titrering. I tillegg ble vannprøver samlet inn for analyse av mikrobiologi i forbindelse med et pågående ph.d.-prosjekt ved NTNU.

En fluvial akvifer på Sjølandsøra benyttes i vannforsyningen av Sunndalsøra og tilgrensende områder. Bruksmessige problemer har oppstått som følge av for høye jernverdier i en av produksjonsbrønnene (brønn B). For høye verdier av jern har vært et vedvarende problem som har påvirket både brønnkonstruksjoner og ledningsnett, samt ført til problemer hos forbruker i form av blant annet misfarging og utfelling i vaskeservanter. Jernproblemene definerer utgangspunktet for flere utbedrende tiltak ved grunnvannsanlegget, i tillegg til arbeidet knyttet til denne masteroppgaven. En ny produksjonsbrønn er etablert og skal erstatte brønn B. For at det skal kunne opprettes vannforsyning fra den nye brønnen må den, i tillegg til kapasitetskrav, tilfredsstille alle krav kvalitet i henhold til Drikkevannsforskriften.

Mineralsammensetningen i avsetningen bidrar med toverdig jern gjennom kjemiske forvitringsreaksjoner i de deler av magasinet der grunnvannet er tilstrekkelig oksygenfattig og surt. Denne interaksjonen mellom mineral og grunnvann kan forklare problemene som erfares med brønn B, som er den av produksjonsbrønnene som har det dypest plasserte filteret. Estimat av CO_2-konsentrasjoner viser at nedbrytning av organisk materiale er en noe mer aktiv prosess i nærheten av denne brønnen. Aerob nedbrytning av organisk materiale etablerer reduserende forhold i de dypere delene av akviferen, som igjen leder til økt løselighet av toverdig jern. Det er vist at jernkonsentrasjonene øker under forhold med redusert grunnvannsnydannelse. Dette knyttes i størst grad til nedsatt infiltrasjon av elvevann fra Driva. Beskrivelser av grunnvannets strømningsmønster og oppholdstid gir større forståelse av forholdene i akviferen og bedre innsyn i faktorer som kan virke inn på de geokjemiske forhold som styrer løseligheten av jern. Studier og observasjoner av ulike styrings(drifts)forhold ved anlegget tilsier at det er en nær sammenheng mellom vannkvalitet og funksjonalitet av ulike installasjoner ved anlegget, så vel som mellom kvalitet og kapasitet.

Avstengningen av brønn B ga en økning i jerninnhold i overliggende brønn A, som et resultat av at denne brønnen nå begynte å ta ut dypere og mer redusert grunnvann. Infiltrasjonsbassenget har også en betydning for vannkjemien I form at infiltrert elvevann fra bassenget bidrar i nydannelsen av grunnvann i nærheten av brønn D.

Resultater og observasjoner fra prøvepumpingsperioden viser at utbedrende tiltak bør gjennomføres ved grunnvannsanlegget. Det er dokumentert at den nye produksjonsbrønnen tilfredsstiller alle krav til kapasitet og kvalitet. På bakgrunn av dette foreslås i prioritert rekkefølge følgende tiltak for å forbedre vannkvaliteten med hensyn på å redusere jerninnholdet i grunnvannet:

1. Etablere vannforsyning fra brønn D, som da erstatter brønn B som produksjonsbrønn.
2. Fra brønn B bør det fremdeles tas ut en viss mengde vann (ca. 10-15 l/s), slik at man unngår å trekke redusert grunnvann inn i overliggende brønn (A). Grunnvann fra brønn B kan pumpes ved avverging til Driva eller infiltrasjonsbassenget. Sistnevnte alternativ vil kunne bidra til økt pH og alkalitet gjennom re-infiltreringen som da vil foregå med grunnvann med noe høyere CO_2-innhold. Det forventes at jernhydroksider vil felles ut på bassengbunnen, og derfor bør man vurdere å etablere et eget infiltrasjonsbasseng.
3. Utbedre elveinntaket ved å etablere en ny skråstilt brønn under elvebunnen og/eller forlenge eksisterende inntak lenger ut i elvens dypere parti. Dette vil sikre en mer stabil vannforsyning på årlig basis.
4. En økning i infiltrasjonsbassengets overflate må ses i sammenheng med etableringen av et nytt basseng (basseng 2), der vann fra brønn B infiltreres. Dette kan bidra til å sikre vannforsyningen i tørrværsperioder.

Forslag til andre tiltak som kan bidra til å forbedre den generelle kvaliteten ved anlegget:

5. Det anbefales på det sterkeste at grunnvannsmagasinet sikres bedre gjennom en områdebeskyttelse (klausulering), slik at en framtidig konflikt med aktiviteter og arealbruk unngås. Områdebeskyttelsen bør innebære opprettelsen av fire sikkerhetssoner rundt anlegget og en utvidelse av sikkerhetssonen sørvest for brønn D. Brønnområdet (sone 0) må gjerdes inn og sikres med låsbar port.
6. Mulige reservevannskilder bør vurderes, så vel som en utbedring av Kalken vannverk, for å øke leveringssikkerheten på årlig basis.
7. Det bør foretas en desinfeksjon (kloring) av brønn D før denne tas i bruk, slik at kimtallet reduserer til godkjent nivå. Regenerering (desinfeksjon) bør også gjøres av brønner og ledningsnett.
8. Filtermassen i infiltrasjonsbassenget bør regenereres (byttes). Dette vil innebære tilsetting av knust kalk/marmor, slik at både alkalitet og pH øker. Oppnås ønsket virkning kan denne behandlingsmetoden erstatte eksisterende løsning med tilsetting av kalkhydrat fra vannbehandlingsanlegget.

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ABBREVIATIONS

AV	Asplan Viak
DNMI	Norwegian Meteorological Institute
DOC	Dissolved organic carbon
EC	Electrical conductivity
GPR	Ground penetrating radar
IGB	Department of Geology and Mineral Resources Engineering
LOI	Loss on ignition
NGU	Geological Survey of Norway
NTNU	Norwegian University of Science and Technology
NVE	Norwegian Water Resources and Energy Directorate
ODEX	Overburden Drilling and Exploration
SOM	Soil organic matter
TDS	Total dissolved solids
TOC	Total organic carbon
WGR	Western Gneiss Region
XRD	X-ray diffraction
XRF	X-ray fluorescence

INTRODUCTION

This section will provide an outline of the assignment with a presentation of the background for investigations, purpose of the work conducted, and aspects concerning accomplishments. A chronological overview of previous research and investigations leading to the establishment of the existing groundwater works is also given.

BACKGROUND

The starting point of the assignment was the need for an evaluation and suggestions for improvement of Sunndalsøra groundwater works in Sunndalen, Møre og Romsdal county. The water works experiences from time to time reduced water quality related to elevated levels of iron. According to investigations made by Asplan Viak (AV) this is caused by high iron contents in one of the production wells (well B). This is the well with the most deeply situated screen. Problems experienced are related to terms of use, by means of decreased functionality of the well systems and the system of transmission lines, as well as problems related to staining of e.g. clothes and sinks at the consumers. Based on this, it was decided in a project by AV to establish one more production well to replace well B.

AV has conducted the pre-investigations to decide on the best location and dimensions of a new production well. During the winter of 2009 a well was drilled, and test pumping was initiated subsequently. The thesis is accomplished as part of the well establishment and the subsequent test pumping of approximately six months duration.

AIMS OF THE STUDY

Main focus is attributed to assessing the water quality at Sunndalsøra groundwater works. This is made in relation to the test pumping of a newly established well, as well as sampling from already existing production wells (wells A, B and C) and infiltrated river water. The study also includes documentation/observations of effects of the following operational settings at the groundwater works:

1. Documentation of changes in groundwater quality from the closing down of well B.
2. Documentation of how infiltrated river water affects the groundwater quality.

A description and evaluation of the groundwater quality will also be given with respect to:

1. Mineral and elemental composition of the sediments.
2. Organic contents of the sediments.
3. Groundwater flow patterns and residence time.

Groundwater quality should mainly be assessed with focus on physico-chemical parameters, but aspects of microbiology are also of relevance. An evaluation should include how water quality changes from the river, via the infiltration basin and to the production wells. Iron content and relevant geochemical factors controlling iron solubility should be emphasized. On the basis of groundwater quality assessments and the capacity of the new well, other actions should be suggested with the aim of improving the water quality at the works. This should also include actions aiming at protecting the aquifer. The degree of success of establishing a new well shall also be assessed with respect to how this action functions in solving the problems with iron.

The assignment includes field work (step-drawdown tests/capacity testing, sediment sampling, water sampling and groundwater level measurements) and laboratory analyses of sediments (grain size analyses, XRD, XRF and LOI). The test pumping period will indicate variations on both capacity and quality parameters with time. Evaluations of results from the test pumping will define the basis for suggestions for improvements. This was achieved through the closing of well B and by observing effects of reduced capacity of the infiltration basin.

A literature study shall also be included in the study with the focus on literature providing an overview of the geochemistry that governs iron solubility. Problems experienced with high values of iron in groundwater will be described, reflecting national and international problems related to terms of use. A short presentation of in-situ and conventional treatment methods for iron removal from groundwater will be given.

ACCOMPLISHMENT

The thesis is conducted at NTNU (Norwegian University of Science and Technology) in cooperation with AV and Sunndal municipality. It comprises a literature study, field investigations and laboratory analyses. Data has been collected through field work and laboratory analyses, as well as from studies of existing data and reports. The literature study bases upon the collocation of national and international literature, as well as reports connected to relevant projects. A more detailed description is given of the laboratory and field work:

LABORATORY WORK

Sediment samples have been analyzed in the Engineering geology laboratory and at the X-ray laboratory at the Department of Geology and Mineral Resources Engineering (IGB), NTNU. Samples were analyzed for grain size distribution and mineralogical and elemental composition using methods of X-ray diffraction (XRD) and X-ray fluorescence (XRF), respectively. Organic contents of the sediments were quantified with the method based on loss on ignition (LOI).

FIELD WORK

Field work has been carried out in connection to the drilling of the new well, as part of aquifer tests and in connection with the test pumping. One set of sediment samples was taken for analyses during well drilling. These samples, as well as samples from two investigation wells collected at an earlier stage by AV, have been analyzed at IGB. Water samples were sampled from Driva, the infiltration basin, all production wells and the newly established well as part of the test pumping. Collected water samples have been sent to, and analyzed by, the Geological Survey of Norway (NGU) for major ion contents, turbidity, pH, alkalinity, color, electrical conductivity and temperature. Measurements of groundwater level and field analyses of selected physico-chemical parameters have been made. In addition, water samples for microbiological analyses were sampled and analyzed as part of a ph.d-project at NTNU (ph.d.-student Hanne Kvitsand).

PURPOSE OF METHODS AND INVESTIGATIONS

Step-drawdown tests will provide data for calculation and characterization of groundwater basin parameters and for estimating the maximum capacity of the new well. Analyses of sediment samples will give an insight to the mineralogical and elemental composition of the aquifer sediments, as well as the contents of organic matter of the deposit. Grain size analyses of collected sediments are also an important tool to understand the geological aspects of the deposit, e.g. depositional environment and sedimentary processes, but also as an additional source for data to the understanding of aquifer hydraulics. Consequently, this will be important for understanding groundwater flow, which mainly bases upon recordings of groundwater level. Groundwater flow will be essential for estimating residence time, understanding aspects of recharge and aspects of water chemistry. Groundwater level measurements are essential for defining the area of infiltration and consequently for the protection of the aquifer. Water samples will make the basis for assessments of the water chemistry and microbiology of infiltrated river water and production wells, forming the fundament for describing how water quality depends on natural (seasonal) variations and how these sources are affected by changes in operational situations. Additionally, the test pumping period will provide essential data of how water quality parameters and capacity vary with time. Assessments of results from the test pumping as a whole will define the basis for suggestions for improvements with respect to reducing iron concentrations and other actions to improve the water quality at the works, as well as providing a basis for evaluating the degree of feasibility of establishing a new production well (well D) as an action for solving the problems with iron.

The literature study will provide a useful frame for understanding processes and factors controlling water quality, as well as giving an insight into methods that might be applied as a water treatment step (conventional and/or in-situ methods).

PREVIOUS STUDIES AND WORK – DEVELOPMENT OF A GROUNDWATER SUPPLY

The study area has been subject to studies and investigations by NGU in connection with the development of the groundwater works, and more recently by AV with the purpose of improving the existing works (2008-). In the following part, a short chronological overview of the different steps leading up to the existing water works will be given. A time scale is shown in Figure 1.

The first hydrogeological investigations were conducted by NGU in 1983, concerning the suitability of establishing a groundwater works for the extraction of potable water to the public water supply of Sunndal municipality (NGU 1984 a). Investigations aimed at finding a groundwater basin dominated by river infiltration feasible for withdrawal of groundwater close to the community at Grøa. After some research it was found that none of the locations were suitable, and it was decided that one should investigate other locations nearby. One of the alternatives was the fluvial deposits close to Sunndalsøra in vicinity of the existing facility.

During the year of 1984 NGU made supplementary groundwater explorations on the initiative of Sunndal municipality (NGU 1984 b). Investigations were made on the flood plains at Sjølandsøra. Geo-electrical surveys were conducted together with probing, and wells were established for both investigation and production purposes. A pump test was initiated with the aim to collect and evaluate water samples and sediments. Results from the test were considered positive for groundwater utilization, and it was recommended to start a test pumping program.

This pumping aimed at providing information to evaluate the long term groundwater quality and the capacity of the basin. The outcome of these analyses could decide if the works would get approval to function as a public water supply. After conduction of probing and test pumping, the production wells were tested by NGU from 1984 trough to 1986. From these investigations and evaluations of possible conflicts with activities in the area, it was concluded that the location at Sjølandsøra was feasible for the extraction of groundwater for public water supply.

However, problems with iron occurred shortly after the establishment. In order to reduce problems with iron, the water works installed an in-situ treatment method based on the VYREDOX-method. This method failed to reduce the iron content and it became a necessity to come up with another solution. The solution involved further investigations and test pumping in the period 1990-1992 upstream the VYR-facility and closer to Driva (Furu 1991). The outcome of this was the establishment of existing production wells A, B and C. All wells were constructed as tilted wells and supplied the municipality from week 15 in 1992 (NGU 1992 b). The water treatment facility was built the same year.

In 1997 an infiltration basin was constructed as an action to reduce iron concentrations and to increase groundwater recharge (Asplan Viak 1997). The basin was in use only for some time after the establishment. As a response to a new period of negative experiences with increased iron contents, the basin was once again put into operation in 2005.

Remarks from *the Norwegian Food Safety Authority* concerning the safety of the water supply and the water quality led to the engagement of AV in 2008 (Asplan Viak 2008 a). In line with regulations of the legal framework AV described the different components of the works, its discrepancies and degree of vulnerability. Alternative solutions for improvements of the works were outlined with respect to hygienic barriers and problems related to iron, as well as suggestions for possible actions for improvement.

As an integrated part of the improvements of the groundwater works, supplementary hydrogeological investigations were made the same year in order to secure an optimal location and construction of 1-2 new production wells (Asplan Viak 2008 b). The aquifer was mapped thoroughly with ground penetrating radar (GPR). Detailed studies were made to evaluate the water yield of the sediments, water quality was found from test pumping and water sampling from investigation wells.

Parallel to the conduction of this assignment, investigations have been performed in accordance with improving actions suggested in this study and as part of the ongoing project at AV (Asplan Viak 2010). This involves hydrogeological investigation with GPR, probing, drilling of investigation wells and subsequent test pumping in relation to the establishment of a new tilted well beneath the river bed (section 5.4).

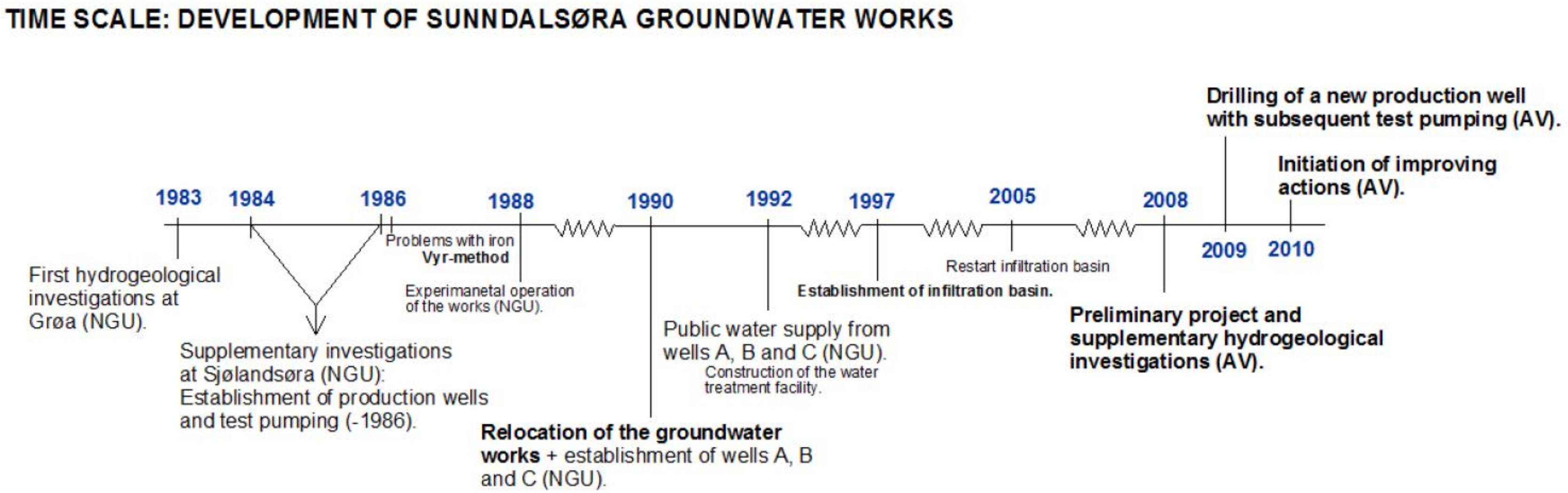


Figure 1: Time scale showing the development of Sunndalsøra groundwater works.

1. DESCRIPTIONS OF THE STUDY AREA

1.1 GEOGRAPHICAL OVERVIEW AND LAND USE

Sunndalsøra groundwater works and the study area are located at Sjølandsøra in Sunndalen, in the municipality of Sunndal. The municipality is by size the greatest in the county of Møre og Romsdal. The area at Sjølandsøra comprises a flood plain on the southwest side of the river Driva, defining the location of the aquifer (Figure 2). Driva runs through the valley of Sunndalen, which finally drains into Sunndalsfjorden. Sunndalen is a U-shaped valley, deeply incised into the surrounding mountain massif reaching some 1500 m a.s.l. In total, c. 90 % of the municipality is situated above 700 m a.s.l.

The land areas closest to the groundwater works are mainly woodlands, with some farmland and cultivated sections located further upstream from the works (Figure 2). The area is also utilized for recreational purposes, and Driva is used for fishing.

With respect to the legal framework, issues of land use and municipal master plans are attended to by *The Planning and Building Act*. The location of the groundwater works is regulated through this act *(Section 20-4, 1. Nr 4),* and defined according to this as an area that is reserved or is to be reserved for specifically defined use in line with this or other acts. Surrounding areas are also regulated through this act (*Section 20-4, 1, Nr. 2*) as areas intended for agricultural, nature and open-air recreational purposes (Lovdata). *Appendix VIII* (CD) provides a more detailed map with current land utilization in the municipality.

1.2 SOURCES OF POLLUTION

Farming activity takes place further upstream from the groundwater works as mentioned in the previous section (1.1) and can represent a possible threat to the water quality by means of farmland runoff. A shooting range is located at the foot of the mountain to the south, but present level of activity is unknown. Runoff from this site could potentially be a threat. Incidents of acute pollution (point source) from e.g. vehicles in the well area can also be crucial. Driva constitutes another potential source of pollution, by means of having the capability to infiltrate polluted river water through aquifer recharge.

The Pollution Control Authority (SFT) has no reports of contaminated ground in the area in vicinity of the groundwater works. Two areas downstream the aquifer, defined as a municipal landfill and contaminated ground, respectively, are classified in *category 2* with respect to possible influence on the environment (SFT). The approximate locations can be viewed in Figure 2. *Category 2* means the risk of influence on the surroundings is low. These sources are not considered to be of any harm to the facility because of the great distance and location downstream of the aquifer.

Overview map: Location of important stations/locations in Sunndalen (2010)

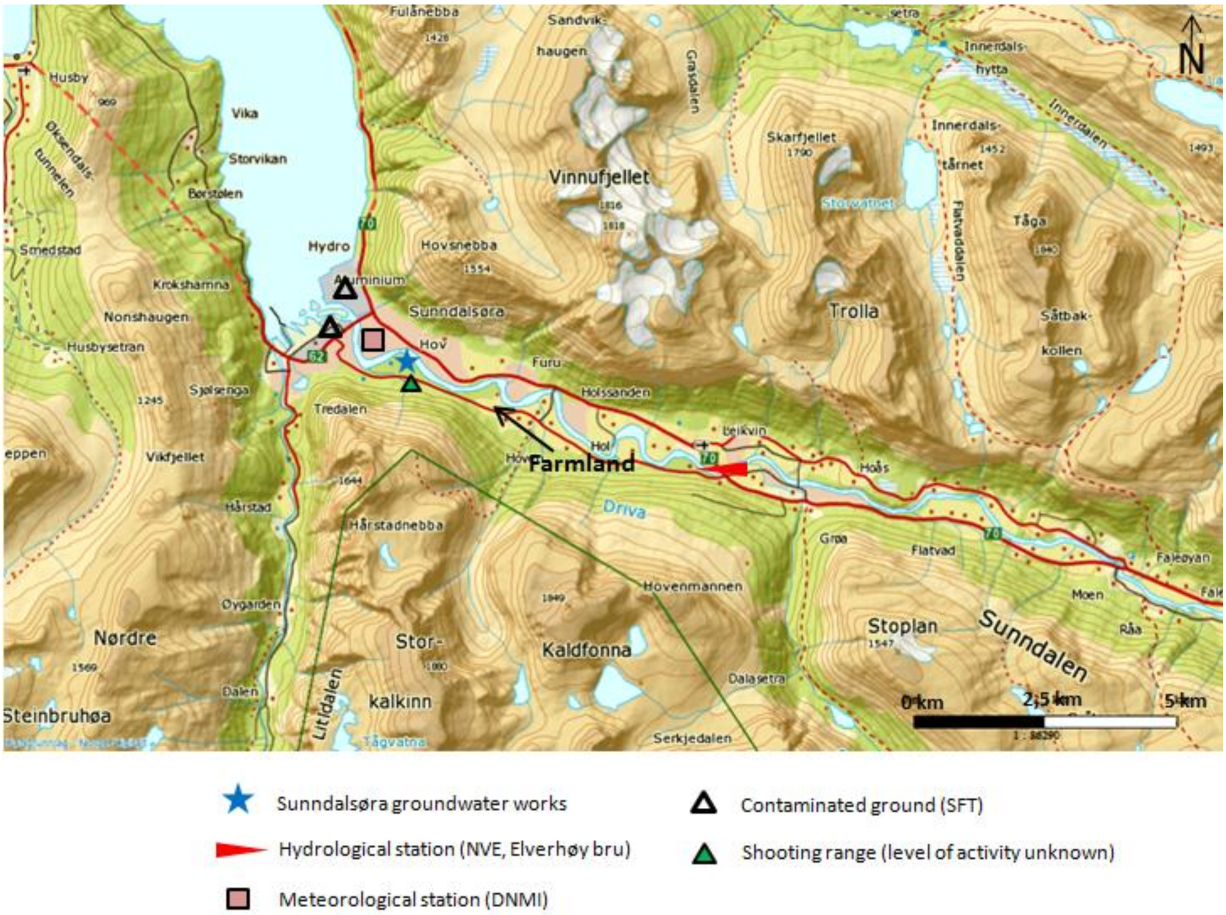


Figure 2: Overview map of Sunndalen with stations/locations of importance to the study (2010).

1.3 HYDROLOGICAL ASPECTS

1.3.1 CLIMATIC CONDITIONS

The study area is located at latitudes with a temperate climate. The climatic conditions are dominantly that of a coastal temperate climate, sometimes called a sub-oceanic climate (NOU 1983). Winters are experienced as mild with equally mild summers. Mean annual air temperature reaches 6,7 °C, and the mean annual precipitation is 961 mm. January is the coldest month with an air temperature of -0,3 °C, and the only month recorded to have a mean monthly value below zero. In the other end of the scale, the warmest month is represented by July with a temperature reaching 14,0 °C (Figure 3).

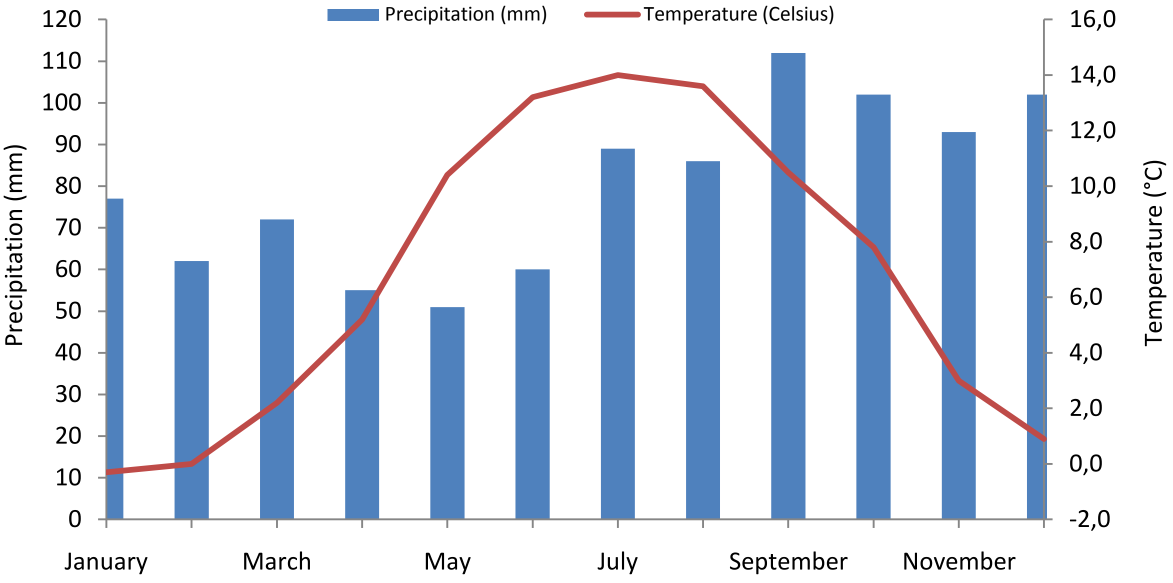


Figure 3: Mean monthly air temperature and precipitation based on data recorded at the meteorological station at Sunndalsøra in the climatic period of 1961-1990 (Data modified from DNMI).

Sunndalsøra receives the greatest amount of precipitation during autumn months with the highest values of 112 mm recorded in September. May is the driest month with 51 mm precipitation. These values represent the average of a data series from the climatic period 1961-1990, and derive from the meteorological station at Sunndalsøra administrated by the Norwegian Meteorological Institute (DNMI).

1.3.2 WATERSHED AND RIVER CHARACTERISTICS

Sunndalen is contained in one of the largest watersheds in the region of Mid-Norway, the Driva watershed. The watershed covers approximately 2493 km^2 and stretches over the three counties of Møre og Romsdal, Sør-Trøndelag and Oppland (Petterson 1999). The river originates from springs in the mountainous area of Dovre at elevations up to 2286 m.a.s.l, draining a distance of some 150 km from source area to the sea (NOU 1983). More than 70 % of the watershed is located above elevations of 1000 m a.s.l. This is experienced through relatively stable snow condition and a period of snow melting typically occurring in May-July associated with increased discharge. Autumn months are known for low discharges, and usually no significant flooding events (Petterson 1999). Discharge data from the test pumping period is displayed in Figure 4:

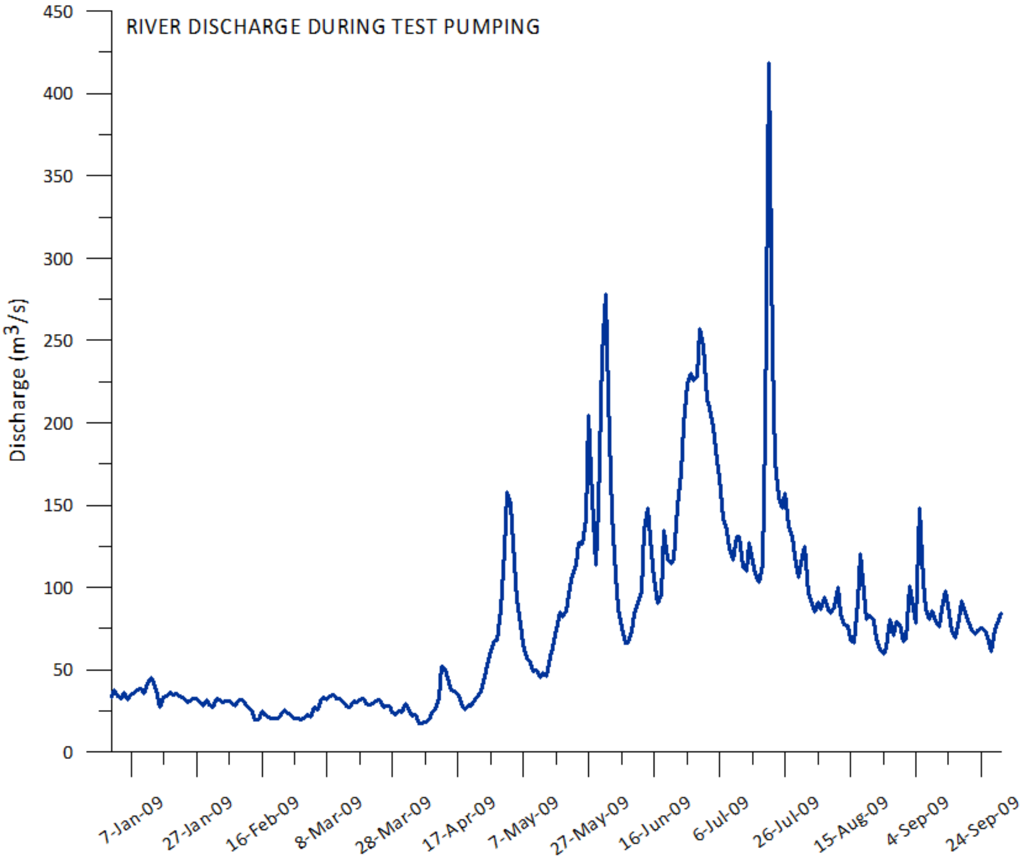


Figure 4: River hydrograph of Driva recorded during the test pumping (Trønderenergi).

From its nucleation area Driva runs north through Drivdalen making a westerly turn into Sunndalen at the community of Oppdal. In the westernmost part of Sunndalen the community of Sunndalsøra marks the location where the river finally drains into the basin of Sunndalsfjorden. The river course displays at some locations great reliefs accounting for a drop in elevation of some 1700 meters. Several river tributaries contribute to the discharge along its course and some are characterized by falls and turbulence. Driva is on the other hand recognized with characteristics of a much calmer meandering river in Sunndalen (NOU 1983).

The watershed runoff in the east and west is in the range of 20 and 60 l/s·km^2, respectively. River discharge is low during winter, but the stretch from Driva power station to Sunndalsøra is recognized with greater winter discharge as an effect of river regulation by means of the redirection of other creeks and rivers to increase discharge (Petterson 1999, Soldal and Rye 1995). Regulation was initiated in 1975 with average discharge values of 75 m^3/s. For comparison discharge was c. 65 m^3/s before regulations were made (Petterson 1999).

1.3.3 ESTIMATES OF EVAPOTRANSPIRATION

Estimates of evapotranspiration can be calculated from values of mean monthly precipitation and temperature displayed in Figure 3, using the Thornthwaite estimation technique (Thornthwaite 1948). Formulas used are described in more detail in *Appendix II*. The estimate brakes down for conditions prevailing in colder regions i.e. Norway, because it does not take into account negative mean monthly temperatures that most often occurs during winter time. Months with a mean value below 0 °C acquires an evapotranspiration of 0. Another weakness is that daily temperature may vary considerably on a monthly basis. January is the only month with a negative value at Sunndalsøra, and by taking this into account the method still has some relevancy. These estimates are close to the value of 476 mm/yr and 485 mm/yr suggested by Soldal and Rye (1995), respectively. Total evapotranspiration is approximately 481 mm/yr (Table 1). Net precipitation is c. 478 mm/yr when subtracting total evapotranspiration from annual precipitation.

Table 1: Mean monthly and annual evapotranspiration based on data from the climatic period of 1961-1990.

Jan	Feb	March	April	May	June	July	Aug	Sept	Oct	Nov	Dec	Annual (mm/yr)
0,0	0,0	15,0	32,8	61,6	76,5	80,7	78,6	62,1	47,4	19,9	6,7	481,3

1.3.4 ESTIMATES OF RUNOFF AND STORAGE

The magnitude of runoff across the flood plain is dependent on surface characteristics, i.e. factors that might act as an obstruction on infiltration. Since the river plane is covered by vegetation to a great extent and the top layers consist of permeable gravelly material, a loss of 10 % due to runoff is chosen. Taking this into account, approximately 432 mm can represent the average (potential) annual amount of water available for infiltration. In other words this is water contributing to aquifer recharge or storage through flood plain infiltration. Groundwater recharge also derives from infiltration of river water from Driva, as well as artificially from the infiltration basin. Recharge from Driva is controlled by groundwater withdrawal (induced recharge) as a gradient is established from the river towards the production wells. Aquifer recharge will be presented in more detail in section 4.7.

1.4 REGIONAL AND LOCAL GEOLOGICAL SETTINGS

1.4.1 TECTONIC EVOLUTION AND PETROLOGY

The study area is included in the geological region known as the Western Gneiss Region (WGR). This region constitutes the north western part of South Norway, occupying an area of approximately 50.000 km^2 between Trondheim to the north and Bergen in the south. Gneisses of Precambrian age dominate, defining a great tectonic window surrounded by allochthonous units associated with overthrust nappes emplaced during the evolution of the Caledonides. The exposure of WGR rocks are believed to be the result of extensional tectonics taking place at least in two stages (Cuthbert et.al. 2000). Several orogenies have left its imprints on the WGR during the Precambrian (Koeneman 1993).

Rock types comprising the basement is dominantly gneissic, varying from granitic, migmatitic to augen orthogneisses of Proterozoic age. Other lithologies of importance are the meta-anorthosites, metamorphic pyroxene syenitic/monzonitic rocks, pelitic schists, quartzites, calc-silicates and marbles. The latter rock types represent a broad assembly of lithologies that are related through fold geometry, often downfolded into the basement orthogneisses (Cuthbert et.al. 2000). With respect to the bedrock map of the area of Sunndalen published by NGU (2009 a), a gross description of the rock types can be outlined on a more local scale. The area is dominated by gneisses, ranging from granitic, migmatitic, dioritic to augen gneiss. Soldal and Rye (1995, therein Sigmond et.al. 1984) described this as an area dominated by migmatitic gneisses. Meta-sediments, quartzites and schists are present, as well as calcerous phyllites, schists and paragneisses. Locally, one can also find gabbros and diorite (Figure 5).

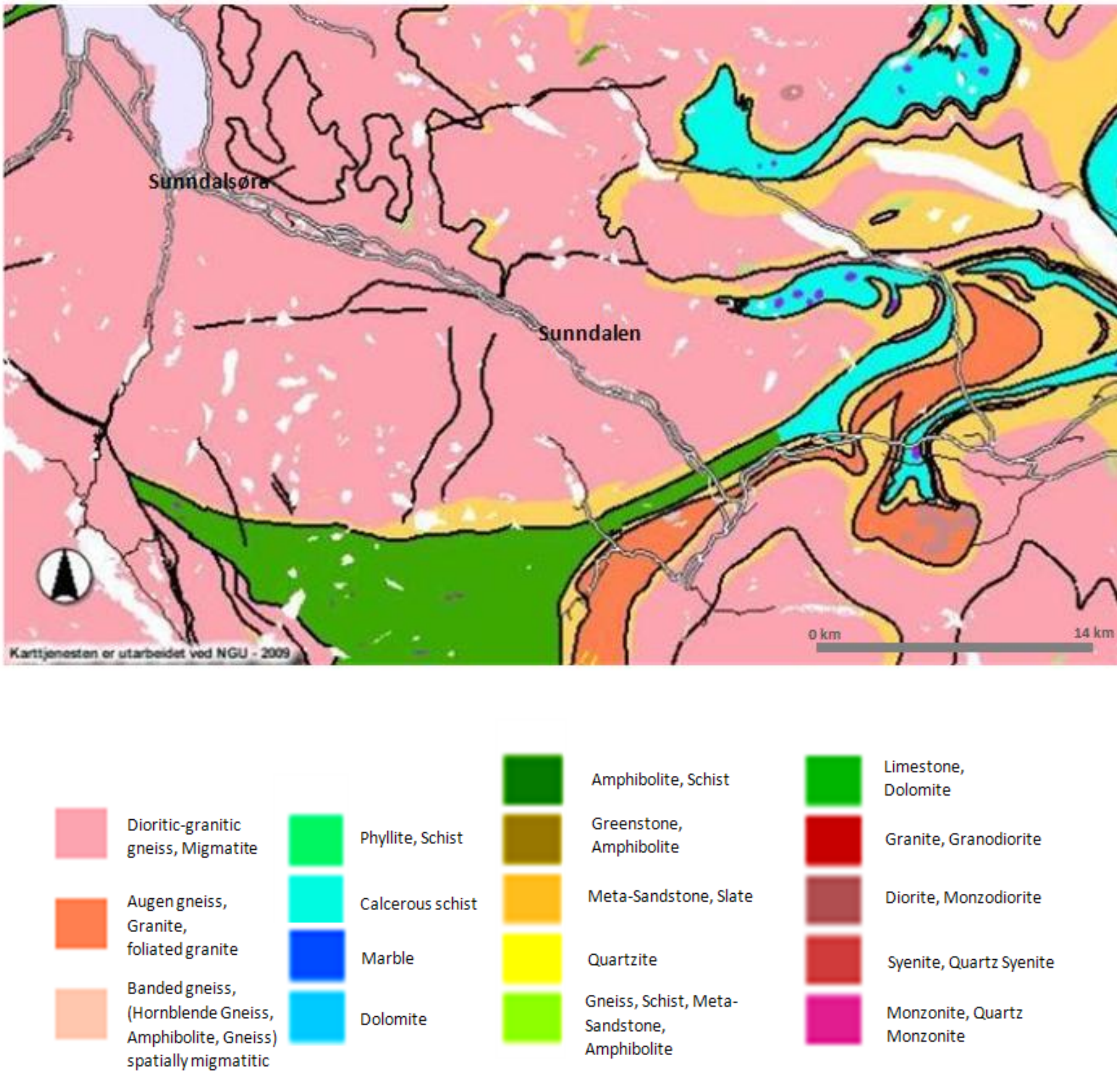


Figure 5: Bedrock map of Sunndalen with adjacent areas (Modified from NGU 2009 a).

1.4.2 QUATERNARY GEOLOGY

The Quaternary period was dominated by larger scale climatic fluctuations that gave rise to several ice ages. During the ice ages, ice sheets nucleated in both the North-American and Scandinavian continents as a response to the cold climate. In the county of Møre and Romsdal and in Sunndalen, one can find numerous landforms and deposits that were formed during this geological time period. In order to give a more comprehensive description of the Quaternary geology in the study area, the following outline will give an overview of both regional and local aspects.

PRIOR TO THE LAST GLACIAL MAXIMUM

Some of the deposits found in Møre og Romsdal county originate from an ice free period that pre-dates the last glacial maximum. These sediments have survived the erosive forces of the last great ice cover, and are older than c. 20.000 years. Strata of this origin are stratigraphically underlying lodgment tills, and have been excavated at e.g. Godøya, Barstadvika and Skorgenes. Sea shells have also been detected in tills in the outermost coastal areas of the Sunnmøre region. Dating reveals that these shells were deposited some 30.000 years ago, and implicates an ice free coast at that time (Figure 6). This ice free period is known as *the Ålesund Interstadial* (Follestad et.al. 1994).

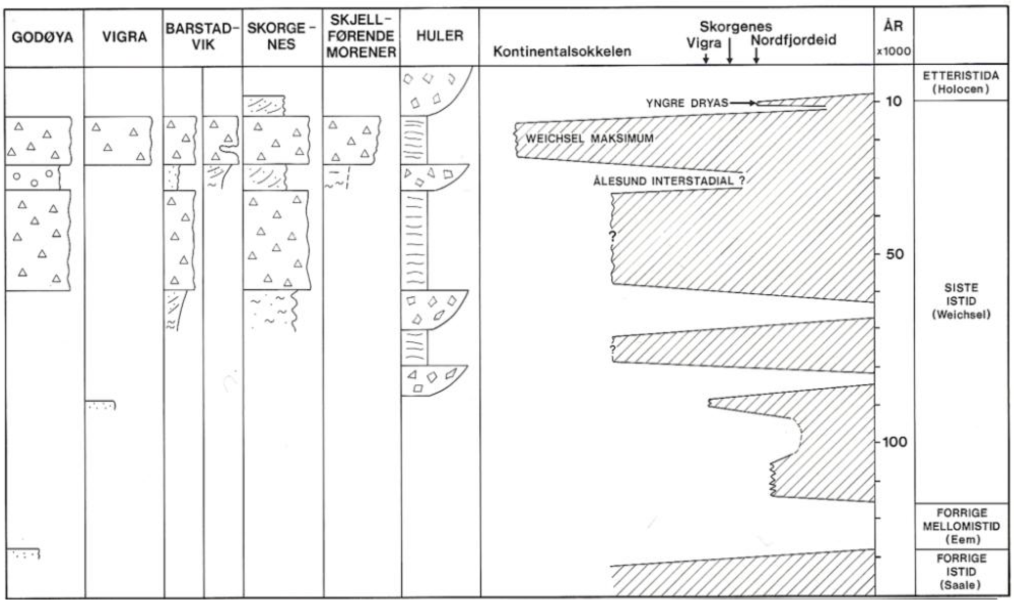


Figure 6: Stadials and interstadials from Saale to Holocene, showing the Ålesund Interstadial and the glacier advances of Weichsel maximum and Younger Dryas (Follestad et.al. 1994).

Certain beds in the stratigraphic record of a coastal cave at Skjonghelleren can be correlated with this interstadial. The faunal contents of these beds suggest that the climate was colder than at present (Larsen et.al. 1987 therein Mangerud 1981). ^{14}C-dating of shells contained in till deposits at Kortgarden in Fannefjorden implicates that the fjords were ice free 27.000 years ago. The last glacial maximum ended c. 20.000 years ago. At that time the Scandinavian ice sheet reached all the way to the continental slope in the west. The dominating glacial movement was to the northwest, but ice also drained through valleys and fjords in a northwest-westerly direction (Follestad et.al. 1994).

LAST GLACIAL MAXIMUM AND DEGLACIATION

Sollid and Sørbel (1981, in Follestad 1987) concludes that the coast experienced the last glaciation 25.000 years ago, and Rise and Rokoengen (1984, in Follestad 1987) states that the western parts of the shelf in the county was ice covered until 13.500 years ago. ^{14}C-dated shell fragments from the neighboring Eide municipality define the initiation of the deglaciation in the county. The ages of these shells suggests that the ice had retrieved 12.800 years ago, starting at some 13.000 years ago in the outer coastal areas (Follestad et.al. 1994). The glacier calved inland in the deeper parts of the fjords, and stagnated when reaching the shallower sills. Marginal moraines located in the outer, northern parts of the county are evidence that the deglaciation took place early in this area. Reaching the sills in Tingvollfjorden and Halsafjorden the glacier stagnated, while calving still was going on in a southerly direction through Kvernesfjorden and Harøyfjorden. Sunndalsfjorden was probably ice covered during the Tingvoll stage (Tingvolltrinnet) c. 12.000 years ago (Follestad 1984, 1985, 1986 in Follestad 1987). When the glacier melted in this area, the sea level rose faster than the inland areas managed to rise. As a result, the sea flooded Sunndalen and the sea level was probably some 120-130 meters higher than at present (NGU 1984). After some time the isostatic uplift of the landmass became more prominent than the sea level rise and glacial deposits was redeposited through erosion by rivers and creeks. Sand, gravel and clay were transported and deposited downstream along the waterway and as deltas in the valley- and fjord areas.

YOUNGER DRYAS

In Younger Dryas (10-11.000 years ago) a period of colder climate again governed, resulting in glacier growth. This climatic regime gave rise to the most prominent deposits in the inner fjord areas. Some reconstructions implicate that marginal moraines in Litldalen and Eresfjorden in Sunndal and Nesset municipalities, respectively, can be correlated. These valleys both display ice marginal deltas and end moraines in the valley sides (Follestad et.al. 1994).

HOLOCENE (PREBOREAL-)

Milder climate about 10.000 years ago resulted in glacial retreat, and marked the beginning of the Holocene. This warming was only interrupted by a minor glacier advance some 9200-9400 years ago. As a result of this advance, marginal moraines were formed in several valleys. In the following, the country moved into a state of warm climate, much warmer than what is experienced today. This resulted in a larger scale melting with the inner parts of the county becoming ice free about 8500 years ago. The glacier ice melted almost entirely, only to give rise to the nucleation of cirque glaciers in the most elevated areas again some 3400 years ago. The response of the deglaciation and associated weight loss resulted in an isostatic uplift of the landmass. This land emergence resulted in the development of beach lines and river terraces because of the increased erosion. The most important beach lines are of Younger Dryas age. Traces of the Tapes-transgression, originating from the post-glacial warming period 6000 years ago, can also be seen in outer coastal areas where the sea level rose faster relatively to the landmass.

DISTRIBUTION OF SEDIMENTS

The Sunndalen area displays close to all sediment types associated with the last ice age and subsequent deglaciation. In the following, a brief description based on Follestad (1987) and Follestad et.al. (1994) will be given with respect to sediment types found in the area. The sedimentary distribution is shown in the Quaternary geology map (Figure 7).

Quaternary geology map: Sunndalen with adjacent areas

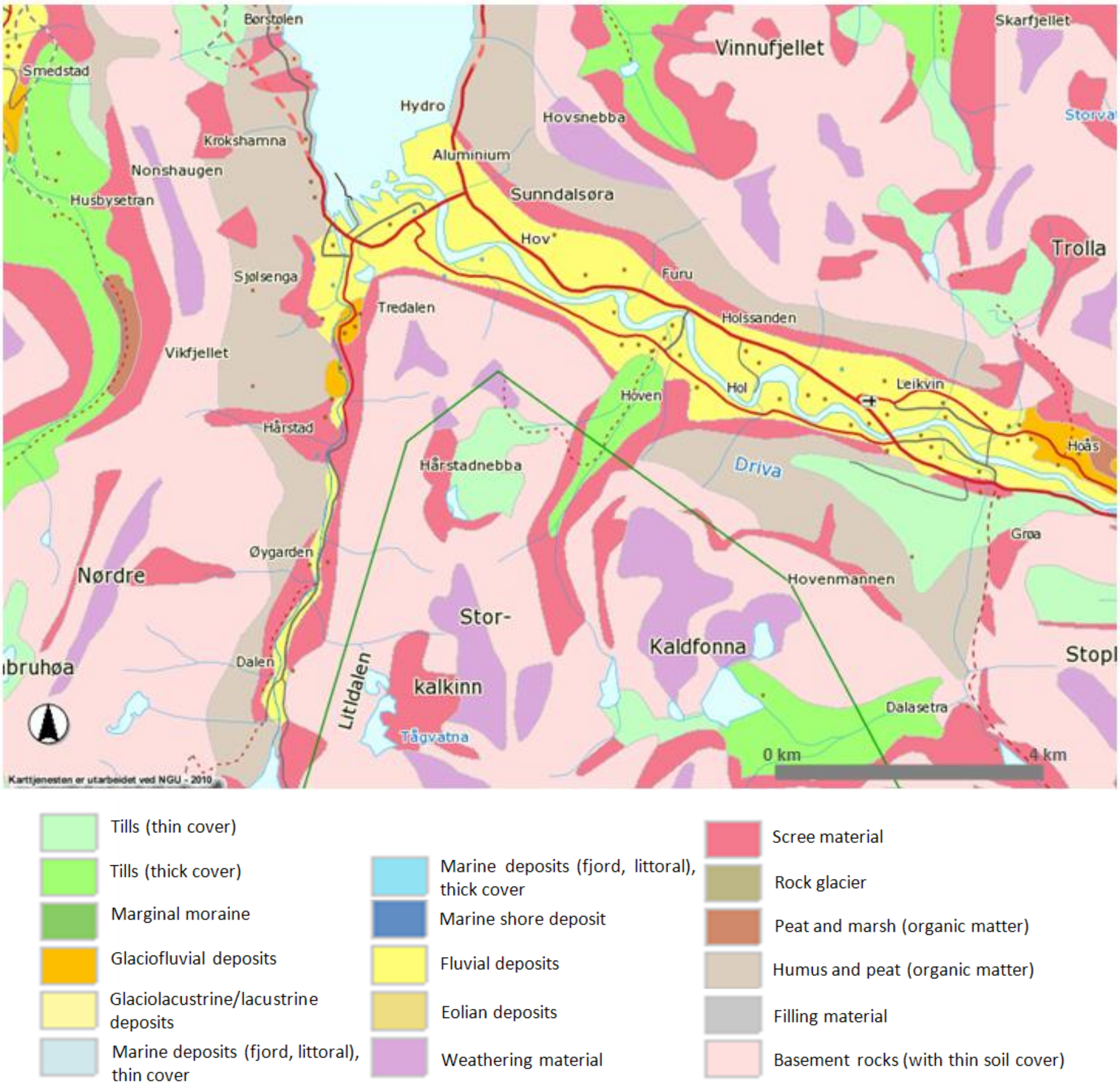


Figure 7: Quaternary geology map of Sunndalen with adjacent areas (Modified from NGU 2009 b).

Tills

Tills are represented as coherent and incoherent and/or thin layer. The former till type is limited to locations at great altitudes and can be found to the north of the valley above elevations of c. 1000 m a.s.l. Incoherent or thin till covers are more widespread and can be located on the south side of Driva. End moraines are linked to glacier advances, e.g. Younger Dryas. This moraine type take on different sizes reflecting advances not only related to the Younger Dryas, but also local glaciation (Follestad og Henningsen 1984). Thicknesses vary and the extent can be from only a few meters to several kilometers.

Glaciofluvial deposits

Glaciofluvial deposits can be seen as terrace landforms in the valley sides of Sunndalen. At Hoås and Gikling one can find some of the most prominent marginal moraines in the valley. The first location represents the westernmost and oldest deposit. Erosion and transport of sediments from these deposits define the most important source material for the fluvial deposits. The marine limit at 145 m a.s.l. has been established on the basis of where one has found a boundary between top sets and foreset beds of the deposit at Hoås.

Marine deposits

Marine deposits are typically found in the valley floor overlain by glaciofluvial and fluvial facies. Marine clay and silt can be seen in vertical profiles of terraces. Some places they show structures derived from river erosion and a cover of talus/rock fall debris. Seismic profiles suggest that the thickness range from 60 meters to 500 meters at the community center of Sunndalsøra.

Littoral deposits

Deposits descending from littoral processes also exist. These are typically thin layers of sorted gravel interbedded with sorted sand showing signs of a glaciofluvial or fluvial origin.

Block fields and talus deposits

Products from weathering processes can be seen in the form of block fields and talus. Block fields are found at greater elevations of 1400-1500 m a.s.l. The thickness of these deposits rarely exceeds one meter. Talus deposits found in the valley sides in the shape of cones and fans. Sediments derived from avalanches and slides can also be observed, often in connection with channeling gullies in the valley sides.

Fluvial deposits

Fluvial deposits include sediments derived from rivers and creeks. These deposits consists mainly of redeposited glaciofluvial sediments, but rock fall debris and sediments derived from other mass movement episodes in the surrounding hill sides might also have been entrained. As the base level dropped in accordance with the relative increase in isostatic uplift, great volumes of sediment were transported downstream finally settling to form river deltas. These deltas can be seen as terraces in the valley. The thickness of the fluvial deposits varies as the facies grades into fine grained marine deposits at varying depths.

Driva has meandered through the valley floor of Sunndalen for a long time, and as a result several oxbow lakes have evolved. These old river beds have a potential for draining river water during floods.

River-dominated delta

Driva meanders through Sunndalen until it reaches Sunndalsfjorden where the river drains through a river-dominated delta. This active delta was formed as a result of the coalescence of deltas formed from the rivers in Sunndalen and Litjdalen. The delta consists of several bed bars and a great number of older river beds forming bed structures. Further out on the delta plain, and in the submarine environment, the river has eroded into the deltaic deposit creating deep cut channels that are traceable to great depths.

1.5 AQUIFER CHARACTERISTICS

1.5.1 HYDROGEOLOGICAL FRAMEWORK

The fluvial aquifer at Sjølandsøra comprises a Quaternary sedimentary sequence resting on top of basement gneisses of Proterozoic age. The aquifer sediments are dominantly sand and pebbles with an upper stratum consisting of predominantly cobbles and pebbles. This upper layer of some 2-3 meters is ascribed the work of a dynamic meandering fluvial system. Foreset beds composed of sand and pebble occur at varying depths representing river delta deposits that prograde in a NW-direction. Values of hydraulic conductivity are in the range of $3{,}5\text{-}7{,}8 \cdot 10^{-4}$ m/s as suggested by investigation wells Ub-1 ad Ub-2 in vicinity of the well area at the works (Asplan Viak 2009). Good hydraulic contact between sediments constituting the unsaturated zone and the sandy pebbles of the saturated zone defines this basin as an open aquifer. A good hydraulic contact also exists towards the river bed of Driva to the E-SE, which constitutes the most important source of aquifer recharge. Groundwater table fluctuations are roughly in the range of 2-5 meters below the terrain surface at some 5-8 m a.s.l., depending on production needs (withdrawal) and recharge (Asplan Viak 2008 b, 2009). Two terrace levels represent the main geomorphologic features of the river plain recording base level drops in Sunndalen. One terrace is situated 2-3 meters above the present water level of Driva (c. 5-6 m a.s.l.), while the second terrace to the east reaches elevations of 7-8 m a.s.l (NGU 1986).

Early investigations conducted by NGU suggest sediments of good hydraulic properties to be traceable down 10-14 meters, and that a grading towards finer sediments takes place at some 20 meters depth (NGU 1986). Subsequent investigations performed by AV have reported thicknesses in the range of 5-20 meters with a general increase in a SW-direction from Driva. The successions of foreset beds vary in thickness from 10 meter close to the river and up to 20 meters to the SW. With respect to the infiltration basin, thicknesses of sediments with good hydraulic properties are registered down to some 12-14 meters to the SE and SW. It became evident that the aquifer is restricted with depth and laterally by finer grained sediments. These fine grained sediments are represented by marine and fluvial sediments, which spatially have been found to contain organic matter (Asplan Viak 2008 b, 2009, 2010). The marine deposits occur at varying depths limiting the aquifer downwards. Fine sand and silt have also been deposited during floods as overbank sediments and/or in ox-bow lakes defining the lateral extension of the groundwater basin. In the case of oxbow lakes, such abandoned meander bends act according to NGU (1986) as hydraulic (negative) boundaries in the basin. These palaeo-channels have also been mapped by NGU (Follestad and Henningsen 1984).

1.5.2 INTRODUCTION TO THE GROUNDWATER QUALITY

GENERAL WATER QUALITY

The groundwater can in general be characterized as soft and acidic. This acidic imprint results in turn in low alkalinity. A relatively low content of dissolved ions is reflected in low electrical conductivity. This is linked to the mineral composition of the aquifer sediments and is also an effect of the short residence time in the aquifer. Despite a relatively short residence time the groundwater is of good quality with respect to microbiological parameters. The groundwater quality is overall good with respect to hygienic factors and meets all requirements related to health related issues, but high concentrations of iron causes problems related to terms of use. On occasion, e.g. during test pumping of investigation wells, hydrogen sulfide gas (H_2S) has been detected (Asplan Viak 2008 b).

HIGH VALUES OF IRON

Sunndalsøra groundwater works has for a long time experienced problems with elevated values of iron in groundwater from one of the production wells. Problems have occurred as a consequence of the withdrawal from the production well (well B) with the most deeply situated well screen. Iron concentrations clearly exceeds the guide values defined in *the Regulation Concerning Water Supply and Water Intended for Human Consumption* (in the following: *the drinking water regulation*s) with some values even > 0,5 mg Fe/l. The high iron content is linked to discoloration and staining at of sinks and light clothing. At the groundwater works problems are related to encrustations in well systems and transmission systems.

Problems with iron have been an issue all since the establishment of the works in the 1990's. Hydrogeological investigations conducted on the river plain at Sjølandsøra prior to establishment, and to this date, have made evident that iron concentrations vary substantially across the river plain. Asplan Viak (1997) reports how the iron content change in relation to the different locations of observation wells on the river plain. A brief description follows (after Asplan Viak 1997):

Iron concentrations were generally low upstream the well area (at observation wells 5 and 6). This was also the case when moving in an easterly and westerly direction relative to well C towards observation wells 11, 12 and 14 (c. 0,03-0,07 mg/l). Between the river bank and well A, observation wells gave low values of iron in the shallow parts of the aquifer, but values increased in the deeper parts in vicinity of the transition to finer grained sediments (c. 0,2-1,0 mg/l). Despite good hydraulic properties of the aquifer sediments in this part, iron concentrations are generally high downstream from wells A and B (observation wells 4, 10 and 17), and the guide value was exceeded even at shallow depths (0,25-0,5 mg/l). Maximum concentrations in this part was some 1,0-1,4 mg/l. In the case of the observation wells located closest to the river bank, high values of iron was related to the transition towards finer grained sediment in the fraction of fine sand. This relationship was encountered during subsequent investigation, e.g. Asplan Viak (2008 b). High values of iron were reported in samples collected from parts of the aquifer dominantly consisting of fine sand and silt. This layer occurs at varying depths in the aquifer, initially suggested by NGU (1992 b).

1.6 SUNNDALSØRA GROUNDWATER WORKS

Sunndalsøra groundwater works is operated by Sunndal municipality and serves together with the Kalken waterworks as public water supplies for the community of Sunndalsøra and surrounding areas. In total, some 5.500 consumers are dependent on potable water from these supplies. An overview is given in the following of Sunndalsøra groundwater works, with Figure 8 making the basis for descriptions. Detailed information on investigations and evaluations involved in the establishment leading to the current configuration of the groundwater works was provided in the introductory sections.

Overview map: Constructions and facilities (2009)

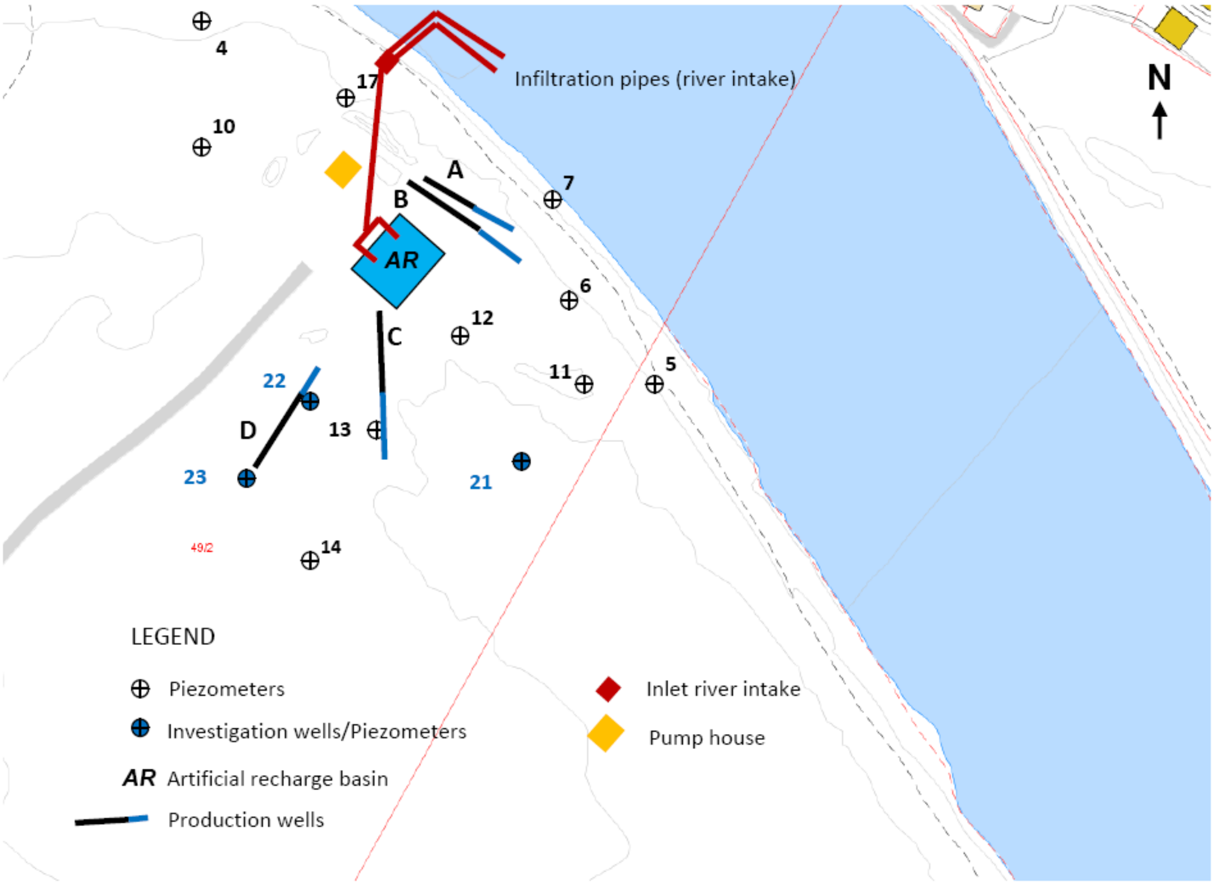


Figure 8: Overview map of Sunndalsøra groundwater works with constructions and facilities (2009).

1.6.1 PRODUCTION WELLS

The groundwater works consists of three tilted production wells (A, B, C). Table 2 shows the well specifications of all wells.

Table 2: Well specifications of production wells A, B and C (Modified from Asplan Viak 2008 a).

Well	Dim. (mm)	Material	Total length	Location screen (m)	Angle of tilt (°)	Depth to filter$_{\text{top}}$ (m)		Depth to filter$_{\text{end}}$ (m)	
						Below groundwater level (m)	Below terrain (m)	Below groundwater level (m)	Below terrain (m)
A	140	PVC	33	21-33	6,6	2,2	5,2	3,6	6,6
B	168	Stainless steel	38	23-38	19,4	7,0	10,0	12,0	15,0
C	168	Stainless steel	40	20-40	12,5	5,0	8,0	9,3	12,3

1.6.2 PUMPS AND SYSTEM OF TRANSMISSION LINES

Wells A, B and C are connected at a pump house at contour +7 where two suction pumps runs by turn. Water is transferred c. 0,4 km to the water treatment facility. These pumps lift the water directly to the elevated basin at Killurda (100 mVs) with a volume of c. 2700 m^3. The basin is placed at contour +84 at the east side of Sunndalsøra, delivering water by gravity directly to consumers at the community center.

1.6.3 WATER TREATMENT FACILITY

The treatment facility was installed in 1992 and is situated on contour +6. Water is delivered directly from the pump house, and water is transmitted directly to the facility without breaking the pressure line. The water treatment is based on three different steps operated in the order as follows:

- UV-disinfection: Secures hygienically safe water.
- $Ca(OH)_2$: Increases the hardness of the water.
- CO_2: Aids the dissolution of $Ca(OH)_2$, which in turn leads to an increase in pH and hardness.

1.6.4 INFILTRATION BASIN

The infiltration basin was constructed in 1997 between wells B and C as an action to reduce iron concentrations. The area occupied by the basin is rectangular and measured c. 270 m^2 (15×18 m) up until recently when the basin was extended some 50 % (Asplan Viak 2010). Water is pumped from a river intake placed in the bottom of the river. A filter at the intake obstructs e.g. sediments from entering the basin and transfers the water through a drain pipe connecting the intake with the basin. Water enters the basin through a cascade placed in the basin center. Infiltrated water is subsequently filtered through a filter mass of marble sand with the purpose of increasing groundwater pH. Excess water drains towards the river through a weir, and accounts for c. 5 l/s. Some 40 l/s are supposedly pumped to the basin, but by taking into account what goes in the weir this should result in an effective infiltration of c. 35 l/s.

The infiltration basin was in use only for some time after the establishment in 1997. After negative experiences with increased iron contents, the marble sand was regenerated and the basin was once again put into operation at the water works in 2005. The effect on iron content is uncertain because of lack of data from the period prior to construction. There are no clear evidences for the effect of marble sand on pH.

2. THEORY OF IRON AND INTERACTIONS WITH GROUNDWATER

Occurrences of iron will be presented from a geological and geochemical perspective, with greater emphasize on interactions between iron and the hydrosphere. Geochemical factors influencing solubility and lability of iron will be in focus. Problems experienced with high values of iron in groundwater utilized for household purposes are presented and relevant treatment methods aiding the removal of iron will be described with respect to both conventional and in-situ methods.

2.1 IRON FROM A GEOLOGICAL PERSPECTIVE

Iron (Fe) is considered a refractory element in terms of geochemistry, a siderophile element found in greatest abundance in the Earth's core. By weight percent it is the most abundant element on Earth. With respect to the crust on the other hand, iron represents the fourth most abundant element (4,7 %) following oxygen, silicon and aluminum (Ottesen et.al. 2000, Brattli 2009). The siderophile characteristic of iron explains why this element accumulates in the mantle and core. Iron is often associated with elements like cobolt, cromium, copper, magnesium, manganese, nickel, scandium, zink, titanium and vanadium (Albarède 2003, Ottesen et.al. 2000).

Iron occurs in its most reduced form in the Earth's core, as elemental iron (Fe^0). The mantle composition is associated with greater fractions of iron in a higher oxidation state as ferrous iron (Fe^{2+}). Ferrous iron is found here in most silicate minerals substituting for magnesium. In the upper mantle it is contained in the minerals olivine, pyroxene, garnet and amphibole. The crust contains igneous rocks that include e.g. amphibole and biotite, as well as containing ferrous oxides (hematite, ilmenite and pyrrhotite) and minerals with iron in its higher oxidation state. Alkali feldspars host for example ferric iron (Fe^{3+}), which is an important rock-forming mineral in e.g. granites.

In nature, iron in its pure elemental form is considered rare. Minerals containing iron is besides the mafic silicates earlier mentioned, also oxides (e.g. hematite, magnetite, ilmenite), carbonates (siderite) and sulfides (e.g. pyrite). Upon exhumation to the Earth's surface and exposure to standard state conditions, iron may transform into iron-oxyhydroxides as a result of chemical weathering of primary minerals of iron-bearing bedrock. One important chemical reaction process is oxidation that leads to the formation of ferric iron and associated iron oxyhydroxide products, e.g. goethite, hematite and limonite. These minerals are common components of sediments and soils (Albarède 2003). Upon weathering, ferrous iron is considered mobile, while ferric iron is typically immobile. Iron also shows an affinity for the formation of bonds to bacteria that are linked to the affinity of transition metals to polymeric material and the metal-binding capacity of bacteria (Konhauser 1997).

The fate of iron and its speciation in an aquatic environment will be outlined in the following with emphasize on aspects of solubility and lability.

2.2 IRON SOLUBILITY AND LABILITY

Iron solubility and lability (mobility and bioavailability) are dependent on several factors. Both geological and hydrological factors are important, but different physical, chemical and biological processes also dominate. Chemical weathering leads to disintegration of bedrock, which in turn can provide dissolved iron. Two different weathering processes might aid the dissolution process. Upon dissolution from the solid phase, the stability and solubility of iron is to a great extent determined by oxidation state, acidity and contents of organic matter. The different factors will be outlined in the following.

2.2.1 CHEMICAL WEATHERING PROCESSES

In contact with water or air, iron may dissolve from iron-bearing minerals. Weathering of bedrock is considered the most important source for dissolved iron in groundwater, with two possible contributing chemical weathering processes:

- Incongruent dissolution (hydrolysis)
- Redox-reactions

INCONGRUENT DISSOLUTION

Incongruent dissolution is the most important chemical weathering process responsible for the decomposition of silicate minerals. This reaction disintegrate primary silicate minerals with the consumption of H^+-ions, producing cations, silica acid and OH^--ions. It can provide both ferrous and ferric iron, depending on the rock type and minerals in question. In this context, the weathering of dark (mafic) silicates is the most relevant. Mafic minerals containing iron (e.g. olivine, pyroxenes, amphibolites and biotite) are more easily weathered than intermediate and felsic minerals (e.g. feldspars, muscovite and quartz). This reaction series follows the weathering series established by Goldich (Appelo and Postma 2005, therein Goldich 1938). BOX 1 presents some relevant reactions involving iron:

BOX 1. SILICATE WEATHERING AND THE FATE OF IRON

Example I: Olivine, oxidizing conditions (after Brattli 2009):

$2MgFeSiO_4 + 1/2O_2 + 9H_2O + 4CO_2 \rightarrow Fe(OH)_3\ (s) + 2Mg^{2+}\ (aq) + 4HCO_3^{-} + 2H_4SiO_4$

Example II: Olivine, reduced conditions (after Brattli 2009):

$MgFeSiO_4 + 4H_2O \rightarrow Mg^{2+}\ (aq) + Fe^{2+}\ (aq) + 4OH^{-} + H_4SiO_4$

Example III: Pyroxene weathering- combined oxidation and hydrolysis close to the surface (after Brattli 2009):

$2CaFeSi_2O_6 + 1/2O_2 + 13H_2O \rightarrow 2Ca^{2+}\ (aq) + 2Fe(OH)_3\ (s) + 4OH^{-} + 4H_4SiO_4$

Example IV: Weathering of biotite (after Appelo and Postma 2005):

$2K(Mg_2Fe)(AlSi_3)O_{10}(OH)_2 + 10H^{+} + 1/2O_2 + 7H_2O \rightarrow Al_2Si_2O_5(OH)_4 + 2K^{+} + 4Mg^{2+} + 2Fe(OH)_3\ (s) + 4H_4SiO_4$

These examples also illustrates that silicate weathering often is combined with redox-reactions.

REDOX-REACTIONS

Of great importance to iron solubility is the redox-state of the aquatic environment. Dissolution of iron from the solid phase to groundwater is favored in reducing environments (anaerobic), meaning that mineral dissolution processes mobilizes iron when in contact with waters of low redox potential. Anaerobic conditions favor the formation of ferrous iron, which is highly soluble in water given pH is in the range of 5-8 (Appelo and Postma 2005). This acidity range of water is often representative of groundwater. The oxidized form however, will dominate at higher redox potentials represented by insoluble ferric iron. In comparison with surface waters, groundwater is generally more susceptible to high concentrations of iron as a result of a deficiency of dissolved oxygen. According to Droste (1997), redox-reactions are generally slower than acid-base reactions. Oxidation and hydrolysis often occur at the same time when silicates weather. This is the case for the weathering of olivine in an oxidizing environment (BOX 1). BOX 2 lists some redox-reactions that involve iron:

BOX 2. REDOX-REACTIONS INVOLVING IRON

Example I: The total reaction involved in the oxidation of pyrite (after Brattli 2009):

$FeS_2 + 15/4\ O_2 + 5/2\ H_2O \rightarrow FeOOH + 2H_2SO_4$

Example II: Ferric iron aiding the oxidation of pyrite under reducing conditions (after Brattli 2009):

$FeS_2 + 14Fe^{3+} + 8H_2O \rightarrow 15Fe^{2+} + SO_4^{2-} + 16H^{+}$

2.2.2 REDOX-POTENTIAL AND ACIDITY

In the previous paragraphs of this chapter it was illustrated that iron dissolution processes are dependent on the accessibility to acids and oxygen. The redox-reaction determining iron species distribution is accompanied by an electron transfer and a change in H^+-ion activity. This is illustrated with the general reaction below (after Hallberg and Martinell 1976):

$$3H_2O + Fe^{2+} \leftrightarrow Fe^{3+} + 3OH^- + e^- + 3H^+$$

This equation shows the dependency of redox-state (Eh) and degree of acidity (pH) on iron solubility. It is clear that a chemical regime favorable of iron dissolution is characterized by an increased electron (e^-) and H^+-ion activity. In other words, under these conditions the reaction will proceed to the left; ferrous iron will be the dominating species in reduced and sufficiently acidic groundwaters. (However, this reaction proceed along with silicate weathering, which on the other hand contributes to a consumption of H^+-ions.)

The relationship between different iron species can be presented in stability diagrams (Figure 9) with pH on the abscissa and Eh (and pe) on the ordinate scale:

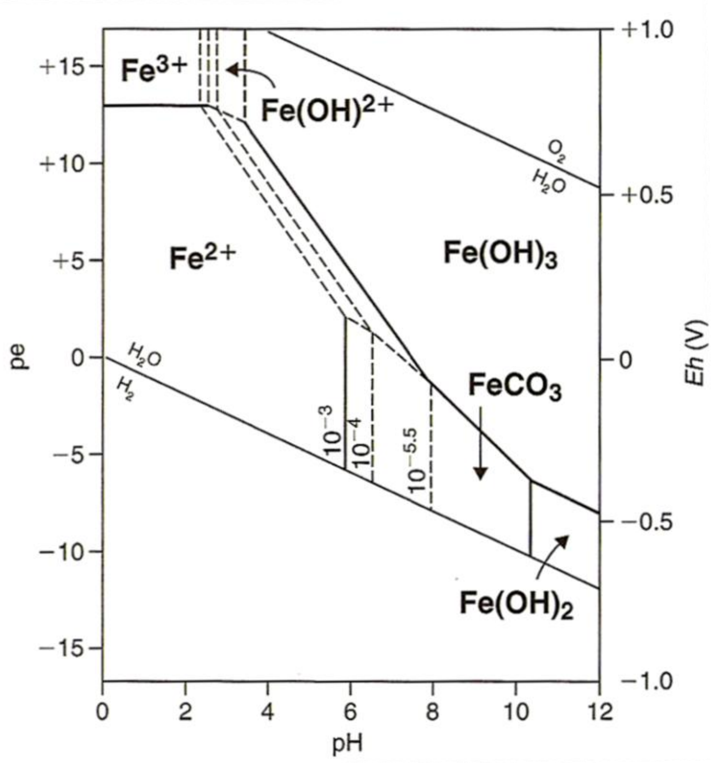


Figure 9: Stability diagram of iron species with respect to pH and Eh (Appelo and Postma 2005).

BOX 3 illustrates with an example how redox-conditions can change in an aquifer as a response to decomposition of organic matter:

BOX 3. REDUCING CONDITIONS CAUSED BY THE DECOMPOSITION OF ORGANIC MATTER

The contents of organic matter in aquifer sediments in the solid phase (SOM) or as dissolved organic content (DOC) are important in the context of iron solubility. This is because of the great reduction capacity of organic matter. Decomposition of organics incorporated in the aquifer will demand oxygen, resulting in consumption of oxygen and the production of carbon dioxide and water. The general equation (after e.g. Stumm and Morgan 1996):

$(CH_2O)_x$ (organic matter) + $O_2 \rightarrow CO_2 + H_2O$

This situation might result in an oxygen deficit establishing reducing conditions, which favors iron dissolution when the concentration of dissolved CO_2 is high and as long as pH is sufficiently low.

2.2.3 SORPTION AND COMPLEXATION

Humic substances play an important role in mobilizing iron in groundwater. Organic matter often occurs as colloids that can function as complexing agents with the effect of binding iron. This makes iron mobile and bioavailable by means of enabling sorption in the form of adsorption (Brattli 2009, Benjamin 2002). Also, Konhauser (1997) points out that relative to other trace metals, iron to a greater extent tend to be bound to organic sites. Metal complexes with iron as central ion can also form. This involves bonds being formed between iron as a central ion and different ligands, i.e. Cl, SO_4 and OH (Fitts 2002).

2.3 PROBLEMS RELATED TO HIGH VALUES OF IRON IN GROUNDWATER

2.3.1 INTRODUCTION

Problems with iron most often occur when reduced groundwater, containing high values of dissolved iron, is introduced to an aerobic environment. In such an environment, there is a potential that iron precipitates given concentrations of dissolved oxygen are more than 1-2 mg/l (Government of Oregon 2009). Relevant situations resulting in precipitation occur when an interaction is allowed with air in the water tap, oxygen-rich water (shallow groundwater) in the aquifer or in the well or transmission system.

The guide value for iron content in potable water is not related to negative health effects, but is defined as a response to problems experienced with high iron contents in relation to groundwater utilization. A water treatment step is as a result of this often required as the only action to reduce the high iron concentrations and associated problems. Sufficient dissolution of free oxygen in groundwater with high levels of iron will in most cases cause a reduction of the groundwater quality to a level less satisfactory for use. Still, iron is essential to plants and animals in low concentrations, which usually are found in drinking water. The element is vital for transporting oxygen through blood and during the process of photosynthesis (Brattli 2009), as well as for certain bacteria that are dependent on the oxidation of iron for their metabolism (NGU 2009 c).

GUIDE VALUES

Good drinking water should have a concentration of iron of less than 0,1 Fe/l (NGU 1991 a), but according to the Norwegian legal framework the acceptable guide value goes up to 0,2 mg Fe/l as stated in *the Regulation Concerning Water Supply and Water Intended for Human Consumption*

(Drinking water regulations). The Worlds Health Organization (WHO) and other nations on the other hand, e.g. USA, Canada and Finland, have established the acceptable limit at 0,3 mg Fe/l (Droste 1997).

A WORLDWIDE PROBLEM

Problems with iron concerning groundwater extraction for drinking water purposes is a relevant situation worldwide (see e.g. Teunissen et.al. 2008, NGU 1991 a, Carlson and Schwertmann 1987, Maogong 1988, Houben 2004). According to NGU (1991 a), iron is the most frequently reported parameter in groundwater causing problems related to use, and it is reported that iron occur in concentrations exceeding the guide value in c. 17 % of water wells that are placed in unconsolidated aquifers (NGU 2009 c). It is also reported that c. 25 % of the groundwater used in Sweden is considered unsuitable for direct use as potable water as a consequence of high iron contents. Groundwaters with high values of iron will require one or more treatment steps (methods) in order to reduce concentrations and to provide good drinking water (Hallberg and Martinell 1976).

RELATIONS TO AQUIFER PROPERTIES

When establishing water wells for groundwater extraction, it is hard to predict whether iron will cause any problems. The challenge lies in the predictability of locations and settings resulting in high concentrations of iron (NGU 1991 a). Hallberg and Martinell (1976) points out that groundwater extracted from different wells located only few meters apart in the same geological formation can produce water with different quality with respect to iron contents.

A useful example was presented in Hallberg and Martinell (1976), illustrating this problem and its complexity, showing that two wells only some ten meters apart in an esker produced water with different quality with respect to iron. High iron contents in only one of the wells were found to be a result of one of the wells penetrating a reduced part of the aquifer. Such precipitating zones have been reported also other places.

Hallberg and Martinell (1976) points out the dependence of aquifer properties on iron content. It is emphasized that besides mineral composition, the amount of dissolved oxygen, degree of permeability of strata and groundwater flow are the predominant factors controlling the occurrence of iron in aquifers. Dissolved oxygen is important in that it will increase along with recharge derived from e.g. rain water, melted snow and ice. As a consequence, lowered iron contents will result. Permeability of strata may vary from place to place in the aquifer and from one aquifer to another, in ways that often obstruct the seepage of surface water and consequently impede mixing with oxygen-rich water. Groundwaters below impermeable layers, e.g. clay and other fine grained strata, are often more stagnant and iron contents are often elevated (NGU 1991 c). In the case of river plains, groundwater flow on river plains proceeds with low gradients and flow is relatively slow (Colleuille et.al. 2004). Generally, one can expect areas with greater topography and coarser sediments to be more beneficial for groundwater withdrawal as groundwater flow is faster.

It is possible to distinguish aquifer types that more frequently causes high iron contents, e.g. by comparison of fluvial deposits with glaciofluvial deposits. Sediments of fluvial systems usually contain fine grained sediments (fine sand, silt) derived from episodes of flooding. Entrainment of this fine grained sediment occurs on the flood plain when the energy of flow is sufficiently low, consequently allowing the settlement of finer sediments. This sedimentary process is also a very important mechanism for the introduction of organic matter in river plains (Colleuille et.al. 2004, Thoresen 2009). Hartog et.al. (2005) point out the importance of organic matter in the form of SOM in the establishment of reducing conditions in aquifers. This form of organic matter is considered one of the most ubiquitous reductants of ferric iron and other oxidized chemical species in groundwater (BOX 3).

Compared to *hardness* of the water and associated high concentrations of calcium, which can be correlated with calcerous bedrock found e.g. in relation to the Caledonides, there is no such link between high iron contents and regional geology. In the case of iron, distinct aquifer properties seems to play a major role.

2.3.2 PRESENTATION OF AN ASSEMBLAGE OF PROBLEMS

High concentrations of iron in groundwater used in a public water supply can be experienced as a single problem or as a combination of several problems. The most relevant problems with iron are described in this section, and can be related to the following:

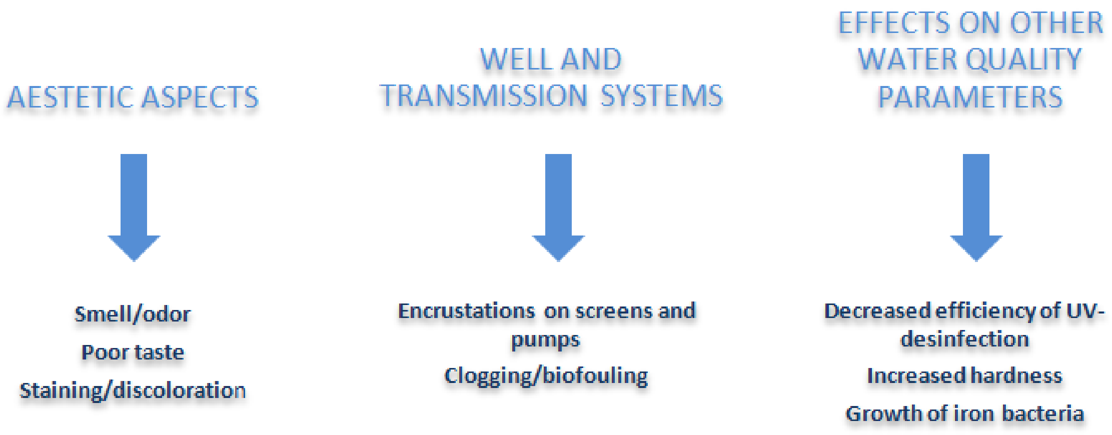


Figure 10: Flow sheet displaying the diversity of problems that might occur in relation to high contents of iron in groundwater used for (public) water supply.

AESTHETIC ASPECTS

The precipitation of iron typically results in staining of light clothing, textiles and plumbing fixtures. A distinct metallic taste and smell is also characteristic, often accompanied by a smell of sulfur in the form of hydrogen sulfide (H_2S). The problems listed here are classified as aesthetic parameters as the reduced groundwater quality is possible to sense by the users (NGU 1991 a, Droste 1997).

WELLS- AND TRANSMISSION SYSTEMS

Wells, well screens and other components of the distribution system are also vulnerable to the formation of iron encrustations caused by ferric iron hydroxides, as noted by Houben (2004). Components of a groundwater works that interacts with reduced groundwater are exposed in relation to the development of iron oxide encrustations. It was shown by Houben (2004) that initial iron oxide content in the water supply system is of great relevance for autocatalysis, setting the stage for an even more efficient iron hydroxide precipitation. This might explain the development of well yield with time, which is little influenced by precipitation at first; after some time with precipitation and establishment of encrustations, the continuing precipitation is accelerated by the already existing iron hydroxide. Regeneration can aid in increasing hydraulic yields, but if the step turns out incomplete autocatalysis proceeds with greater efficiency and continues building up encrustations. Bacterial communities thriving in the encrustation masses can also contribute to the process of biofouling (e.g. Tuhela et.al. 1992, Folkehelseinstituttet 2008, NGU 1991 a). Groundwater extracted from encrusted wells often has a decreased quality (Tuhela et.al. 1992). Water quality can also be heavily reduced as a result of precipitation or post-treatment settling of dissolved iron in the form of iron hydroxide particles in the distribution system (Teunissen et.al. 2008). Colter and Mahler (2005) also reports that iron precipitate in corroded pipes tend to break free during episodes of turbulence and cause rusty tap water.

INTERACTIONS WITH PHYSICO-CHEMICAL AND MICROBIOLOIGCAL PARAMETERS

The corrosiveness and hardness of groundwater is also highly dependent on the content of dissolved iron. High values of iron often results in harder water, as well as causing water to react more aggressively with metallic installations (Droste 1997). Negative effects from the formation of iron encrustations are an important link to increased corrosion of metal components in the water supply system (Tuhela et.al. 1992). The color of groundwater is dependent on organic content, but if iron is present in sufficient amounts the color can be even more distinct (Brattli 2009).

If UV-radiation is applied as disinfectant in the water treatment, high concentrations of iron may reduce the disinfection efficiency as a response to reduced transmission. This might cause a situation with microorganisms reaching the consumers instead of being immobilized by the UV-radiation. Fine particular rust reported equal to a turbidity value of 1 FNU reduces the transmission of rays to 97 %. High iron contents may also impact on the operational aspect, in that the quartz glass used might be coated by burnt iron particles (Folkehelseinstituttet 2008 b).

Certain bacteria can oxidize ferrous iron with the result of clogging e.g. wells. These bacteria do not necessarily represent any health threat, but other bacteria that thrive in their biomass may on the other hand (NGU 2009 c).

2.4 TREATMENT METHODS FOR IRON REMOVAL

Several treatment methods can be applied to reduce high values of iron in groundwater administrated for use in public water supply. A water treatment method and process will function as a modifier of the physical, chemical and/or microbiological regime of a specific source of groundwater, and is a remedial action chosen to maintain or improve water quality, i.e. to provide a good drinking water quality in terms of use and satisfactory hygienic conditions.

When discriminating between different methods and configurations of treatment facilities, general factors related to the character of the water source, quality of the raw water, the size of the groundwater works, operational factors and economy, all have to be taken into consideration (Folkehelseinstituttet 2008 a). At present, 19 groundwater works in Norway reports treating the groundwater for iron and manganese contents (Folkehelseinstituttet 2008 a, NGU 1991 a). Manganese removal follows the same principles but it is considered a much more complicated process (Droste 1997, Folkehelseinstituttet 2008 a).

Removing iron from drinking water can be achieved with treatment facilities based on conventional and/or in-situ methods. Removal of iron from groundwater is based on the redox-reaction forming insoluble ferric iron from soluble ferrous iron, where the superior aim is to accelerate the oxidation process. The outcome of this reaction is the formation of the less soluble iron oxyhydroxide. BOX 4 lists the different methods that will be described:

BOX 4. TREATMENT METHODS FOR IRON REMOVAL/REDUCTION IN GROUNDWATER

The methods involved in iron removal are based on the oxidation of ferrous iron to ferric iron as illustrated in the equation (after Droste 1997):

$Fe(OH)_2 \rightarrow Fe^{2+} \rightarrow Fe^{3+} \rightarrow Fe(OH)_3$

Conventional methods
Aeration
Oxidation aided by addition of chemicals and subsequent filtering
Greensand
Bacteriological and chemical oxidation/biofiltration
Ion-exchange
(Biological removal)
Sequestration

In-situ methods
Vyredox
Artificial recharge basins

2.4.1 CONVENTIONAL METHODS

AERATION

This method is one of the most frequently used methods for iron removal. Removal is achieved through the stepwise process of oxidation, precipitation and a final removal of the precipitate through a stage of coagulation/sedimentation and filtration. Oxygen functions as the only oxidizing media in the transformation of ferrous iron to ferric iron. The reaction results in the precipitation of highly insoluble iron hydroxide, which can be removed by filtering through e.g. a sand filter. The method can be adjusted for different settings, and if concentrations of iron are low the equipment can be installed in the well itself (Folkehelseinstituttet 2008 a). Several different aeration devices can be applied to assess the iron removal. Gravity aerators with the configuration of cascade-, inclined apron-, towers with countercurrent flow of air/water- or stacked perforated plates are considered effective for this purpose (Figure 11). These devices all aim at separating the water mass into smaller volumes that will allow a more efficient mass transfer between the increased areas of water and the oxidant. Aeration can also be achieved using spray- and diffused aerators, but different designs will be prevailing (Droste 1997).

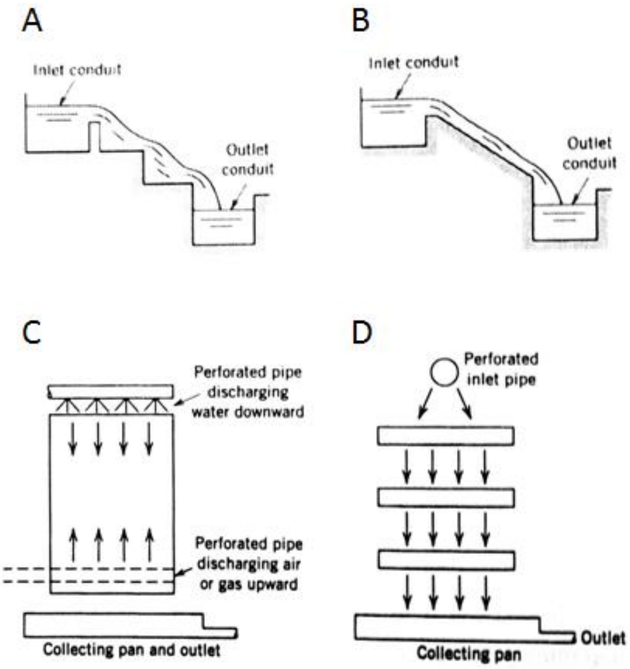


Figure 11: Gravity aerators may aid the removal of iron upon mixing air with groundwater. A: cascade, B: inclined apron, C: tower and D: stacked perforated plates (Modified from Droste 1997).

BOX 5 provides an example on how aeration is utilized at a Norwegian groundwater works:

BOX 5. EXAMPLE: AERATION AND FILTERING AT EIDE WATER WORKS

Eide water works in Hemne municipality experiences high concentrations of iron, as well as elevated values of manganese and ammonium. This has been linked to the reducing conditions in the aquifer and a low pH that is also favorable of iron dissolution.

In order to reduce concentrations of these compounds the groundwater is aerated and subsequently filtered through a marble filter. This treatment step reduces iron contents through the mixing with oxygen, while the marble contributes to an increase in pH, alkalinity and calcium (Asplan Viak 2007).

The aeration method utilizes oxygen as the only oxidizing media. This general reaction illustrates how aeration affects ferrous iron (after Droste 1997):

$4Fe^{2+} + O_2 + 10H_2O \rightarrow 4Fe(OH)_3 + 8H^+$

OXIDATION WITH CHEMICALS AND SUBSEQUENT FILTERING

Several oxidants are available for the process of oxidizing ferrous iron. The method is most often used in situations where iron concentrations are very high and a more aggressive approach is needed. Effective oxidation can be achieved with the aid of potassium permanganate ($KMnO_4$), ozon (O_3), chlorine (Cl_2) and chlorine dioxide (ClO_2). This method is dependent on a preferable pH, which implies that a regulation of pH might be needed before the process is initiated. As with the aeration process, a sand filter can be applied for the separation of the iron precipitate (Folkehelseinstituttet 2008). According to Folkehelseinstituttet (2008 a) only one water works use ozon as a pre-treatment step for the process of iron/manganese-removal, and the filtering step is made with a filter of the greensand/potassium permanganate type. None reports exists on the use of $KMnO_4$ as oxidant.

GREENSAND

The method is based on filtration of water through greensand (glauconite) of the type zeolite. The zeolite is a greenish mineral, commonly found as granules with the composition $(K,Na,Ca)_{1,2-2}$-$(Fe^{3+},Al,Fe,Mg)_4Si_{7-7,6}Al_{1-1,4}O_2(OH)_4 \cdot nH_2O$. Water is filtered through the greensand that is coated with adsorbed manganese, which catalyzes the oxidization. Sorption takes place at the same time as iron is oxidized. The sorbed, oxidized iron can be removed through regeneration of the filter mass which is accomplished with the addition of potassium permanganate. The process of regeneration re-establishes the oxidizing capacity of the greensand (Droste 1997).

The greensand is most often placed in pressure filters with permanganate or other oxidants frequently added. Dissolved oxygen does not influence on the process, but a pH above 7,5 is preferred. Backwashing is required from time to time in order to avoid clogging, and this can be accomplished during regeneration (Droste 1997). Folkehelseinstituttet (2008 a) states that the method so far only has been used in thirteen works, all considered smaller scale water works.

BOX 6. PROCESSES INVOLVED IN THE USE OF OXIDIZING CHEMICALS AND GREENSAND

Example I: Relevant reactions involving the use of permanganate, chlorine and chlorine dioxide are (after Droste 1997):

1. **Permanganate:** $3Fe^{2+} + MnO4^{-} + 4H^{+} \rightarrow 3Fe^{3+} + MnO_2 + 2H_2O$
2. **Chlorine:** $2Fe^{2+} + Cl_2 + 6H_2O \rightarrow 2Fe(OH)_3 + 6H^{+} + 2Cl^{-}$
3. **Chlorine Dioxide:** $Fe^{2+} + ClO_2 + 3H_2O \rightarrow Fe(OH)_3 + ClO_2^{-} + 3H^{+}$

Example II: Processes of greensand (***Z*** indicates the zeolite/greensand):

1. Sorption and oxidation (after Droste 1997): $\mathbf{Z}\text{-}MnO_2 + \{Fe^{2+}, Mn^{2+}\} \rightarrow \mathbf{Z}\text{-}Mn_2O_3 + \{Fe^{3+}, Mn^{3+}, Mn^{4+}\}$
2. Regeneration of greensand (after Droste 1997): $\mathbf{Z}\text{-}Mn_2O_3 + KMnO_4 \rightarrow \mathbf{Z}\text{-} MnO_2$

Four additional methods will be presented, but their relevancy towards iron removal in the context of groundwater treatment is not high.

ION-EXCHANGE

Ion-exchange relies on filter types that remove iron through ion exchanging processes. No reports indicate that this method is in use at any Norwegian groundwater works at present (Folkehelseinstituttet 2008 a).

BACTERIOLOGICAL AND CHEMICAL OXIDATION/BIOFILTRATION

The process combines both chemical and bacteriological reactions for the oxidation of iron. Air is added to the raw water with an ejector and slowly filtered. The infiltration velocity has to be low in order to aid the establishment of a microbiological mat in the filter. In this way, the microbiological conditions in the filter act as an additional oxidation step. The slow-filters are underlined with an impermeable cover to avoid that recently treated water infiltrates the ground. The experience with this form of iron removal is scarce in Norway, but in Finland the process has been used as an alternative to the VYREDOX-method if the ground conditions are unfavorable of a good treatment otherwise (Folkehelseinstituttet 2008 a). For this reason, the method could also be defined as an in-situ method.

BIOLOGICAL REMOVAL

In order to remove iron biologically one is dependant of a successful establishment of a filter that supports autotrophic iron-oxidizing bacteria. This method is not considered relevant, but further information is provided in Droste (1997).

SEQUESTRATION

With the process of sequestration one can bind iron with the addition of different sequestering agents, e.g. phosphates, sodium silicates and polyphosphates. This does not involve a physical removal of iron like the other methods, which utilizes the filtering effect of using e.g. a sand filter. The chemical reaction creates so called polymeric colloid complexes that are unable to cause turbidity and color in the transfer system. The most effective agent for the purpose of sequestering iron is silicates (Droste 1997).

2.4.2 IN-SITU METHODS

Two quality improving actions are considered applicable in the in-situ removal of iron (BOX 7). Both methods are based on re-infiltration of groundwater, usually following a pre-treatment aeration step. If wells are used, filtration is usually also required prior to infiltration. These artificial infiltration methods aid the oxidation of dissolved iron, causing the formation of insoluble oxyhydroxides that are retained in the aquifer or filter mass, preventing iron from precipitating in the well system (NGU 1991 a).

BOX 7. IN-SITU METHODS FOR REMOVAL/REDUCTION OF IRON

Example I: VYREDOX-method in use at Elverum water works:

Elverum water works supplies c. 12 900 people in the municipality of Elverum. The works experiences problems with high concentration of iron and manganese, and the groundwater is consequently treated in-situ. The VYREDOX-method has been applied with great success, and progresses as previously described. Subsequent pH-adjustment is made (Colleuille et.al. 2004, NGU 1991 b).

Example II: Infiltration basin covered with marble sand for additional pH-adjustment at Sunndalsøra groundwater works:

One of the aims of constructing an infiltration basin at Sunndalsøra groundwater works was to reduce iron contents by means of infiltrating oxygen-rich river water. Water in the infiltration basin is filtered through marble sand by means of increasing the pH. However, concentration of CO_2 in the river water makes the dissolution of marble less efficient at the moment (Asplan Viak 2008 a, 2009). The general reaction responsible for a pH-adjustment: $CaCO_3$ (marble sand) + $CO_2 \rightarrow Ca^{2+} + HCO_3^-$

VYREDOX-METHOD

This method was developed in Finland, and has been reported to be in use in several countries, e.g. Sweden, Norway and China. Some 150 Vyr-facilities exists, whereof 70-80 facilities are found in Sweden and Finland. This method is in use at present at Elverum water works in Norway (see Box 7). A total of three different methods have been developed according to NGU (2001 b). Development of this method provides the basis for another in situ method, the NITREDOX-method, which also includes the removal of nitrate and nitrite (Braester and Martinell 1988).

Based on complex chemical and biological processes in order to oxidize as much iron as possible while the groundwater is still in the aquifer, this method involves a comprehensive oxidation in the strata surrounding the production well(s). Aquifers consisting of sand and gravel are preferred in order to provide sufficient permeability. Oxygen-rich water is pumped into the ground through satellite wells surrounding the supply well. This form of iron removal is often known as a *contact oxidation process*, meaning that iron hydroxide precipitate forms a film around the grains (sand, pebble) in the aquifer. The iron coating initiates a catalytic reaction between oxygen and iron on sediment surfaces (Maogong 1988). This means that Eh and pH have to be kept at a high level.

Upon extraction, the water should be free of iron and problems related to iron precipitation ought to be insignificant (Hallberg and Martinell 1976). In places where iron concentrations are very high, the VYREDOX-method has been reported utilized as a pre-treatment method prior to a conventional treatment step (Maogong 1988). Oxygenated water in turn establishes an environment where iron oxidizing bacteria thrive, providing another mechanism for iron removal (Armstrong 1978). For a more detailed description on this method, see Hallberg and Martinell (1976).

ARTIFICIAL RECHARGE BASINS

Infiltration basins are artificial constructions aiding the removal of iron. Groundwaters with high values of manganese and humic substances have also been treated successfully with this method. Iron is often also found complex-bound to humic substances (NGU 1991 a).

Basins act as aerators that at the same time provide a filter mass, by means of the retaining processes that takes place in the sediments constituting the basin floor. Iron dissolved in groundwater precipitate as hydroxides upon mixing with the basin water. The precipitate is obstructed from infiltrating and settles on/in the filter mass. It has also been reported that hydroxides form directly beneath the filter mass, but this is still considered successful with respect to iron removal. The method is also beneficial in that it can be combined with a pH-adjustment for acidic water. This will require additions of CO_2 prior to infiltration through a more calcerous filter sand, e.g. marble sand. As a consequence, an alkalinity increase will be achieved (NGU 1991 b). Infiltration basins are in most cases established to augment groundwater basin capacity (Houwer 2002, Helmisaari et.al. 2005). However, re-infiltration of groundwater through basins is much similar to ordinary basin infiltration. Clogging is though a common aspect of all infiltration basins, but this can easily be removed mechanically. The frequency of filter mass regeneration depends on different factors but e.g. good operational routines can limit the need of such (NGU 1991 b). Schuh (1990) and Helmisaari et.al. (2005) present different factors contributing to biofouling and possibilities for efficient regeneration. Figure 12 A and B illustrates some of the aspects of how artificial recharge may progress:

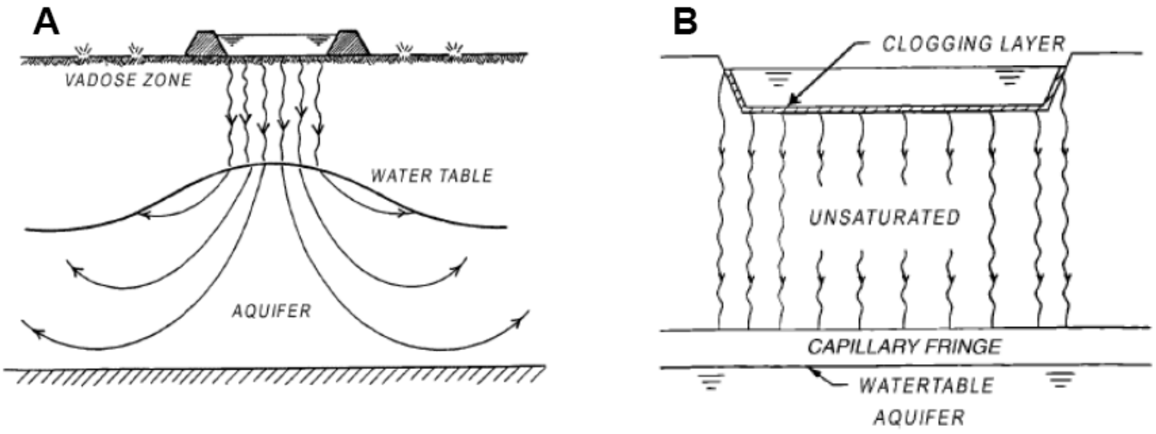


Figure 12: Infiltration basins. A. Infiltration through infiltration basins might cause mounding at the groundwater table interface. B. Clogging is a common aspect of all infiltration basins, but regeneration of the filter mass can easily be done mechanically (Modified from Houwer 2002).

3. STUDY DESIGN AND METHODS

Different methods and equipment have been applied in the field and laboratory to approach the aims of the assignment. An outline of field investigations will be given with emphasize on three stages presented in an order that represent the sequence in which investigations were conducted from well drilling to the termination of test pumping of the new well. Descriptions of laboratory analyzing techniques include four methods that were applied to characterize collected sediments.

3.1 FIELD WORK

3.1.1 ACCOMPLISHMENT

Fields investigations can be recognized as three stages, all conducted in cooperation with AV as part of the work for improving the groundwater works:

- Stage 1: Well drilling
- Stage 2: Step-drawdown test (aquifer test)
- Stage 3: Test pumping

All stages are presented in the following. The *Appendix* provides more detailed information where stated in the text.

STAGE 1: WELL DRILLING

Well drilling was conducted by *Båsum Boring Trøndelag* in the period 26-Jan – 30-Jan-09 with a mobile drilling rig (Figure 13). Decisions on well dimension and location were made based on results from investigations accomplished prior to this stage, as part of the preliminary investigations conducted by AV (Asplan Viak 2008 b). Sediment samples were collected during drilling.

The drilling method applied for the establishment of well D is known as *ODEX-drilling* (Overburden Drilling and Exploration). The method is based on drilling of a borehole with the aid of a downhole percussion hammer that drills and reams at the bottom of the well casing. The casing is essential in that it prevents a cave in of unconsolidated sediments. The hole is drilled slightly larger than the outside diameter of the well casing with a pilot bit, in conjunction with an eccentric reamer. The hammer also impacts on the casing with the aid of a shoe that is attached to the bottom joint of the casing. Consequently, the casing advances downwards as the hole is drilled deeper (USGS 2010). The casing was made (extended) through the welding of the connections and was installed to the full depth of the well. The well screen was subsequently installed by lowering it inside the casing. The casing was then pulled back to expose the well screen to the aquifer sediments. This method is known as the *pull-back method* (Driscoll 1986). Grouting of the space between the well casing and the drilled hole was made with bentonite (clay slurry). This is an action conducted in order to prevent leakage of contamination from the surface (Driscoll 1986). More information on this method and the sediment sampling procedure can be viewed in *Appendix I*.

Figure 13: Mobile drilling rig operated by Båsum Boring Trøndelag during establishment of an Ø273 mm well casing with a deviation from the horizontal line (20,5°) and characteristics of the unsaturated zone.

STAGE 2: STEP-DRAWDOWN TEST

Step-drawdown test I

The 16-Feb-09 was used for preparations in relation to the step-drawdown test. Groundwater level was measured in all piezometers and well D. It was not possible to measure a hydrostatic water level, because of continuous withdrawal from production wells A, B and C, as well as influence from testing of the pump to be used in relation to the testing of well D. Electrical conductivity (EC) and temperature were measured for all production wells.

Table 3: Stage 2. Step-drawdown test I (16-Feb-09).

Date: 16-Feb-09	Piezometers	Driva	Inf. basin	Well A	Well B	Well C	Well D
Groundwater level	√						√
Field analyses:							
Electrical conductivity				√	√	√	
Temperature				√	√	√	

The step-drawdown test was conducted on the 17-Feb-09. This test involved three steps of approximately one hour duration each. The change in drawdown with time was noted for the new well, as well as for the two piezometer wells in vicinity of well D (piezometers 22 and 23). The test was cancelled after some time because of problems with the pump capacity. Samples of water were still collected for physical-chemical analyses, field analyses of iron, and measurements of electrical conductivity and temperature.

Table 4: Stage 2. Step-drawdown test I (17-Feb-09).

Date: 17-Feb-09	Piezometers	Driva	Inf. basin	Well A	Well B	Well C	Well D
Groundwater level	√						√
Step-drawdown test	√						√
Water samples:							
Physical/chemical parameters			√	√	√	√	√
Field analyses:							
Iron					√	√	√
Electrical conductivity		√	√				√
Temperature		√	√				√

Step-drawdown test II

A second aquifer test was initiated again on the 30-Mar-09, consisting of three steps with capacities of 9,3 l/s, 12,3-14 l/s and 18 l/s, respectively. Pumping continued successfully until the next morning with a pumping capacity of 18 l/s. Drawdown in meters was registered for well D and all other piezometers before the initiation of the test, at the termination of step three and the next day.

Drawdown versus time was recorded in well D during the test period using a measure tape. Drawdown was also registered in two adjacent piezometer wells (22 and 23). Measurements of groundwater level in well D and piezometer 22 constitutes the basis for calculations of maximum yield of the new well presented in sections 4.2.2-4.2.5.

Electrical conductance and temperature were measured for each step, and water samples were collected from all wells on the 30-Mar-09. Water sampling was also done in line with a collection of samples for analysis of microbiological parameters. This was conducted as part of a ph.d.-project at NTNU, and involved the concentration of microbes in large volumes of water in field.

Table 5: Stage 2. Step-drawdown test II (30-Mar-09).

Date: 30-Mar-09	Piezometers	Driva	Infiltration basin	Well A	Well B	Well C	Well D
Groundwater level	√						√
Step-drawdown test	√						√
Water samples:							
Physical/chemical parameters		(√)	(√)	√	√	√	√
Microbiology		√	√	√			
Field analyses:							
Electrical conductivity		√	√	√	√	√	√
Temperature		√	√	√	√	√	√
pH		√	√	√	√	√	

STAGE 3: TEST PUMPING

Test pumping was initiated on the 31 Mar-09 as a continuation of step-drawdown test II. Pumping took place with a capacity of 18 l/s. As a result of reduced river discharge at that time, groundwater level was lowered to such a degree that the capacity of production wells dropped to a critical level. Test pumping was consequently cancelled on a temporary basis from the beginning of April.

A restart of the test pumping was accomplished in May/June, still with a capacity of 18 l/s. Different operational situations were tested in line with the aims of the assignment.

This involved the closing down of well B in the beginning of June, and documentation of the effects following this action, relying on groundwater withdrawn from production wells A and C only for the rest of the test pumping period (Figure 14).

Table 6: Stage 3. Test pumping (8-Sep-09).

Date: 8-Sep-09	Piezometers	Driva	Infiltration basin	Well A	Well C	Well D
Groundwater level	√					
Water samples:						
Physical/chemical param.			√	√	√	√
Field analyses:						
Iron				√	√	√
Ammonium				√	√	√
Manganese				√	√	√
Carbon dioxide						√
Electrical conductivity		√	√	√	√	√
Temperature		√	√	√	√	√
pH		√	√	√	√	√

OBSERVATIONS OF IMPORTANCE DURING THE TEST PUMPING

Additional observations of importance have been made with respect to natural variations of the groundwater level, as a response to changes in river discharge and the effect of the varying infiltration capacities of the infiltration basin (e.g. effects from artificial recharge through the infiltration basin in section 4.3.8).

Figure 14 provides a chronological overview of the different steps of field work from the establishment of well D to the termination of the test pumping together with sampling dates and different situations that occurred during the test pumping.

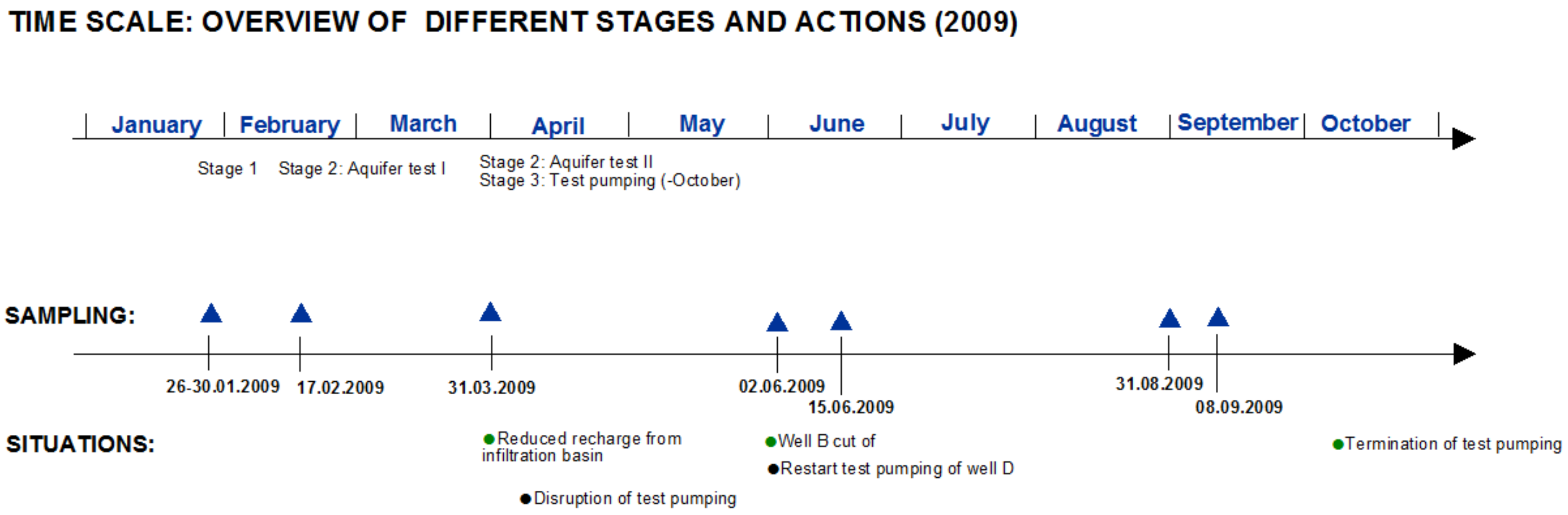


Figure 14: Time scale with an overview of the different stages and actions performed at the groundwater works.

SEDIMENT SAMPLING CONDUCTED DURING STAGE 1

Sediment sampling

Sediment samples of c. 500 grams each were taken from six different depths as drilling progressed. Sampling intervals represent drilling depths of 16-18 m, 19-21 m, 22-24 m, 25-27 m, 28-30 and 31-33 m. The well was drilled with a tilt of 20,5° from the horizontal and total well length was reduced to allow for installation of pump for operational purposes during test pumping (*Stage 3*). Samples were brought to the lab for analysis of grain size distribution, mineral- and elemental composition, as well as organic contents (section 4.1).

METHODS AND EQUIPMENT APPLIED DURING STAGES 2 AND 3

Water sampling

As part of the *Stages 2 and 3*, a total of 21 water samples have been collected for analyses for physical-chemical parameters at NGU. A number of 15 samples have been collected as part of the field work, while the rest have been administrated by the municipality. All water samples have been analyzed for turbidity, color, electrical conductivity, temperature, pH, alkalinity, cations and anions. Samples were collected and stored unfiltered in 500 ml polyethylene bottles, while samples for cation and anion analyses were filtered through a 45µm-filter and transferred into 100 ml polyethylene bottles using 50 ml polyvinyl chloride syringes. Well samples were collected from a water tap in the pump house and represents mixed samples. This means that groundwater samples were extracted through well screens of some ten meters length and not from a discrete depth.

Water sampling for bacteriological analyses were administrated by Sunndal municipality and analyzed at *Trollheimslab,* Sunndalsøra. The municipality was also responsible for the sampling of water for physical-chemical and chemical analyses at NGU in accord with the sampling plan suggested by AV. Because of an unexpected reduction in workforce during summer, the frequency of measurements was reduced.

Field analyses

EC and temperature were measured using WTW *Cond 330i/SET* with *Tetra Con 325* electrode. A WTW *pH/Cond 340i/SET* with a *SenTix 41* electrode was used to measure pH.

Field analyses of selected parameters were conducted with a *Photometer (WTW)*. Of the dissolved ions that were analyzed, were iron, manganese and ammonium. Water samples for this purpose were initially collected in 500 ml plastic bottles. Pipettes were used to transfer the necessary volume of water to glass containers. Dissolved carbon dioxide was measured on one occasion with a method based on titration.

Groundwater level and capacity measurements

Groundwater (piezometric) level was measured in piezometer wells using a water level dipper. This was made in order to document the development of drawdown in relation to the step-drawdown test, and to record the drawdown during the test pumping period to register any modifications of the groundwater flow pattern. Groundwater level in well D was measured using the simple approach of blowing through a plastic pipe in order to detect depth to the groundwater surface. Measurements of well capacity (withdrawal) from well D during *Stages 2 and 3* were made with a water flow meter.

Sunndal municipality was responsible for additional measurements of the groundwater level during the test pumping. Because of an unexpected reduction in workforce during summer, the number of measurements was reduced considerably with respect to the measurement plan initially proposed by AV.

Hemoflow-method

Concentration of large volumes of water in field for analysis of microbes was conducted in relation to a ph.d-project at NTNU (Hanne Kvitsand). This method will be referred to as the Hemoflow-method in the following.

Three water samples for microbiological analyses were collected during field investigations on the 30-Mar-09 (*Stage 2*). The microbiological condition of the river, infiltration basin and well A was investigated with a method based on the concentration of microbes in large volumes of water. The method has been developed in relation to the TECHNEAU-project, an integrated project funded by the European Commission under the Sixth Framework Programme, Sustainable Development, Global change and Ecosystems Thematic Priority Area. The approach suggested by TECHNEAU (2007), able the detection of pathogenic microorganisms that exists at low concentrations in natural waters, as well as allowing quantification.

A filter used for renal dialysis makes the basis for the technique, as it can also be applied for the collection and concentration of microbes. The filter is known as the *Hemoflow-filter,* a cross flow ultrafilter, concentrating viruses/phages, spores, bacteria and parasitic protozoa. Simmons et.al. (2001) give a more detailed description of the filter. The method conducted in the field and the material used, are in line with the guidelines found in the report by TECNEAU (2007). *Appendix V* gives additional information on the setup of this method.

The concentrates were split into three equal amounts and transferred into plastic containers, whereof some 200 ml were sent to *Analysesenteret,* Trondheim, for analysis of indicator bacteria for detection of fecal contamination. Respective raw water samples from all sampling locations were analyzed for turbidity, electrical conductivity, color, organic content (TOC and DOC), cations, anions, *Campylobacter*, e.-coli, intestinal enterococci, coliforms and *Clostridium perfringens*.

Figure 15: Setup of the Hemo-flow installation (Veenendaal and Brouwer-Hanzens 2007).

3.2 LABORATORY ANALYSES

3.2.1 ACCOMPLISHMENT

Sediment samples have been analyzed at the Engineering Geology Laboratory at IGB. A total of three samples were studied for grains size distribution, elemental- and mineral composition and organic content. Distribution of grain sizes was studied through dry weight sieving analyses. X-ray fluorescence (XRF) and X-ray diffraction (XRD) were applied to analyze for elemental- and mineral content, respectively, while organic content was found using a method based on loss on ignition (LOI). Two of the samples have been collected prior to the initiation of the thesis during preliminary investigations by AV. The third sample was collected in relation to the master thesis during drilling of the new well (*Stage 1*).

GRAIN SIZE DISTRIBUTION

To obtain information on the grain size distribution of unconsolidated, siliciclastic sediments, grain size analyses can be performed. The grain size refers in theory to the intermediate dimension of a grain, and all collected sediment samples were analyzed using this technique.

Initially, samples were dried in an oven over night at c. 105 °C. The samples were subsequently split into reasonable and representative sieving portions, and weighted for total dry weight. Sieving was conducted with nested, wire-mesh screens ranging in dimensions from 19 mm to < 0,063 mm. Upon adding a sample, the sieves were placed in a stirring machine to aid the sieving process and to secure that all mud fractions (silt and clay) would fall through the higher situated coarser screens. The content of each sieve was weighted individually, representing the grains size fraction that was obstructed from falling through the respective sieves. Weight was noted “greater than” (>). Grains smaller than 0,063 mm, were measured as one fraction. The relationship between weight on each sieve and total weight was established to give cumulative weight in percent. The classification system used for naming different grain sizes in this study can be viewed in *Appendix VI*.

Figure 16: Grain size analyses performed with dry sieving in the Engineering geology laboratory at IGB.

Grain size distribution curves represent a certain sampling depth interval. A grain size cumulative curve was established for each sample by plotting grain size in µm with logarithmic-scale on the abscissa against a cumulative weight percent frequencies ordinate scale. This presentation provides information on sediment grading (sorting) from the shape of the curves, the passage of grain sizes in percent, as well as enabling an establishment of other statistical parameters, e.g. calculation of sediment grading (Boggs 2006 , Fetter 2001).

X-RAY DIFFRACTION (XRD)

X-ray diffraction analyses were conducted with *Philips PW 1830* to identify the mineralogical composition of the sediments. Bulk samples were ground to powder and compressed into pills. The interpretation software used was the *JCPDS (Joint Committee on Powder Diffraction Standards)* from International Centre for Diffraction Data. New equipment of the type *Bruker S8 Advance* was applied for the second round of analyses and administrated by IGB. Samples were prepared using the same procedure as earlier. For detailed information on the theory and analyzing techniques of XRD, references are given to Nesse (2000) and Klug and Alexandr (1974).

X-RAY FLUORESCENCE (XRF)

X-ray fluorescence analyses identify the elemental composition of the sediment. The analyses were made using *Philips PW* 1480. Bulk samples were pulverized, and an amount of approximately 0,5 grams were prepared by fusing the powder with 5 grams of flux consisting of *66 % Lithium Tetraborat/34 % Lithium Metaborat*-powder. The substance was subsequently casted into 32 mm glass disks. The second round of analyses was conducted by the institute with new equipment of the type *Bruker S8 Tiger (4kW)*. In this case, c. 0,8 grams of ignited sample was cast into glass disks with 8 grams of flux powder. For detailed information on the analyzing technique and theoretical principles of XRF, reference is given to Riddle (1993).

LOSS ON IGNITION (LOI)

The method of loss on ignition enables the quantification of organic content in a sediment sample. LOI was performed by igniting dry-weighted bulk samples of 2,5-3 grams in a muffle furnace at ~ 875 °C. The percent of loss on ignition was calculated from the weight loss as a percentage of the oven-dried weight.

4. RESULTS

Results from field investigations (*Stages 1-3*) and laboratory analyses will be presented in a more or less chronological order in this section. The results presented are assessed and discussed later in section 5. The results are divided into seven main topics:

- Sediment and aquifer properties
- Well yield
- Water chemistry
- Microbiology
- Groundwater flow pattern
- Residence time
- Sources of groundwater recharge

Appendix II and *VIII (CD)* contain methods of calculation and raw data in excel-spreadsheets, respectively.

4.1 SEDIMENT AND AQUIFER PROPERTIES

4.1.1 GRAIN SIZE DISTRIBUTION

Grain size distribution curves are shown for samples collected from investigation well 1 (Ub-1/Pb-21), investigation well 2 (Ub-2/Pb-22) and well D. Each curve represents distinct depth intervals:

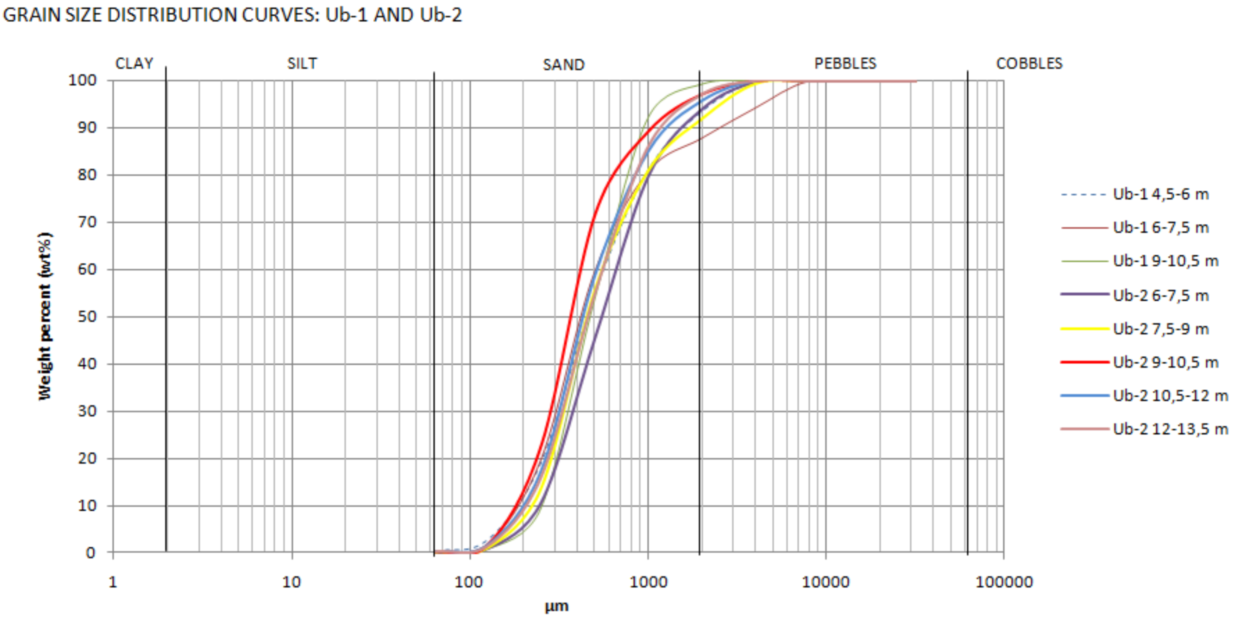


Figure 17: Grain size distribution curves of flushed samples from Ub-1 and Ub-2.

The relatively uniform curve shapes and the distribution of grain sizes in Ub-1 and Ub-2 reflects a well sorted deposit dominated by sand with some pebble. The least sorted part of the aquifer is represented by the lowermost level in Ub-2. Grain size distributions of sediments collected from well D are less uniform in all levels compared to the investigation wells. The sand and pebble fractions are still dominant, but there is a tendency towards coarser sediments in the fractions of cobbles and pebbles. Curve shapes are clearly less uniform resulting in reduced sorting (Figure 18):

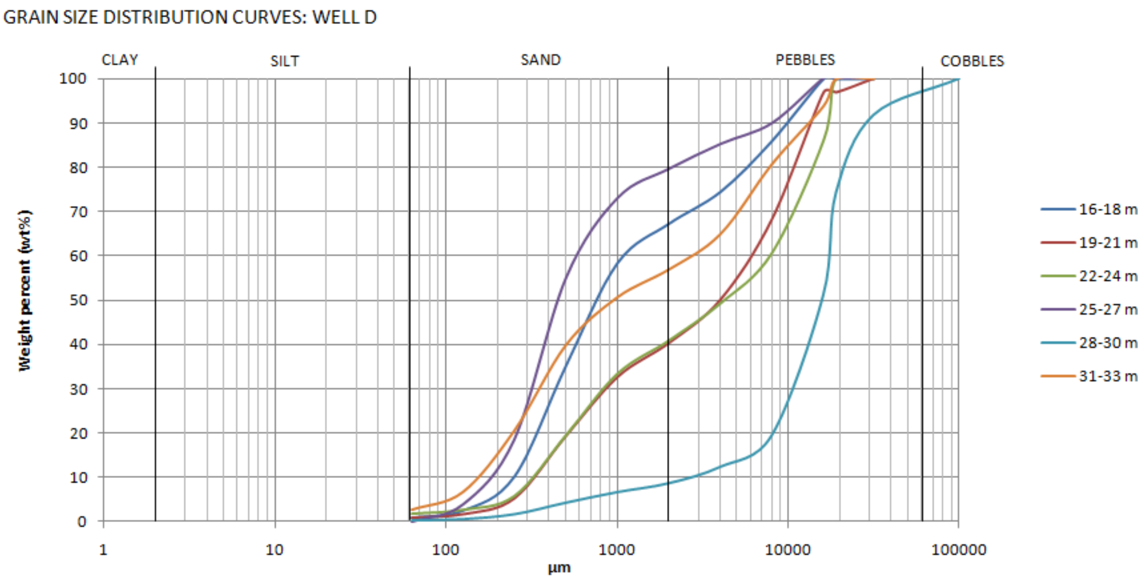


Figure 18: Grain size distribution curves of samples collected from the drilling of well D.

Grain size analyses errors are shown as weight percent in Table 7. Errors are generally low, but one exception can be seen for Ub-2 that represents the lowermost sampling level. Negative values indicate loss of sample material during sieving, while positive values are connected to inaccurate reading from the scale.

Table 7: Errors of grain size distribution analyses as weight percent loss (wt. % loss).

Ub-1	***4,5-6 m***	***6-7,5 m***	***9-10,5 m***			
	-0,04	-0,01	-0,18			
Ub-2	***6-7,5 m***	***7,5-9 m***	***9-10,5 m***	***10,5-12 m***	***12-13,5 m***	
	-0,22	-0,25	-0,07	0,08	8,33	
Well D	***16-18 m***	***19-21 m***	***22-24 m***	***25-27 m***	***28-30m***	***31-33 m***
	-0,6	-0,3	-0,2	-0,2	-0,1	-0,4

4.1.2 HYDRAULIC PARAMETERS COMPUTED FROM GRAIN SIZE ANALYSES

Hydraulic parameters are defined on the basis of grain size distribution curves and from pumping test data in this study. Results from the aquifer test (step-drawdown test) are presented in section 4.2 together with estimates for maximum well yield of the new well (Q_{max}).

INVESTIGATION WELLS

Table 8 presents values of d_{10} and d_{60} from the grain size distribution curves of Ub-1 and Ub-2. All values of grading are in the range of the Hazen method (Cu < 5). Hydraulic conductivity (K) and have been calculated accordingly using the Hazen method only. The values of grading indicate a well sorted deposit. K-values vary between 3,352-6,682·10^{-4} m/s taking into account all levels. All values fit the range of sand and gravel. T-values are dependent on the layer thicknesses and have in consequence been calculated for each depth interval, starting at the groundwater level (3-3,5 m).

Table 8: Hydraulic properties derived from the grain size distribution curves of samples from Ub-1 and Ub-2.

Well	Depth (m)	d_{10} (mm)	d_{60} (mm)	Cu	K (m/s) *Hazen method (C_u < 5)*	K (m/day) *Hazen method(C_u < 5)*	T (m^2/s)
Ub-1 *UTM 32V 478325, 6948540*	3-6	0,170	0,600	3,53	3,352·10^{-4}	28,9	1,006·10^{-3}
	6-9	0,170	0,500	2,94	3,352·10^{-4}	28,9	1,006·10^{-3}
	9-13	0,260	0,580	2,23	7,842·10^{-4}	67,8	3,137·10^{-3}
	SUM						5,15·10^{-3}
Ub-2 *UTM 32V 478300, 6948590*	3,5-7,5	0,240	0,690	2,88	6,682·10^{-4}	57,7	2,673·10^{-3}
	7,5-9	0,200	0,560	2,80	4,640·10^{-4}	40,1	0,696·10^{-3}
	9-10,5	0,170	0,440	2,59	3,352·10^{-4}	28,9	0,503·10^{-3}
	10,5-12	0,180	0,510	2,83	3,758·10^{-4}	32,5	0,564·10^{-3}
	12-13,5	0,190	0,560	2,95	4,188·10^{-4}	36,2	0,628·10^{-3}
	SUM						5,06·10^{-3}

WELL D

Grain size distribution in this case shows there is a greater range between d_{10} and d_{60} with respect to data from the investigation wells (Table 9). The values of grading (Cu > 5) are in such a range that the Gustafsson method has been applied for all levels in order to calculate hydraulic conductivity (K). The values indicate a slightly less sorted deposit relative to samples from Ub-1 and Ub-2. K-values are in the range $4{,}480\text{-}8{,}13 \cdot 10^{-4}$ m/s taking into account all levels. All values fit the range of sand and gravel.

Table 9: Hydraulic properties derived from grain size curves of samples from well D and the use of the Gustafsson method.

Drilling length (m)	d_{10} (mm)	d_{60} (mm)	Cu	E (Cu)	K (m/s) *Gustafsson method (Cu>5)*	K (m/day) *Gustafsson method (Cu>5)*
16-18	0,250	1,150	4,60	0,0141	$8{,}813 \cdot 10^{-4}$	76,1
19-21	0,325	5,800	17,85	0,0070	$7{,}394 \cdot 10^{-4}$	63,9
22-24	0,315	7,800	24,76	*Not able to define $C_u > 20$ graphically with the Gustafsson method.*		
25-27	0,170	0,600	3,53	0,0155	$4{,}480 \cdot 10^{-4}$	38,7
28-30	2,600	17,000	6,54	0,0120	$8{,}112 \cdot 10^{-2}$	7008,8
31-33	0,155	2,650	17,10	0,0071	$1{,}706 \cdot 10^{-4}$	14,7

4.1.3 ELEMENTAL COMPOSITION (XRF)

Elemental composition of sediments from different depth intervals is shown in Tables 10 and 11 for the investigation wells and well D, respectively.

Table 10: Elemental composition of Ub-1 and Ub-2 reported as weight percent oxides (wt. % oxides).

Well/depth (m)	SiO_2	TiO_2	Al_2O_3	Fe_2O_3	MnO	MgO	CaO	Na_2O	K_2O	P_2O_5	Total	LOI
Ub-1/4,5-6	75,52	0,50	12,16	3,53	0,07	1,40	2,65	2,93	2,37	0,07	98,19	0,72
Ub-1/6-7,5	71,94	0,59	12,21	3,59	0,07	1,53	2,79	2,83	2,19	0,07	97,81	0,69
Ub-1/9-10,5	72,33	0,53	12,24	3,57	0,07	1,41	2,65	2,93	2,38	0,07	98,18	0,74
Ub-2/6-7,5	73,70	0,45	12,00	3,05	0,06	1,28	2,39	2,88	2,47	0,06	98,34	0,67
Ub-2/7,5-9	72,42	0,52	12,07	3,53	0,07	1,48	2,65	2,82	2,40	0,06	98,02	0,74
Ub-2/9-10,5	72,87	0,49	11,94	3,49	0,06	1,60	2,50	2,67	2,27	0,07	97,97	0,88
Ub-2/10,5-12	72,63	0,49	11,82	3,45	0,06	1,47	2,52	2,77	2,37	0,07	97,65	0,79
Ub-2/12-13,5	72,80	0,55	11,82	3,59	0,07	1,34	2,66	2,81	2,31	0,07	98,02	0,63

Table 11: Elemental composition of Well D reported as weight percent oxides (wt. % oxides).

Drilling depth (m)	SiO_2	TiO_2	Al_2O_3	Fe_2O_3	MnO	MgO	CaO	Na_2O	K_2O	P_2O_5	Total	LOI
16-18	72,12	0,47	12,33	3,01	0,06	1,47	2,60	3,44	2,58	0,08	98,16	0,73
19-21	70,11	0,51	12,80	3,29	0,06	1,60	2,66	3,51	2,75	0,10	97,39	0,70
22-24	72,08	0,44	12,52	3,07	0,06	1,86	2,71	3,33	2,55	0,09	98,71	0,76
25-27	74,04	0,42	11,85	2,73	0,04	1,44	2,43	3,22	2,48	0,06	98,71	0,70
28-30	72,36	0,46	12,58	3,06	0,05	1,40	2,56	3,23	3,14	0,10	98,94	0,64
31-33	71,58	0,54	12,51	3,53	0,06	1,64	2,74	3,23	2,63	0,09	98,55	0,95

The dominating elements given as weight percent oxides are in descending order: silicon, aluminum, iron, sodium, calcium and potassium. Silicon constitutes more than 70 wt.%, aluminum occurs in the range of 11,8-12,8 wt.% and iron follows in the range of 3,01-3,59 wt.%. Magnesium is found in the range of 1,4-1,86 wt.% , followed by titanium, manganese and phosphorous representing the minor elements in this case. With respect to weight percent oxides, iron is the third most prominent element following silicon and aluminum.

4.1.4 MINERALOGICAL COMPOSITION (XRD)

Sediments from the investigation wells are mainly composed of minerals from the silicate group. Silicates are identified as plagioclase, K-feldspar, quartz, amphibole and muscovite. Calcite was detected in small amounts, and is the only calcerous mineral found in the sediments.

Figure 19 displays the lithological logs of the investigation wells with sector diagrams to illustrate mineral composition and distribution with depth. Lithological logs have been established on the basis of results of pre-investigation by AV (Asplan Viak 2008 b).

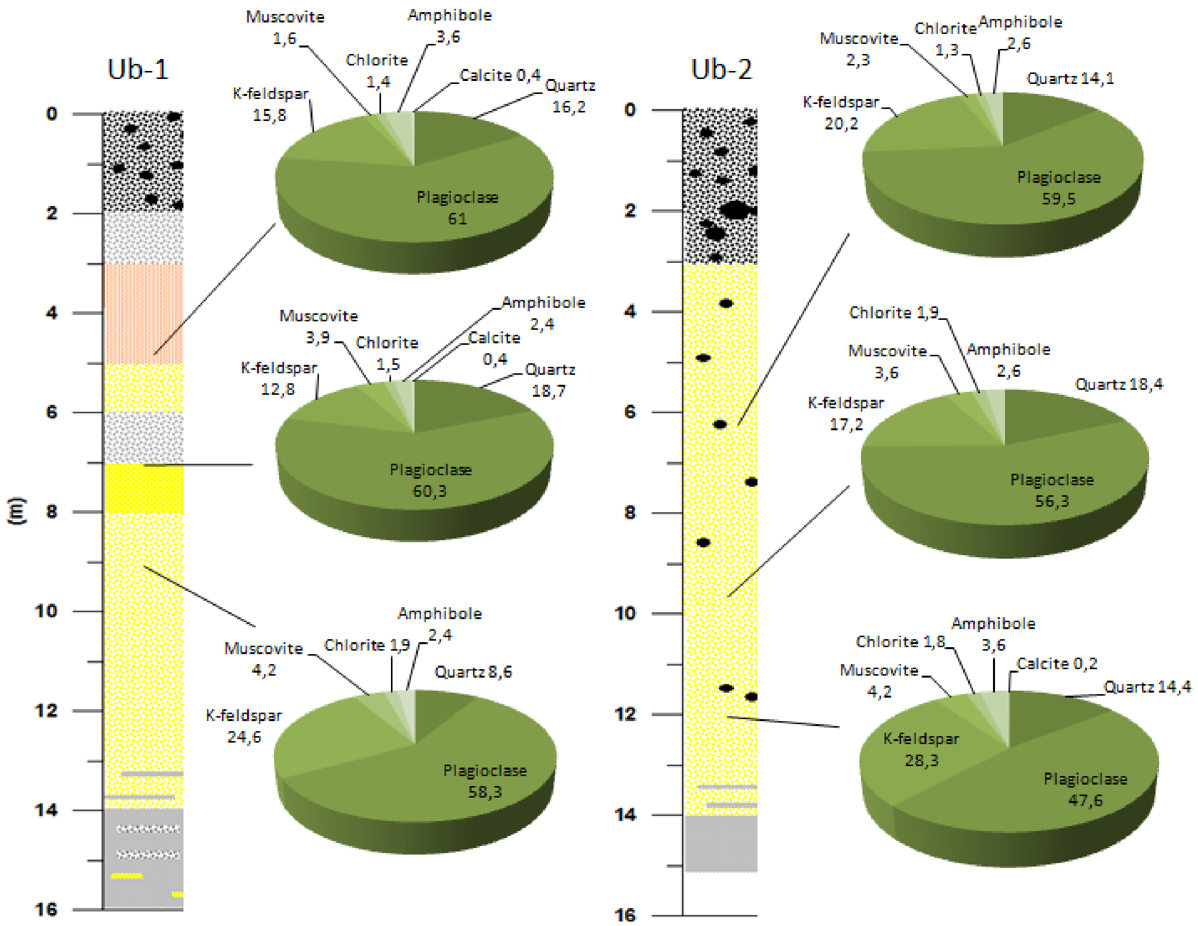


Figure 19: Mineralogical composition of samples from Ub-1 and Ub-2 with lithological logs showing the mineral distribution with depth.

Analyses of sediments sampled from well D have provided a more detailed picture of the mineral types in this case (Table 12); Amphibole is represented by the sodium-rich member riebeckite. K-feldspar is identified as microcline, while plagioclase is reported as albite. Clinochlore is identical to ferroan chlorite, which belongs to the phyllosilicates. A K,Al-hydroxide was found only in the depth interval of 16-18 meters, but only occur in minor amounts. *Appendix VIII (CD)* provides more detailed information on the mineral analyses as a whole.

Table 12: Mineralogical composition of sediment samples from the drilling of Well D in weight percent (wt. %).

Mineral	16-18 m	19-21 m	22-24 m	25-27 m	28-30 m	31-33 m
Quartz	33,3	30,9	30,4	31,7	30,6	35,8
Plagioclase (Albite, calcian)	35,1	31,3	30,5	29,4	28,3	28,8
K-feldspar (Microcline)	15,5	16,2	18,0	19,4	14,5	13,5
Muscovite		16,4	16,0	15,4	24,3	17,5
Chlorite (Clinochlore)	1,1	1,0	1,0	1,0	0,7	0,9
Amphibole (Riebeckite)	3,9	4,3	4,1	3,2	1,7	3,4
K,Al-hydroxide	11,1					

4.1.5 LOSS ON IGNITION

Analyses of loss on ignition (LOI) returned values of < 1,0 % for all samples. Values are in the range of 0,69-0,74 % and 0,63-0,88 % for Ub-1 and Ub-2, respectively. Sediments from well D display a slightly greater range with 0,64-0,95 %. In addition to Figure 20, values of LOI are also displayed together with the elemental composition of sediments in Tables 10 and 11 for samples collected from Ub-1, Ub-2 and well D.

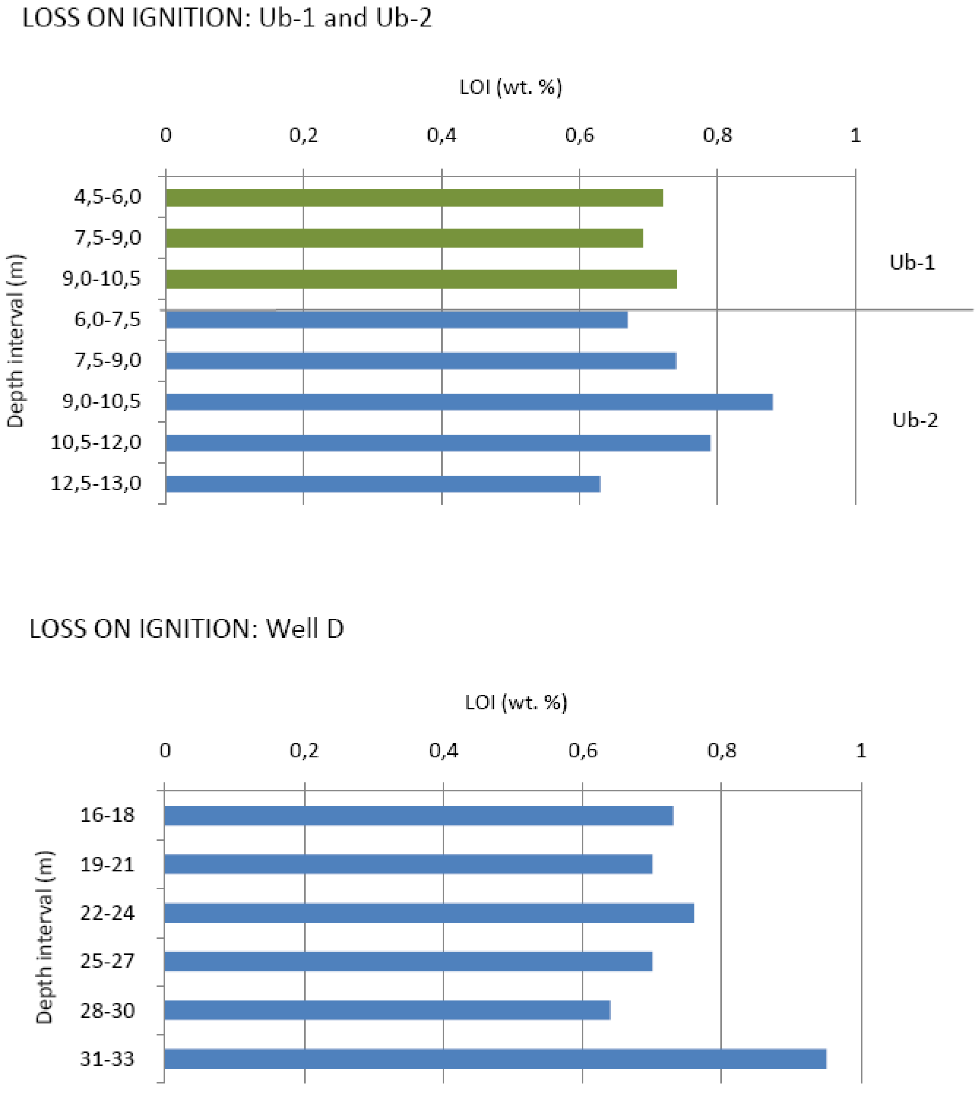


Figure 20: Loss on ignition in weight percent (wt. %) for Ub-1, Ub-2 and well D.

4.2 WELL YIELD

4.2.1 WELL ESTABLISHMENT

The new production well was established in the period 26-Jan to 30-Jan-09 as a tilted Ø168 mm well. The total length of the well casing equals 30 meters with a well screen of twelve meters.

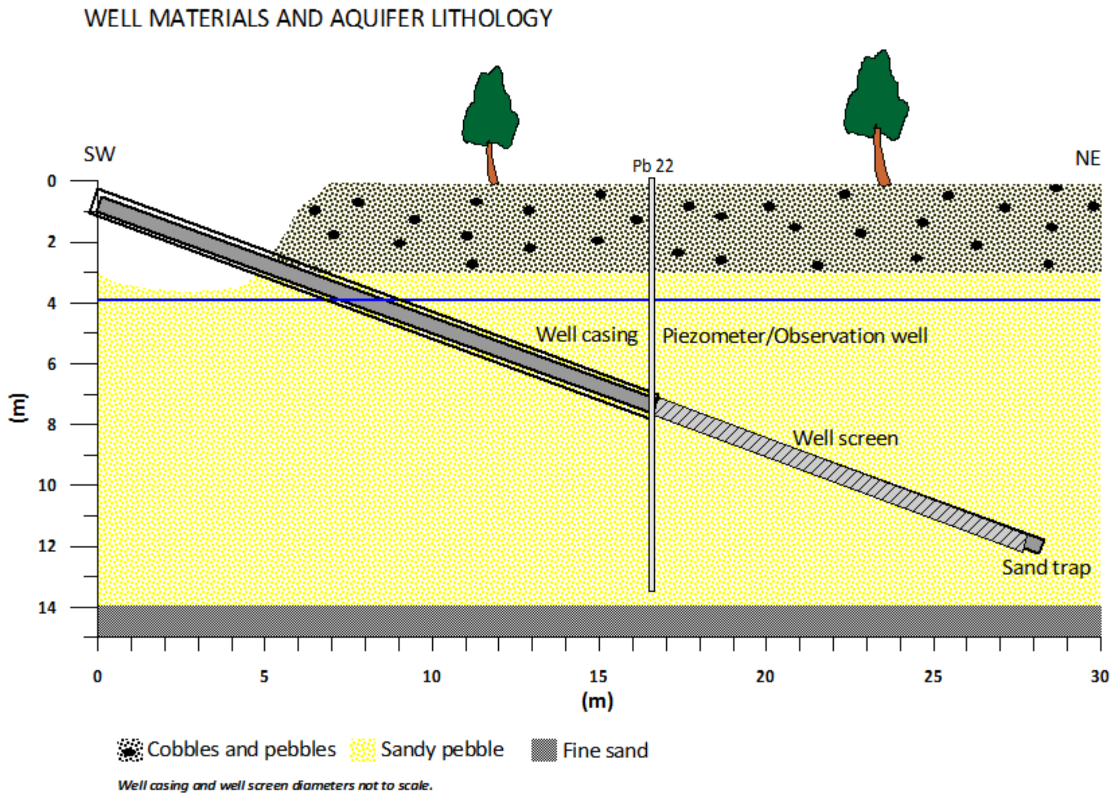


Figure 21: Vertical cross section showing the location and materials of the new production well with aquifer lithology.

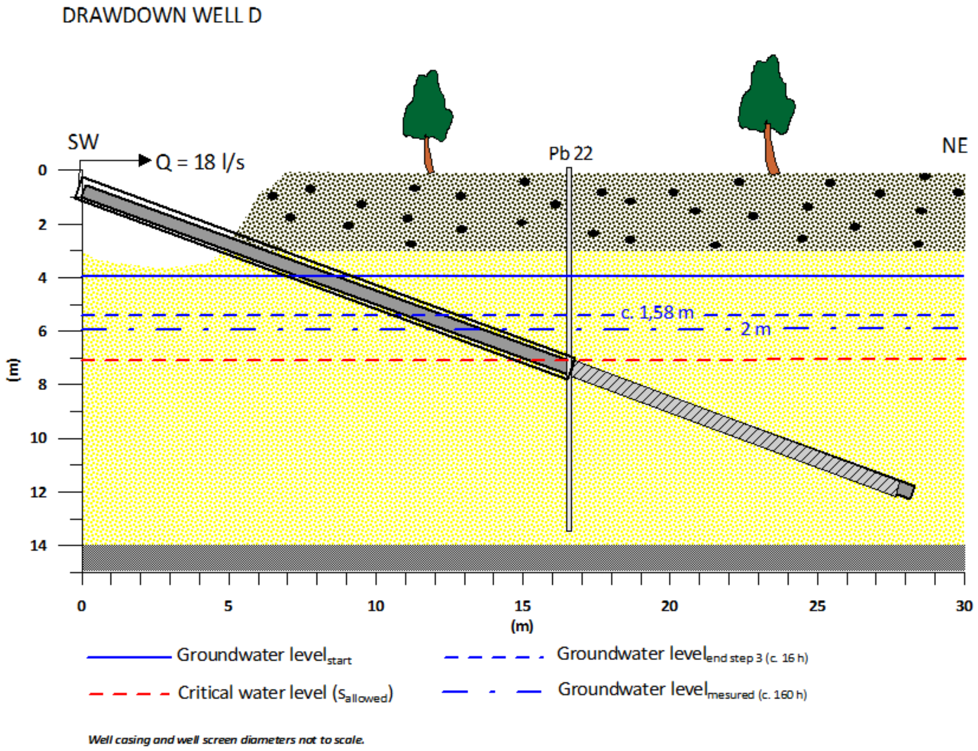
Table 13 provides additional data on well materials and specifications, and constitutes the basis for Figures 21 and 22:

Table 13: Well materials and specifications of well D.

Well D (UTM, zone 32V: E478284, N6948563)	
Diameter well (mm)	168
Diameter well casing (mm)	160-168
Total well length (m)	30
Production well length (m)	17,5
Angle from the horizontal	20,5°
Well screen below terrain (m)	7,1-11,3
Well screen below groundwater level (m)	3,5-7,7
Screen type	Continuous slots
Screen length (m)	12
Screen slots (mm)	1,5
Sand trap (m)	0,5

4.2.2 DEFINING THE BASIS FOR ESTIMATE I AND II OF Q_{MAX}

Figure 22 illustrates the vertical drawdown with time and the maximum allowable drawdown with respect to screen position:



Well casing and well screen diameters not to scale.

Figure 22: Registered drawdown in well D during Stage 2 Step-drawdown test II. (Conversion of drawdown to vertical depth has been made).

Initial groundwater level is shown as a continuous line, indicating a hydrostatic water level. However, this is not completely true, since the groundwater level initially measured was influenced by groundwater extraction from production wells A, B and C. The location of the screen below groundwater level is essential when estimating the maximum well yield. Taking into account the development of the drawdown curve of the well, a steady –state condition has been estimated to occur after c. 160 hours. With this assumption, a total drawdown has been estimated to amount to c. 2 meters.

The following parameters were defined on the basis of the situation presented in Figure 22:

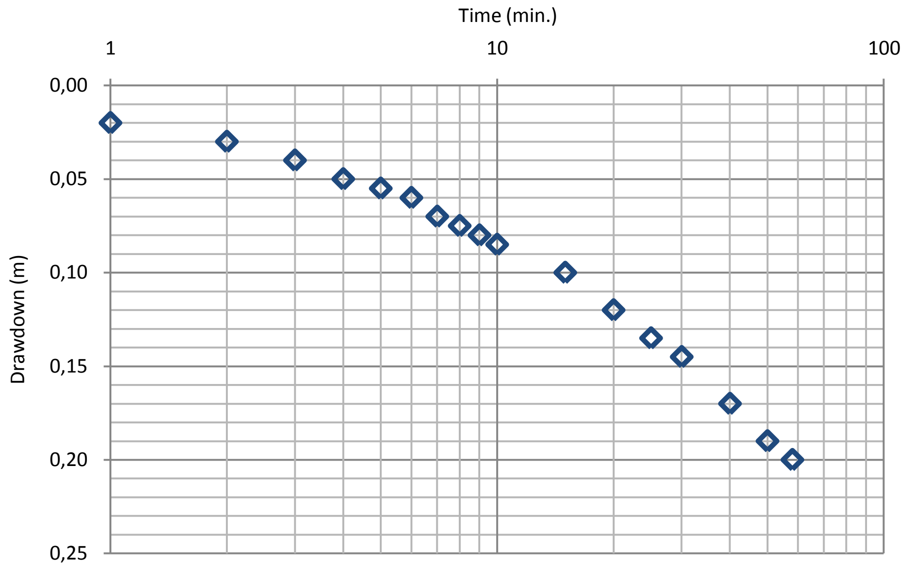
Table 14: Parameters defined from recordings of drawdown in well D.

Capacity and drawdown data	
Q	18 l/s
$s_{allowed}$	3,2 m
$s_{end\ step\ 3\ (c.\ 16\ h)}$	c. 1,58 m
$s_{measured\ (c.\ 160\ h)}$	c. 2 m

4.2.3 ESTIMATE I: JACOB'S METHOD (JACOB TIME-DRAWDOWN STRAIGHT-LINE METHOD)

The Jacob's method has been applied for the interpretation of transmissivity (T) from pumping test data (drawdown with time) collected during *Stage 2 Step-drawdown test II.* T-values constitute the basis for estimations of maximum well capacity, as well as representing an important groundwater basin parameter for aquifer characterization. Two attempts are made to define T-values with this method in order to provide improved data (Figure 23).

STEP-DRAWDOWN TEST: PUMPING OF WELL D -
drawdown with time in Pb 22 during step 1 (Q_1 = 9,3 l/s)



STEP-DRAWDOWN TEST: PUMPING OF WELL D -
corrections for time and drawdown in Pb 22

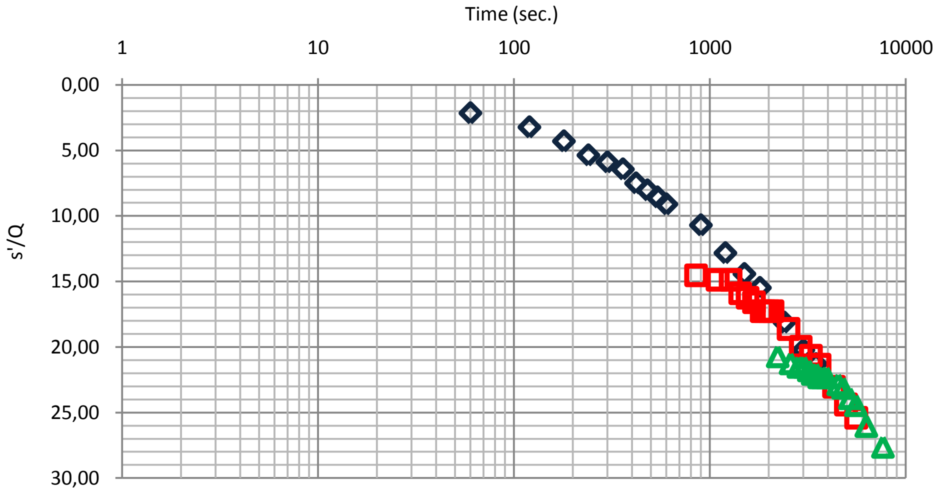


Figure 23: Pumping test data plotted as drawdown with time from Stage 2 Step-drawdown test II. (Lowermost diagram displays data modified with time and drawdown corrections.)

The uppermost diagram in Figure 23 shows a curve representing the development of drawdown with time recorded during the first step of the aquifer test. There has not been performed any drawdown or time corrections in this case. In the lowermost diagram one can see all three steps of the aquifer test, this time with corrections for drawdown and time. Values used for the computation of T-values have been found graphically using adjusted straight lines extrapolated to the abscissa. Recordings were made for 1-log unit:

Table 15: Transmissivity (m^2/s) calculated with the Jacob's method.

Values of transmissivity	
Jacobs's method *Step 1*	$9{,}05 \cdot 10^{-3}$
Jacob's method *Drawdown- and time corrections*	$9{,}86 \cdot 10^{-3}$

Maximum well yield was found using values of allowed drawdown ($s_{allowed}$) in Table 14 and well diameter (r) from Table 13. Influential radius (R_0) was given a value of 100 meters in this case. This returned the following values of Q_{max}:

$$Q_{max} = \frac{3{,}2 \cdot 2\pi \cdot 9{,}05 \cdot 10^{-3}}{ln\frac{100}{168}}$$

Table 16: Calculations of Q_{max} (l/s).

Values of Q_{max}	
Step 1	28,1
Drawdown- and time corrections	30,6

4.2.4 ESTIMATE II: CRITICAL VERSUS OBSERVED DRAWDOWN

Calculations in this case are based on measured drawdown in the well ($s_{measured}$) during *Stage 2 Step-drawdown test II*, the maximum allowed drawdown with respect to the distance between groundwater table and top of screen ($s_{allowed}$) and the amount of water withdrawn during test pumping (Q). Values of drawdown have been converted to vertical drawdown in this case. An illustration of the conditions making the basis for this estimate was presented in Figure 22, while Table 14 provides the relevant data inputs used in the equation below:

$$Q_{max} = 18\; l/s \cdot \frac{3{,}2\ \mathrm{m}}{2\ \mathrm{m}} == \ 28{,}8\ \mathrm{l/s}$$

4.2.5 INCLUDING A SAFETY MARGIN TO ESTIMATE II

Taking into account episodes of extremely low groundwater level, a safety margin can be included. This can been accounted for, supposing that an additional drawdown of 0,5 meters will occur relative to the values of groundwater level obtained from measurements on the 31-Mar-09 (*Appendix IV*). This month, and this specific date, is representative of the groundwater level during dry spells. Taking such conditions into account the resulting capacity would be:

$$Q_{max} = 18\; l/s \cdot \frac{2{,}7\ \mathrm{m}}{2\ \mathrm{m}} = \underline{24\ \mathrm{l/s}}$$

4.3 WATER CHEMISTRY

This section will deal with results of the water analyses from the test pumping. Development of water quality in all wells, as well as Driva and the infiltration basin, will be the focus of the following presentation. *Appendix III* and *V* provide additional information on the respective water analyses and microbiology, respectively.

4.3.1 MAJOR ION CONTENTS

Major ion contents of samples from Driva and the infiltration basin are compared to show the similarity between these two sources. This relationship is also evident from field measurements (EC, temperature and pH) presented in e.g. section 4.3.4.

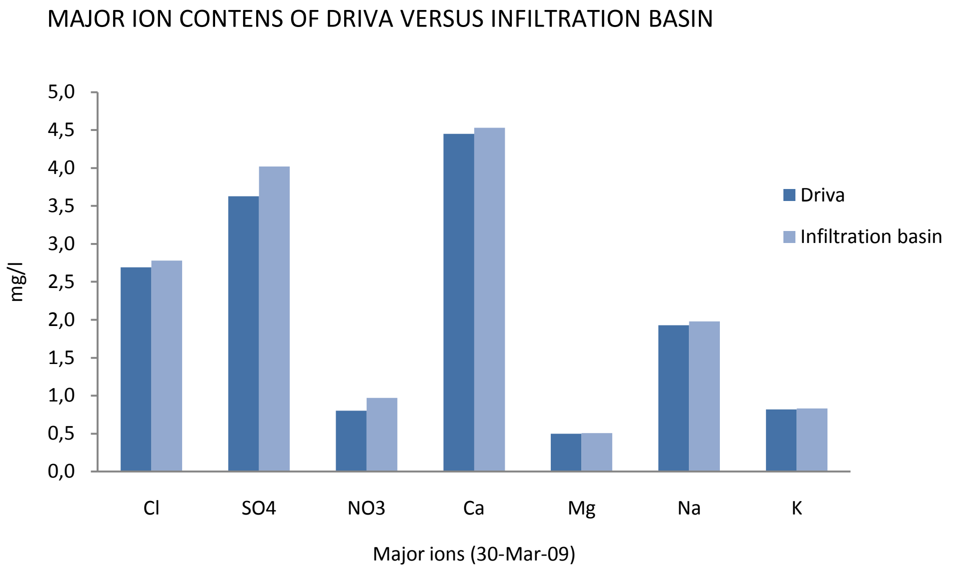


Figure 24: Major ion contents of Driva and the infiltration basin from the analysis of water samples (30-Mar-09).

Water from Driva and the infiltrations basin have quite equal concentrations of the major cations (Ca, Mg, Na, K) and the anions Cl, SO_4 and NO_3. Bicarbonate is not included in the diagram because alkalinity of the river water, as well as other physico-chemical parameters, was not reported for the analysis of samples from Driva. Since the chemical composition of the infiltration basin quite well reflects the conditions in Driva, sampling from the infiltration basin for analyses of physico-chemical parameters have been of priority throughout the test pumping.

Major ions in water from the infiltration basin and all production wells are presented for samples collected on the 17-Feb-09. The results from the water analysis of this date are reported with meq/l as concentration unit. Histograms are used to illustrate how major ion contents vary from source to source, and for a sampling date when well B was operating. In this case, sodium (Na) and potassium (K) are presented individually. Bicarbonate (HCO_3) has been estimated from values of alkalinity, with the argument that the groundwater hold a pH that is in the range that is dominated by this carbonate species (pH = 5-8). The balance between cations and anions are reflected in differences in column heights of anions (blue) and cations (green).

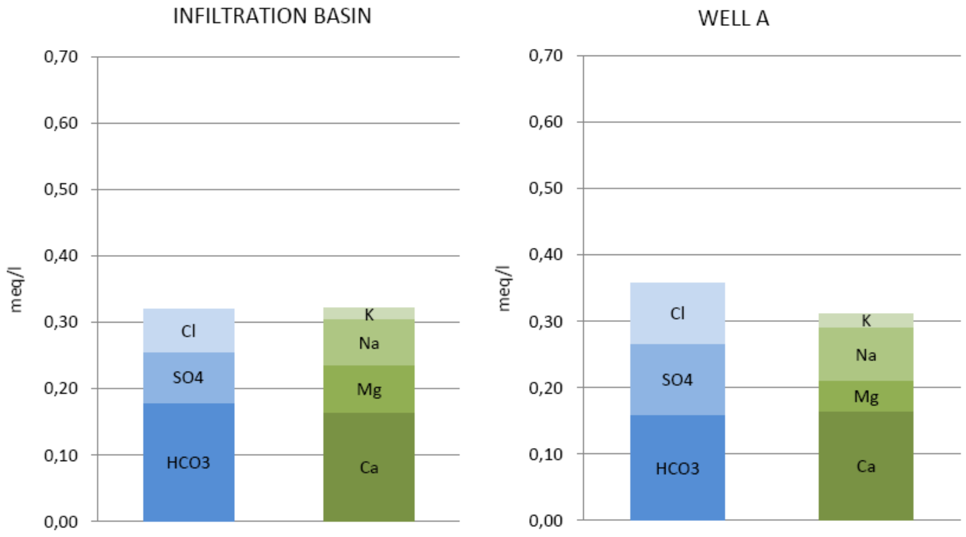


Figure 25: Major ion contents of water from the infiltration basin and well A (17-Feb-09).

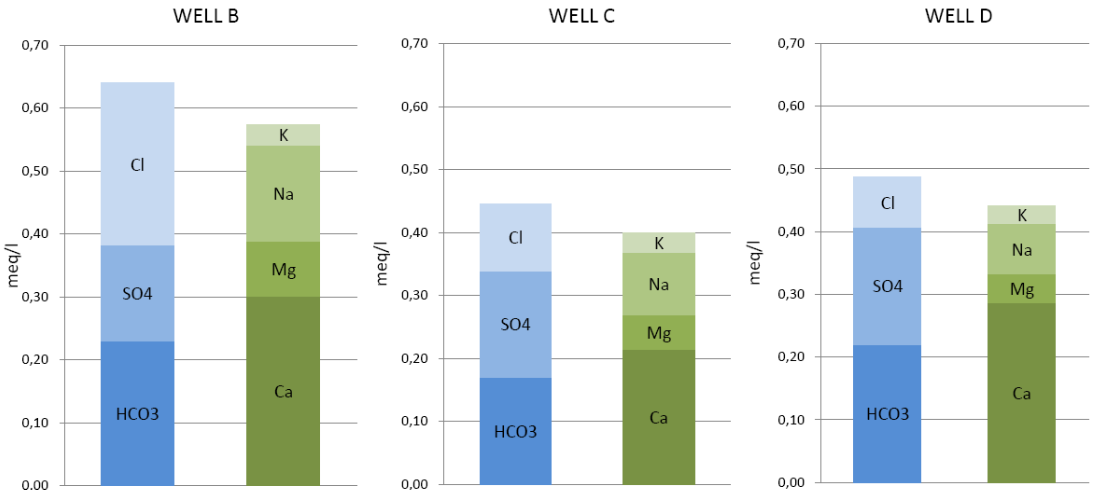


Figure 26: Major ion contents of water from wells B, C and D (17-Feb-09).

The infiltration basin and well A display a distribution of major ions that is very much similar considering the relative abundance of ions (Figure 25). Still, ion contents are slightly higher in well A. Both analyses indicate a faint ionic balance error.

Major ions of wells C and D are distributed in a quite similar manner compared to the infiltration basin and well A. Samples from well B, however, indicate that chloride dominates instead of sulphate and bicarbonate (Figure 26). The dominating cation in all three samples is calcium. Additionally, columns heights indicate the amount of dissolved solids. The sample from well B clearly shows the highest concentration of ions in this case. Analyses of samples from wells B, C and D indicate a minor ion balance error. Ion balance errors have been calculated for each analysis and can be found in *Appendix III*.

4.3.2 STATISTICAL PROPERTIES OF SELECTED PHYSICO-CHEMICAL PARAMETERS

A review of the properties of distribution of major hydrochemical parameters of the infiltration basin and all production wells is given. Selected parameters from all sampling locations are displayed in respective tables as average, maximum and minimum values. Evaluations of these parameters are made subsequently with respect to guide values defined by the *drinking water regulations*. All analyzed parameters from the test pumping period are presented in *Appendix III*. Numbers in bold indicate values exceeding the guide values.

INFILTRATION BASIN

All analyzed parameters of the infiltration basin fall within the requirements of *the drinking water regulations* (Table 17):

Table 17: Statistical properties of distribution of major hydrochemical parameters, infiltration basin (n=2).

Parameter	Unit	Minimum	Average	Maximum	Guide value
pH	pH	6,79	6,93	7,00	6,5-9,5
Electrical conductance	mS/m	3,40	3,57	3,73	250
Alkalinity	mmol/l	0,18	0,18	0,18	
TDS	meq/l		0,847		
Hardness	dH°		< 2-3,5		
SIC (Saturation index of calcite)			< 0		
Carbon dioxide *(n=2)*	mg CO_2/l	3,5	5,3	7,1	
Fe	mg/l	0,001	0,013	0,0245	0,2
Mn	mg/l	0,0005	0,002	0,0032	0,05
Ca	mg/l	3,31	3,41	3,51	
Na	mg/l	1,32	1,46	1,61	200
Mg	mg/l	0,44	0,46	0,478	
K	mg/l	0,66	0,74	0,82	
Si	mg/l	1,32	1,46	1,6	
Cl	mg/l	1,50	1,87	2,24	200
SO_4	mg/l	3,68	3,82	3,96	100
NO_3	mg/l	0,55	1,07	1,58	44

The basin water has an acidic character (pH < 7) and low alkalinity. The value of total dissolved solids (TDS) is also low and in line with the relatively low electrical conductance (EC) with an average value of 3,57 mS/m. Low values of alkalinity is experienced through the soft characteristic of the water (dH° < 2-3,5), which consequently can be related to relatively low concentrations of the cations calcium and magnesium. Estimates of CO_2-concentrations returned an average value of 5,3 mg CO_2/l. The saturation index for calcite (SIC) was defined < 0, indicating undersaturation with respect to the mineral calcite. The parameters pH, alkalinity and SIC show that this trend is valid for water from all production wells. Nitrate and phosphorous concentrations are low. This is also the case for the heavy metals, e.g. copper, zink and lead, which only exists with concentrations close to or below the detection limit (*Appendix III*).

WELL A

Groundwater extracted from well A shows that the quality is generally good. Except for iron and turbidity, all analyzed parameters meet the requirements of the regulations. A too high concentration of iron of 0,214 mg Fe/l was reported on the 31-Aug-09, representing the maximum value for this well during the period of test pumping. Turbidity reached 7,06 F.T.U. on the 15-Jun-09.

Table 18: Statistical properties of distribution of major hydrochemical parameters, well A (n=5).

Parameter	Unit	Minimum	Average	Maximum	Guide value
pH	pH	6,66	6,80	6,92	6,5-9,5
Electrical conductance	mS/m	4,28	4,38	4,48	250
Alkalinity	mmol/l	0,16	0,18	0,20	
TDS	meq/l		1,013		
Hardness	dH°		< 2-3,5		
SIC (Saturation index of calcite)			< 0		
Carbon dioxide, *(n=3)*	mg CO_2/l	4,2	6,2	9,5	
Fe	mg/l	0,001	0,07	**0,21**	0,2
Mn	mg/l	0,001	0,01	0,01	0,05
Ca	mg/l	3,54	4,00	4,55	
Na	mg/l	1,72	1,94	2,12	200
Mg	mg/l	0,54	0,58	0,61	
K	mg/l	0,78	0,92	1,07	
Si	mg/l	1,76	1,96	2,12	
Cl	mg/l	2,28	2,68	3,26	200
SO_4	mg/l	4,57	4,74	5,11	100
NO_3	mg/l	0,48	0,78	1,04	44

The groundwater has an acidic character (pH < 7) and low alkalinity. TDS is also low and is reflected through a relatively low EC with an average value of 4,38 mS/m. Low concentrations of calcium and magnesium result in a low degree of hardness (dH° < 2-3,5). The value of SIC indicates calcite undersaturation. CO_2-concentrations are on average 6,2 mg CO_2/l.

All samples show that nitrate and phosphorous concentrations are low, which is also the case for the heavy metals, e.g. copper, zink and lead that only exists with concentrations close to or below the detection limit (*Appendix III*).

WELL B

Well B produces groundwater with higher mean values of TDS and EC relative to the other production wells at the groundwater works. This is seen as an increase in concentrations of major ions, e.g. calcium, sodium, chloride and sulphate. The elevated values of iron are also characteristic of this well, with a mean value of 0,44 mg Fe/l reported during the test pumping. This is the only analyzed parameter exceeding the guide values. Maximum and minimum concentrations of iron are 0,416 and 0,463 mg/l, respectively.

Table 19: Statistical properties of distribution of major hydrochemical parameters, well B (n=2).

Parameter	Unit	Minimum	Average	Maximum	Guide value
pH	pH	6,59	6,625	6,66	6,5-9,5
Electrical conductance	mS/m	7,83	7,98	8,11	250
Alkalinity	mmol/l	0,21	0,22	0,23	
TDS	meq/l		1,742		
Hardness	dH°		< 2-3,5		
SIC (Saturation index of calcite)			< 0		
Carbon dioxide, *(n=2)*	mg CO_2/l	7,7	8,9	10,2	
Fe	mg/l	**0,416**	**0,440**	**0,463**	0,2
Mn	mg/l	0,026	0,026	0,026	0,05
Ca	mg/l	6,04	6,48	6,91	
Na	mg/l	3,50	3,68	3,86	200
Mg	mg/l	1,06	1,10	1,13	
K	mg/l	1,28	1,41	1,54	
Si	mg/l	3,01	3,15	3,28	
Cl	mg/l	9,13	9,19	9,24	200
SO_4	mg/l	7,31	7,74	8,17	100
NO_3	mg/l	0,65	0,65	0,65	44

The groundwater has an acidic character (pH < 7) and low alkalinity. However, the alkalinity is somewhat elevated relative to the other production wells. Despite slightly higher concentrations of major ions, this does not change the degree of hardness (dH° < 2-3,5). Groundwater of this well is still characterized as soft. The value of SIC indicates calcite undersaturation. The average value of CO_2-concentrations is 8,9 mg CO_2/l.

Concentrations of nitrate and phosphorous are low, and so is the case for the heavy metals that only exists with concentrations close to or below the detection limit (*Appendix III*).

WELL C

Groundwater from well C is generally of good quality. However, reported pH-values are too low in three out of five cases. The average pH-value is 6,47, while the minimum registered is 6,32. All other parameters meet the requirements of the regulations.

Table 20: Statistical properties of distribution of major hydrochemical parameters, well C (n=5).

Parameter	Unit	Minimum	Average	Maximum	Guide value
pH	pH	**6,32**	**6,47**	6,62	6,5-9,5
Electrical conductance	mS/m	5,41	6,43	9,65	250
Alkalinity	mmol/l	0,12	0,15	0,17	
TDS	meq/l		1,366		
Hardness	dH°		< 2-3,5		
SIC (Saturation index of calcite)			< 0		
Carbon dioxide, *(n=3)*	mg CO_2/l	6,3	10,2	15,0	
Fe	mg/l	0,004	0,034	0,107	0,2
Mn	mg/l	0,011	0,019	0,036	0,05
Ca	mg/l	4,30	4,62	5,08	
Na	mg/l	2,29	2,84	3,32	200
Mg	mg/l	0,66	0,78	0,838	
K	mg/l	1,16	1,37	1,63	
Si	mg/l	2,8	3,04	3,59	
Cl	mg/l	3,79	4,67	5,48	200
SO_4	mg/l	7,48	8,06	8,34	100
NO_3	mg/l	0,61	0,74	0,90	44

Alkalinity is low and the groundwater is soft (dH° < 2-3,5). The minimum value of alkalinity for this well represents the lowest value of all productions wells. TDS is higher than what is the case for the basin, but generally lower than well B. This is also the case for the low electrical conductance (EC) with an average value of 6,43 mS/m. The maximum value of EC reported for this well is markedly higher than in water from all the other wells.

The value of SIC indicates calcite undersaturation. CO_2-concentrations are greater compared to the other wells with an average value of 10,2 mg CO_2/l. It has to be taken into consideration that values of well B only reflects two samples (n=2). Concentrations of nitrate and phosphorous are low, which is also the case for the heavy metals, e.g. copper, zink and lead, that only exists with concentrations close to or below the detection limit (*Appendix III*).

WELL D

Test pumping of well D has documented that the well produces groundwater of generally good water quality throughout the period. On one occasion, on the 2-Jun-09, the pH-value reported was only 6,39, which is below the guide values stating that the pH should be in the range of 6,5-9,5.

Table 21: Statistical properties of distribution of major hydrochemical parameters, well D (n=6).

Parameter	Unit	Minimum	Average	Maximum	Guide value
pH	pH	**6,39**	6,635	6,92	6,5-9,5
Electrical conductance	mS/m	5,53	5,93	6,22	250
Alkalinity	mmol/l	0,16	0,18	0,22	
TDS	meq/l		1,379		
Hardness	dH°		< 2-3,5		
SIC (Saturation index of calcite)			< 0		
Carbon dioxide, *(n=3)*	mg CO_2/l	5,0	8,5	13,1	
Fe	mg/l	0,001	0,013	0,041	0,2
Mn	mg/l	0,005	0,015	0,022	0,05
Ca	mg/l	5,36	5,63	6,02	
Na	mg/l	1,84	2,40	2,77	200
Mg	mg/l	0,56	0,68	0,768	
K	mg/l	1,13	1,31	1,62	
Si	mg/l	2,01	2,87	3,23	
Cl	mg/l	1,35	3,35	4,33	200
SO_4	mg/l	8,03	8,97	10,2	100
NO_3	mg/l	0,58	0,75	1,00	44

The same characteristics can be given of pH and alkalinity in the case of this well as well; the water is acidic in character (pH < 7) and has a low alkalinity. In accordance with the dH° estimated to be < 2-3,5, the water is characterized as soft. SIC-values indicates calcite undersaturation, and the average CO_2-concentration is 8,5 mg CO_2/l. Nitrate is present only in low concentrations and phosphate only twice crosses the detection limit. Heavy metals exist, but with concentrations close to or below the detection limit (*Appendix III*).

4.3.3 PHYSICO-CHEMICAL ASPECTS

Average values of four redox species from throughout the test pumping are presented for the infiltration basin and all production wells. Values of pH are presented with iron and manganese in Figure 27.

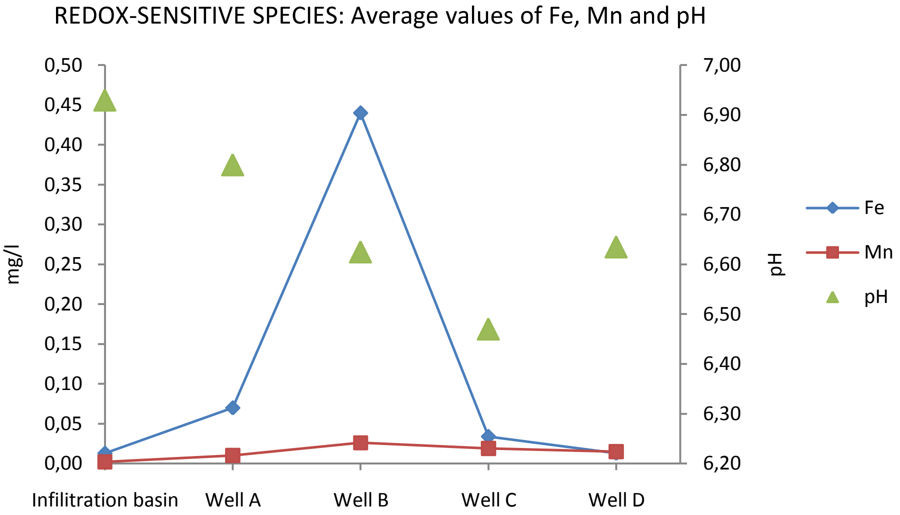


Figure 27: Average concentration of iron and manganese with pH-values of samples from the infiltration basin and all wells.

Water from well B has the highest concentrations of iron and manganese of all wells, while well C and D have the lowest values of these species. Well A is moderate with respect to iron, but represents the well with lowest manganese concentrations. pH-values increase from the infiltration basin towards well A and further to well B. Lowest values of pH is reported for well C. Well D is comparable with well B for this parameter.

Figure 28 shows that the sulphate concentrations are greatest in the case of wells B and D. Nitrate on the other hand, occurs with highest concentrations in well C. Well B has the lowest value of all for this specie.

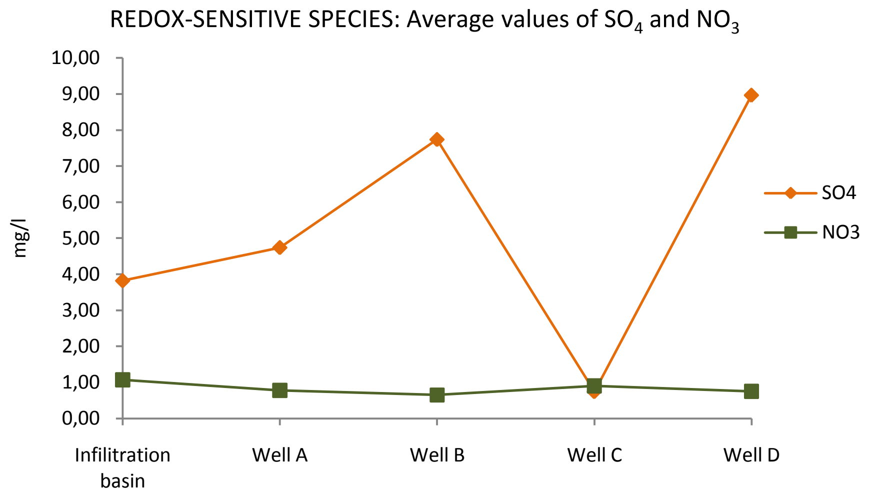


Figure 28: Average values of sulphate and nitrate of samples from the infiltration basin and all wells.

Variations in iron concentrations during the test pumping period are displayed for samples from all wells (Figure 29). The guide value of 0,2 mg Fe/l is represented by a dashed line. Well B was closed down in the beginning of June, which results in only two values of iron from this well.

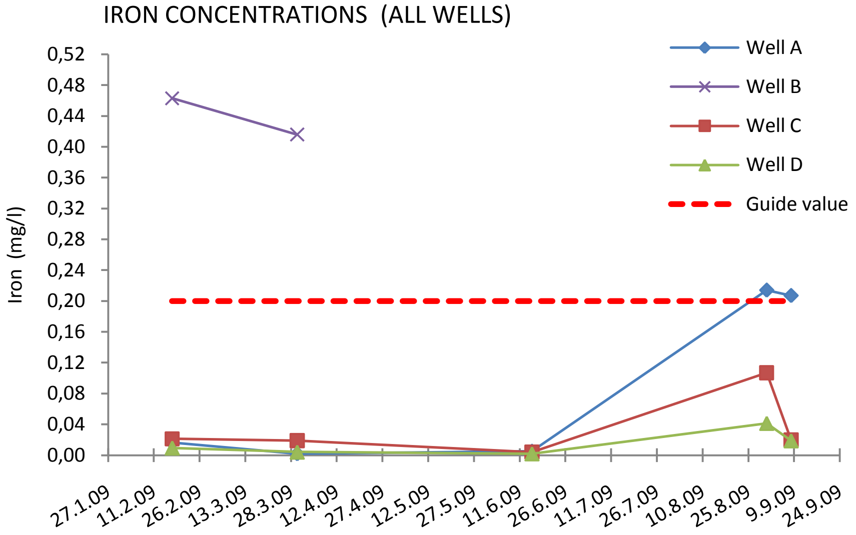


Figure 29: Iron concentrations in water from all wells during the test pumping period displayed together with the guide value of 0,2 mg Fe/l.

Piper-diagrams are used in order to interpret the hydrochemical facies of groundwater samples. The diagrams (Figure 30 and 31) include data from the first and last analysis of the test pumping, the 17-Feb-09 and the 8-Sep-09, respectively. All wells except well B are represented in both diagrams.

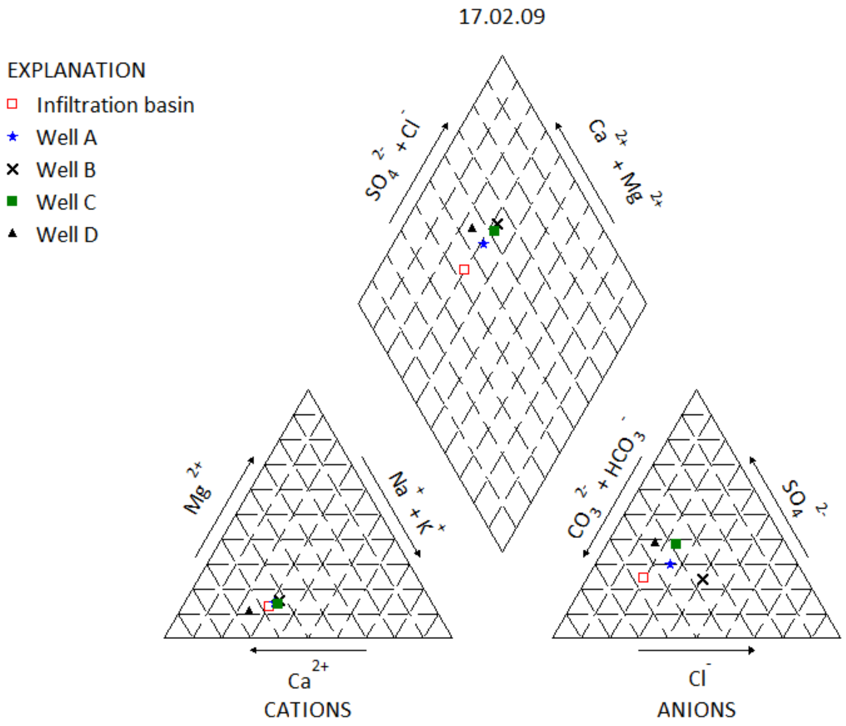


Figure 30: Piper-diagram displaying the location of samples from the infiltration basin and all wells with respect to major ion contents (17-Feb-09).

Calcium is the dominating cation of all wells and the infiltration basin, but wells B and C plot close to the border of no dominant cations. The picture is slightly changed with respect to the anions, in that the infiltration basin plots in the area of bicarbonate. All wells plot in the middle of the triangle with anions, which represents water with no dominant type. The hydrochemical facies derived from the diamond-shaped diagram can be divided into two groups of waters:

- Infiltration basin: Ca-HCO_3-type
- Wells: No dominant type (Ca-HCO_3-type/Ca-SO_4-type)

Figure 31 shows there is no distinctive change in chemical facies with time (from the 17- Feb-09 and the 8-Sep-09) through the test pumping period. This figure represents a composition with well B shut down. The dominating cation is calcium for all samples except well C, which lies on the transitions towards the area with no dominating cations. With respect to anions, a slight change has occurred in the wells. Well A is now less dominated by bicarbonate, and has shifted into the triangle of no dominant anions. This is also to case for wells C and D that now is located more into the middle of the triangle of no dominant anions.

The hydrochemical facies derived from the diamond-shaped diagram can be divided into two groups of waters:

- Infiltration basin: Ca-HCO_3-type
- Wells: No dominant type → Ca-SO_4-type

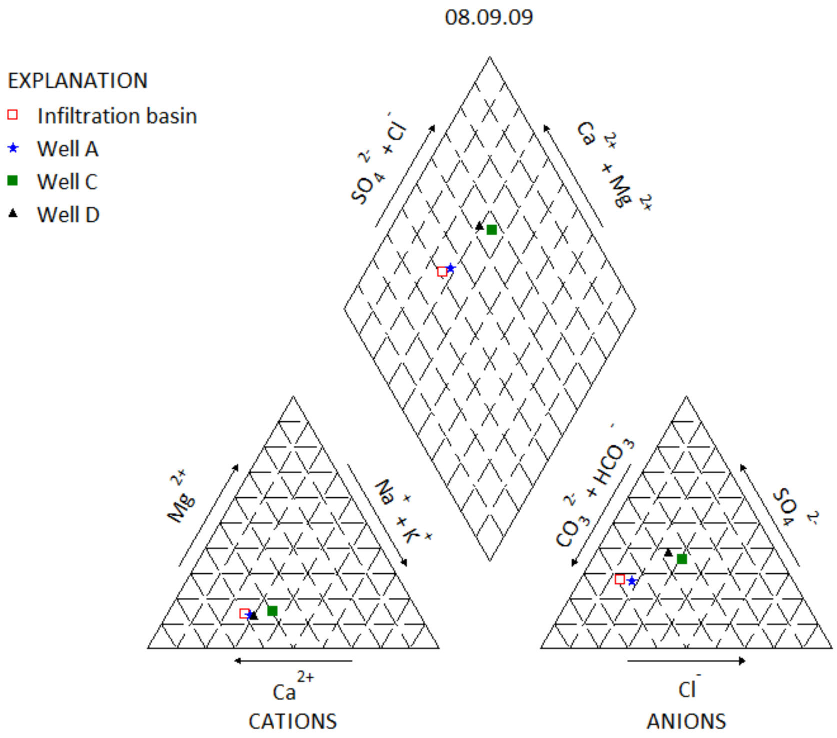

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Figure 31: Piper-diagram displaying the location of samples from the infiltration basin and all wells (except well B) with respect to major ion contents (8-Sep-09).

4.3.4 FIELD ANALYSES: PHYSICO-CHEMICAL PARAMETERS

Temperature, EC and pH were measured in field during *Stages 2 and 3* (Figures 32 and 33). All wells were in production during *Stage 2*, while data shown for *Stage 3* records a situation when well B was cut off.

Samples from Driva, the infiltration basin and well A show these sources have quite similar values of EC and temperature (Figure 32). Comparing the wells only, there is a slightly greater range in these values. Well B has highest EC of all, with wells C and D following. Well A seems to be producing groundwater with relatively low EC. By comparing wells C and D, it seems that well C produces water with somewhat higher values than well D. The temperature of river water is higher than both the infiltration basin and well A on the 30-Mar-09. Well C holds the highest temperature of all sources combined. pH is on the acidic side for all wells, but is close to neutral for Driva and the infiltration basin.

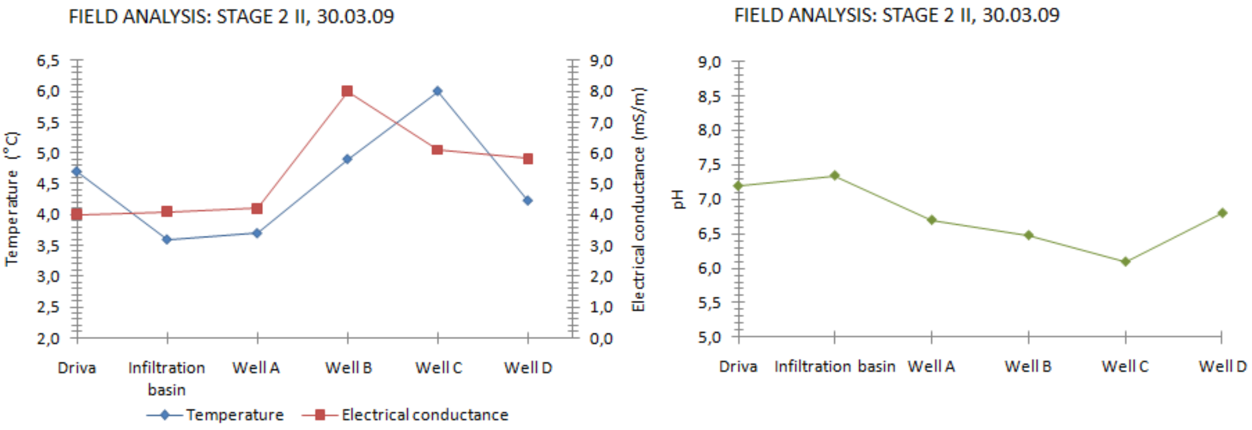

Figure 32: Temperature, EC and pH measured in field for samples from Driva, the infiltration basin and all wells (30-Mar-09).

The picture is somewhat changed for samples analyzed on the 8-Sep-09 (well B closed). The temperature has increased for all sources from c. 3,6-6,0 °C in March to some 9,4-11,1 °C in September. EC is now highest in the case of well D, with wells C and A following with progressively lower values. pH-values are slightly lowered this month compared to the values in March.

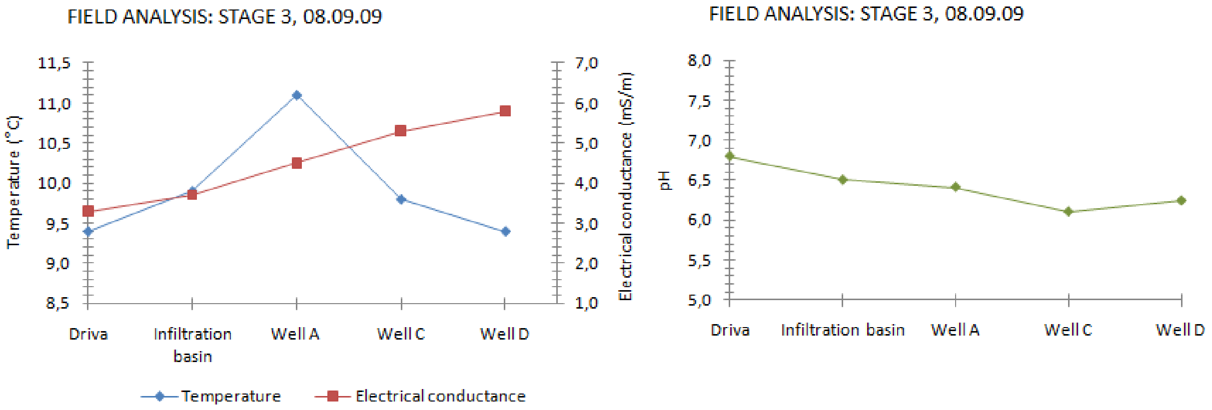

Figure 33: Temperature, EC and pH measured in field for samples from all wells (except well B), Driva and the infiltration basin (8-Sep-09).

Analyses of iron, manganese and ammonium were conducted on two occasions (Table 22). During *Stage 2 Step-drawdown test I,* iron concentrations were measured for samples from wells B, C and D (first row.) As part of *Stage 3*, samples from wells A, C and D were measured for iron, manganese and ammonium concentrations. Apart from well B, all samples reflect a good water quality. The iron concentration in water from well B by far exceeds the limit with a value of 0,82 mg/l.

Table 22: Iron, manganese and ammonium concentrations measured in field with Photometer.

Parameter	Unit	A	B	C	D
Fe	*mg/l*		***0,82***	*0,12*	*0,05*
Fe	mg/l	0,15		< 0,05	0,07
Mn	mg/l	< 0,04		< 0,04	< 0,04
NH_4	mmol/l	< 0,0014		< 0,0014	< 0,0014

4.3.5 CONTENTS OF ORGANIC MATTER

Implications of organic matter are reported from estimated concentrations of dissolved CO_2. Field measurements of temperature and pH, as well as laboratory analyses of calcium concentrations and alkalinity make the basis for the values presented in Table 23.

Table 23: Concentrations of dissolved carbon dioxide (mg CO_2/l).
(Relevant field measurement not conducted.)*

	Inf.basin	Well A	Well B	Well C	Well D
17-Feb-09	3,5	4,2	7,7	6,3	5,0
30-Mar-09	-*	5,0	10,2	9,4	7,3
08-Sep-09	7,1	9,5	*closed*	15,0	13,1

CO_2-concentrations vary spatially and with time for the locations in question. The lowest values are represented by samples from the infiltration basin, while samples from well B have the highest concentrations of all locations. Following well B is well C, then well D, and finally well A with the lowest concentrations of all production wells. There is also a trend in a steady increase in concentrations for all locations from February to September.

In addition to analyses of microbes in samples collected on the 30-Mar-09 (section 4.4), organic matter was measured as total dissolved carbon (TOC) and dissolved organic matter (DOC). These values represent spot tests. Table 24 shows concentrations in mg/l of carbon for Driva, the infiltration basin and well A. Well A was chosen with the argument that this well was expected to have the lowest residence time with respect to distance to the river.

Table 24: Contents of organic matter measured as TOC and DOC, 30-Mar-09.

	Driva	Inf. basin	Well A
TOC (mg C/l)	1,7	1,2	1,0
DOC(mg C/l)	1,3	1,1	0,8

The sample from Driva has slightly more elevated values of TOC and DOC with respect to well A and the infiltration basin. Groundwater from well A shows the lowest values of all sources.

4.3.6 EFFECTS OF NATURAL (SEASONAL) VARIATIONS

Selected parameters of samples from wells A, C and D (*parameter (unit) = well A; well C; well D)* are shown with respect to river discharge (Figure 34). A flooding event in Driva in August/September is in focus here, reflecting a change in groundwater chemistry of all wells. Iron, calcium and EC have been selected as parameters to illustrate this.

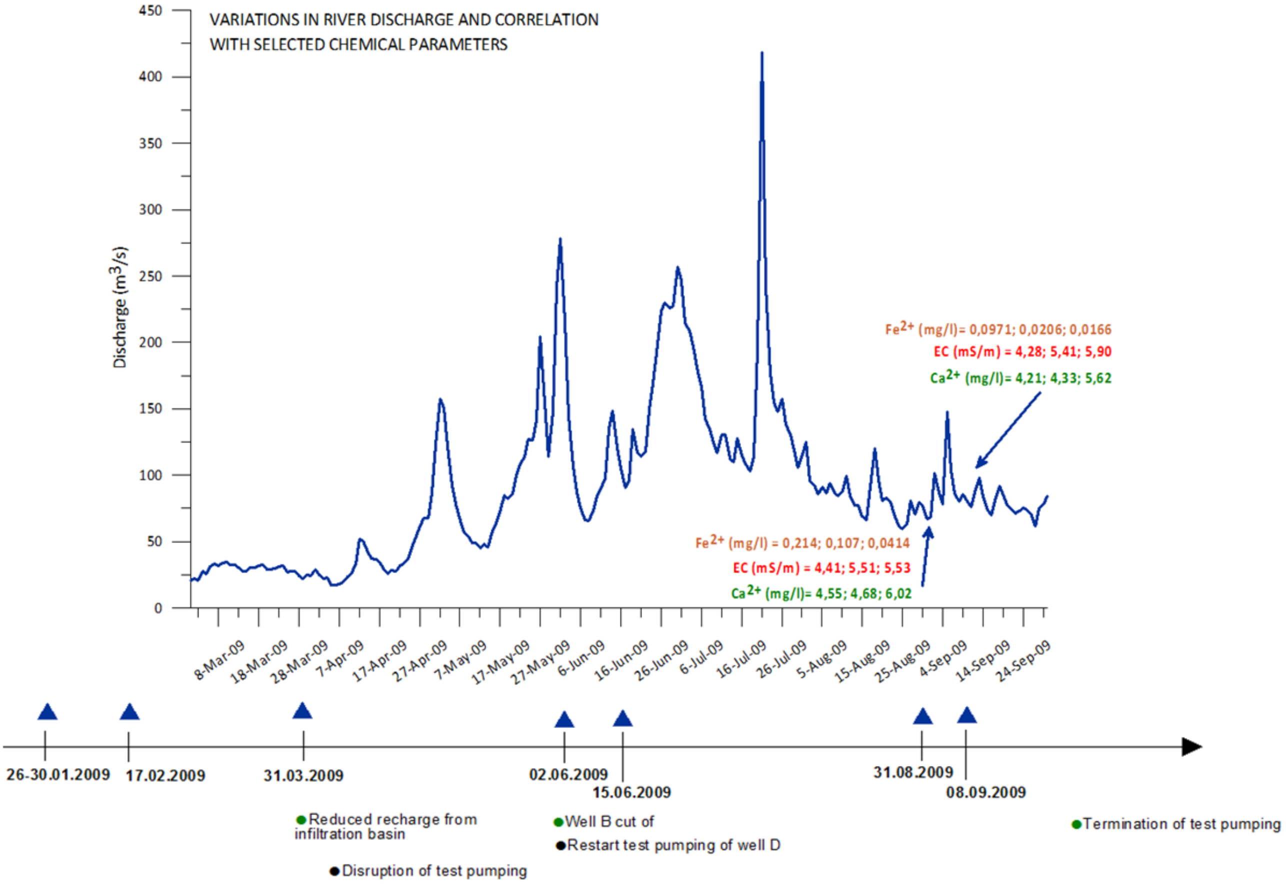

Figure 34: Variations in river discharge and correlation with selected chemical parameters (EC, calcium and iron).

Prior to the flooding, iron concentration in the sample from well A was reported with a level too high with respect to the guide value. This picture changed shortly after the discharge peak, and was registered as a significant decrease in all parameters. Well C can be seen to respond in a similar manner, in that a decrease was experienced for all parameters subsequent to the period of increased discharge. Well D confirms this effect for calcium and iron. The exception is EC, which on the other hand showed an increase.

It is also evident from the water chemistry that the wells experience an increase in EC (except well A), which represents a dry spell from February to March (Figure 4). This is also seen for iron and calcium with the exceptions of wells D and A. A decreasing trend of all values is seen for wells C and D for the 15-Jun-09 as discharge increases. Well A shows on the other hand an increase.

4.3.7 EFFECTS OF DIFFERENT OPERATIONAL SITUATIONS

CLOSING DOWN OF WELL B

Well B was closed down by the water works in May/June in accordance with the aims of the study. The effects of closing down the well are illustrated with concentrations of iron and calcium in samples from well A (Figure 35). A slight increasing trend in the beginning of the test pumping period can be seen. Later, and short after the closing, samples from Well A documented an effect of the new operational situation. Iron and calcium concentrations increased quite rapidly and appear to peak late in August with the highest value reported on the 31-Aug-09 (0,214 mg Fe/l). Well A was the only well reported with iron values exceeding the guide value. Effect on wells D and C is not that distinct, but can be viewed in Figure 29 and Tables 20-21.

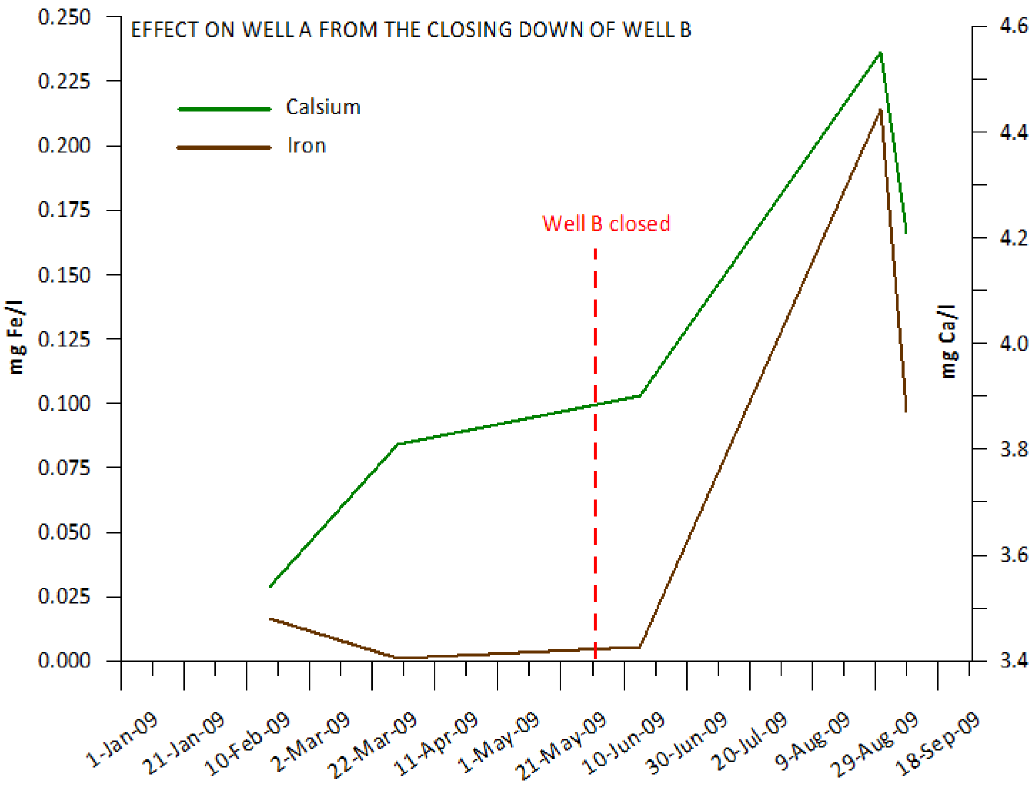

Figure 35: Effect on iron and calcium concentrations of well A from the closing down of well B in the beginning of June.

EFFECTS FROM ARTIFICAL RECHARGE THROUGH THE INFILTRATION BASIN

The effect on water quality is somewhat difficult to document, because the amount of infiltrated water through the basin is closely conditioned by the river discharge at all times. Periods of decreased infiltration capacity coincides with episodes of reduced discharge, meaning that the river's hydrological character camouflages any effects of infiltration through the basin floor. This is due to the dependency on water supply from the river intake that is situated in the river. Based on available water chemistry data, it is not possible to present any results that could document the isolated effect of the infiltration basin on the groundwater chemistry.

The groundwater works, on the other hand, could report that increased problems with iron occurred in this period (Personal comment # 1).

A possible effect of the infiltration basin on groundwater chemistry is implied indirectly from groundwater level maps that are presented in section 4.5.

4.4 MICROBIOLOGY

This section concentrates on issues related to the microbiological quality of the water. Guide values related to microbiological parameters are displayed in *Appendix V*.

Microbiological spot tests of water from Driva, the infiltration basin and well A analyzed with respect to selected microbiological parameters are shown in Table 25. The analysis showed that the microbiological quality of the water is good with exception of the sample from Driva, which returned a result of 2 coliforms per 100 ml. This value exceeds the guide value.

Table 25: Microbiological situation of Driva, the infiltration basin and well A, 30-Mar-09. (Values are number of detections per 100 ml.)

	Driva	Inf. basin	Well A
Coliforms	2	0	0
Escherichia coli (e.coli)	0	0	0
Intestinal enterococci	0	0	0

Analyses of the respective concentrate (Driva, infiltration basin and well A) were also made for the detection of fecal contamination. The indicator bacteria used in this case was *Escherichia coli* (e.coli). The only source for fecal contamination was Driva with 3 detections per 100 ml (*Appendix V*). This exceeds the guide value.

Bacteriological analyses have been administrated by Sunndal municipality throughout the test pumping. Samples were collected by the municipality on the 31-Aug-09 (Table 26). Wells A, C and D, as well as the transmission system, were included in the analysis. Detections of bacterial counts were made for well A and D, the value from well D being the greatest, representing a concentration of 450 per ml that by far exceeds the limit.

Table 26: Bacteriological analysis administrated by Sunndal municipality, 31-Aug-09.

	Unit	Well A	Well C	Well D	Transmission system
Coliforms	/100 ml	0	0	0	0
Escherichia coli (e.coli)	/100 ml	0	0	0	0
Bacterial counts	/ml	6	0	**450**	0

4.5 GROUNDWATER FLOW PATTERN

Groundwater extraction from the production wells and aquifer recharge are factors that varies throughout the entire measurement period. Conductance of a different operational setting was tested with the closing of well B. In addition to constructed operational settings, e.g. the closing of well B and the varying capacity of the infiltration basin, levels are influenced by natural variations defined by the different components of recharge. Aquifer recharge will be outlined in more detail in section 4.7.

4.5.1 GROUNDWATER LEVEL IN PIEZOMETER WELLS

Fluctuations in groundwater level that were observed during the test pumping, shows that withdrawal from well D to a great extent is responsible for drawdown in the aquifer (Figure 36). A lowering of the water table resulting from withdrawal from this well is especially distinct in an E-SE direction, as well as towards the south (Figures 37-40). With respect to an increase in the area of influence and piezometer wells, increased drawdown is most evident for piezometer wells 23, 22 (Ub-2), 14, 13 and 12.

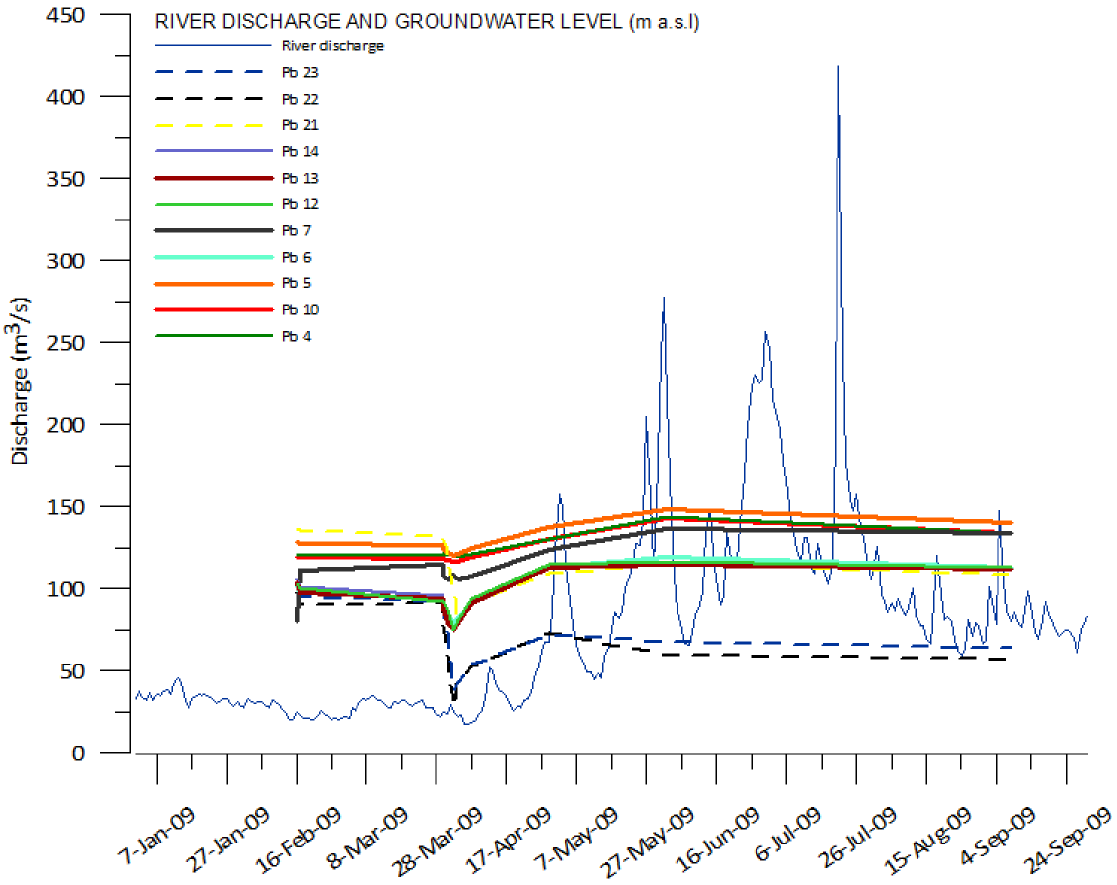

Figure 36: River discharge and groundwater level measured in piezometer wells.

4.5.2 GROUNDWATER FLOW

Groundwater flow during the test pumping period is presented with the aid of groundwater level maps. These maps are the result of measurements presented in Figure 36. An overview of different situations at the groundwater works during the test pumping period was outlined in Figure 14. However, a short introduction to the prevailing situations at the works is provided:

- Groundwater level map I: Level in advance of *Stage 2 Step-drawdown test II.*
- Groundwater level map II: Level on the day subsequent to initiation of *Stage 2*, and the first day of *Stage 3 Test pumping.*
- Groundwater level map III: Level after the closing down of well B and initiation of the second test pumping of well D.
- Groundwater level map IV: Level after approximately three months of continuous pumping from well D.

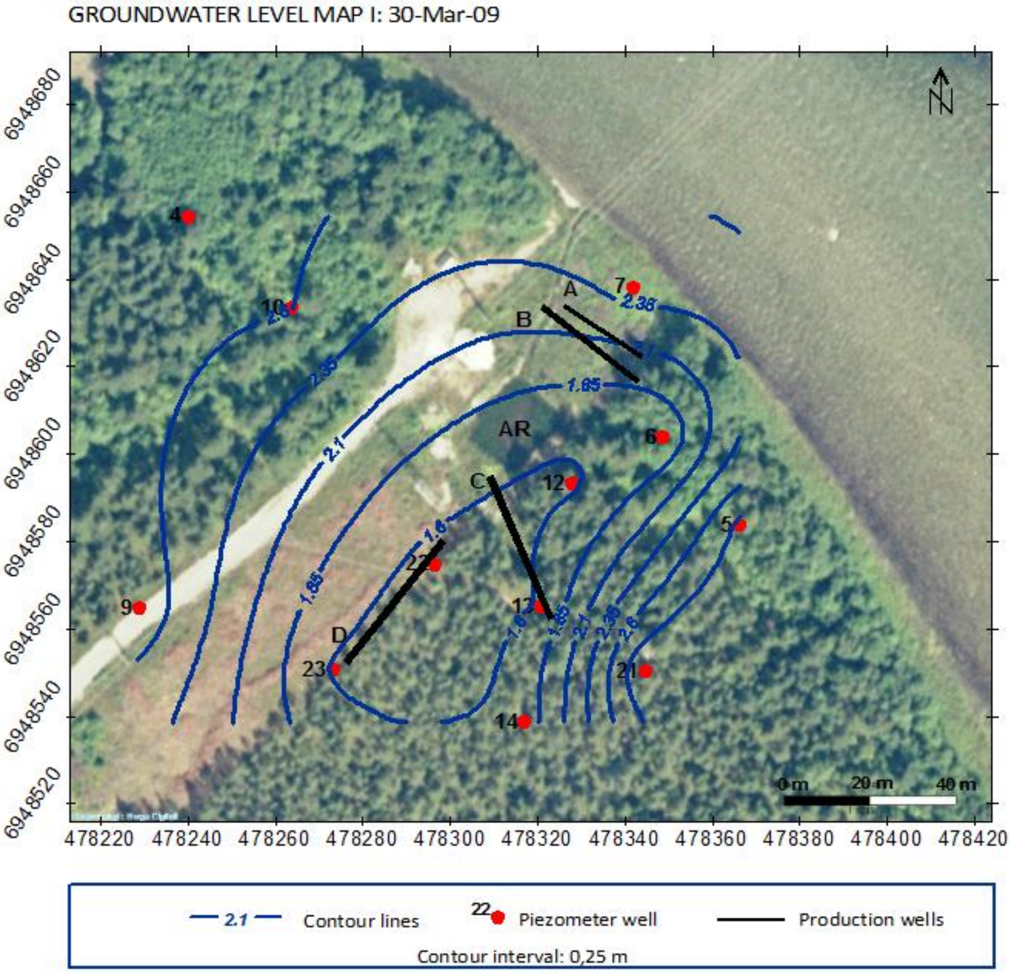

Figure 37: Groundwater level in advance of Stage 2 Step-drawdown test II (30-Mar-09). AR (artificial recharge) denotes the location of the infiltration basin.

The dominating groundwater flow in the vicinity of the works is directed towards the well area with a somewhat more distinct drawdown developed around wells C and D (Figure 37). Flow towards these wells are a combination of water infiltrating from Driva and the infiltration basin. Water from Driva also infiltrate in vicinity of wells A and B. The groundwater level downstream from the works indicates a flow towards the well area and the infiltration basin. Flow appear to be slightly concentrated towards well C from piezometer well 21 (Ub-1). Flow from the W-SW and the mountains is not predominant in this case.

On the second day of test pumping, 31-Mar-09, a greater depression of the groundwater table developed around well D (Figure 38). The well screen is located below piezometer 22 and extends towards the infiltration basin. Flow also occurs from the infiltration basin to the aquifer in the direction of well D. Infiltrated basin water to a greater extent contributes to feed well D relative to the other production wells.

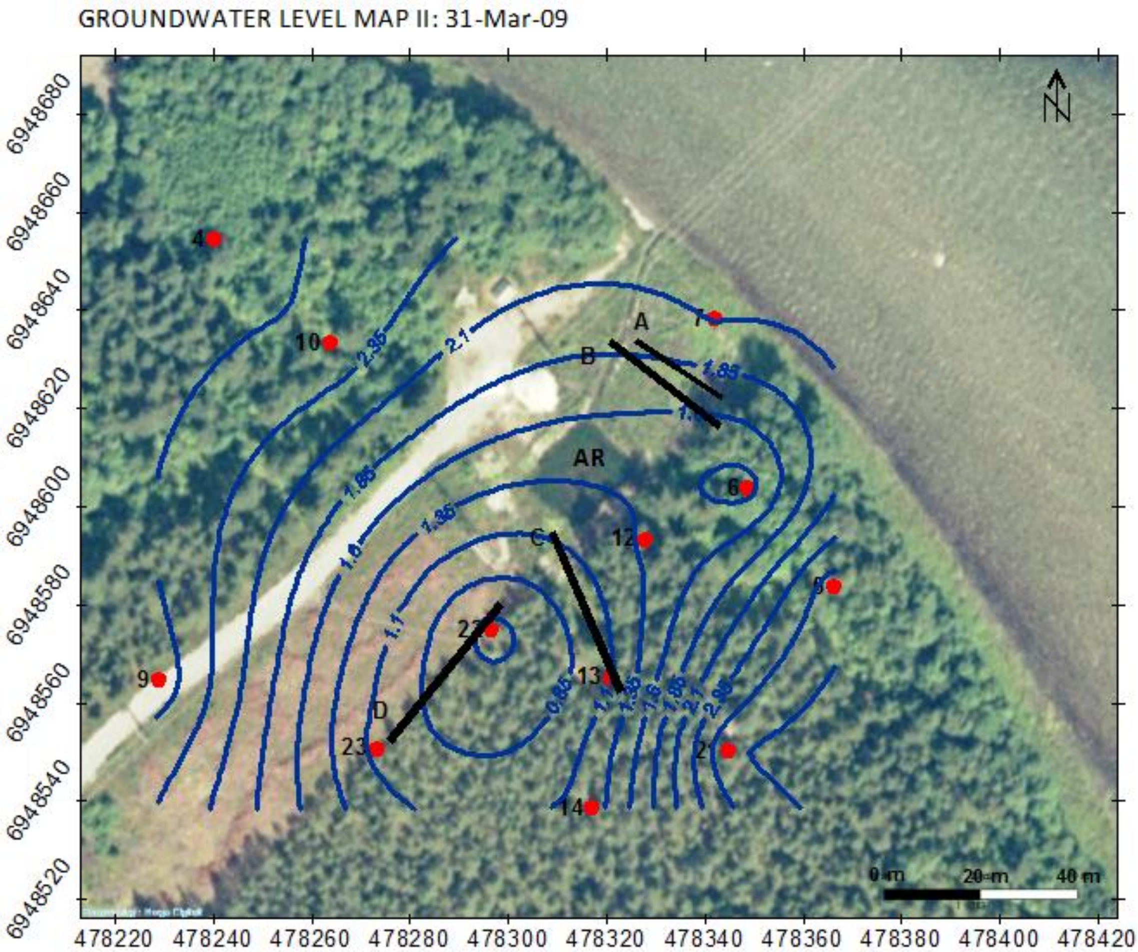

Figure 38: Groundwater level on the day subsequent to initiation of Stage 2, and the first day of Stage 3 Test pumping (31-Mar-09).

The locations of wells A and B are now indicated with a minor lowering of the groundwater level around piezometer well 6 (Pb 6) contra the level documented on the previous day, 30-Mar-09. The withdrawal from well C is on the other hand not that clear in this view (the withdrawal from well C is not known for this month).

Figure 39 displays the flow pattern at the time of the closing down of well B. This coincides with the initiation of the second test pumping of well D during. The drawdown around well D is greater now than what was indicated from the previous maps. The recharge from the infiltration basin to the aquifer is still evident, and slightly more prevalent than in March.

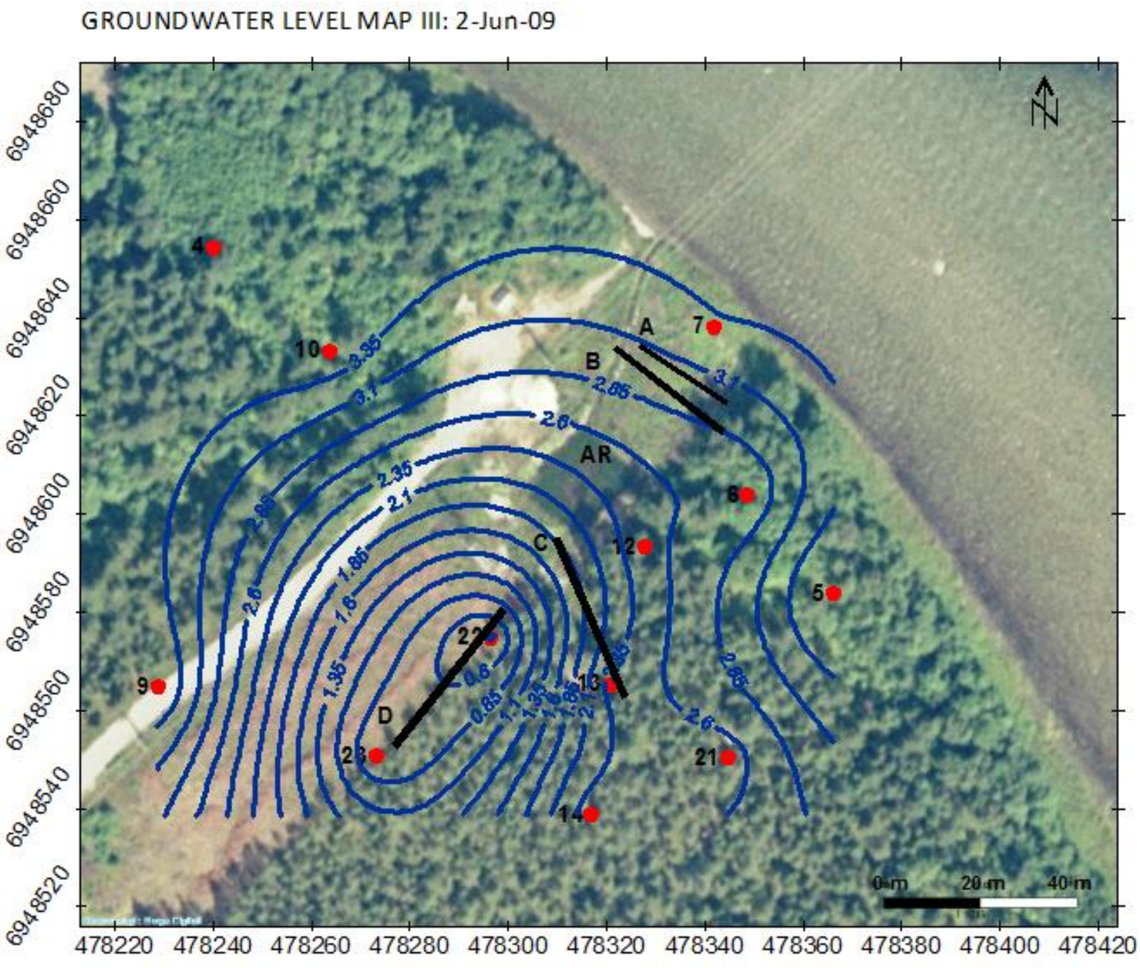

Figure 39: Groundwater level after the closing down of well B and initiation of the second test pumping of well D (2-Jun-09).

Flow pattern has not changed substantially since June, and the withdrawal from well D is still evident as a clear drawdown (Figure 40). Flow towards well A is no longer that prominent compared to the previous presentations. The infiltration basin still feeds the aquifer.

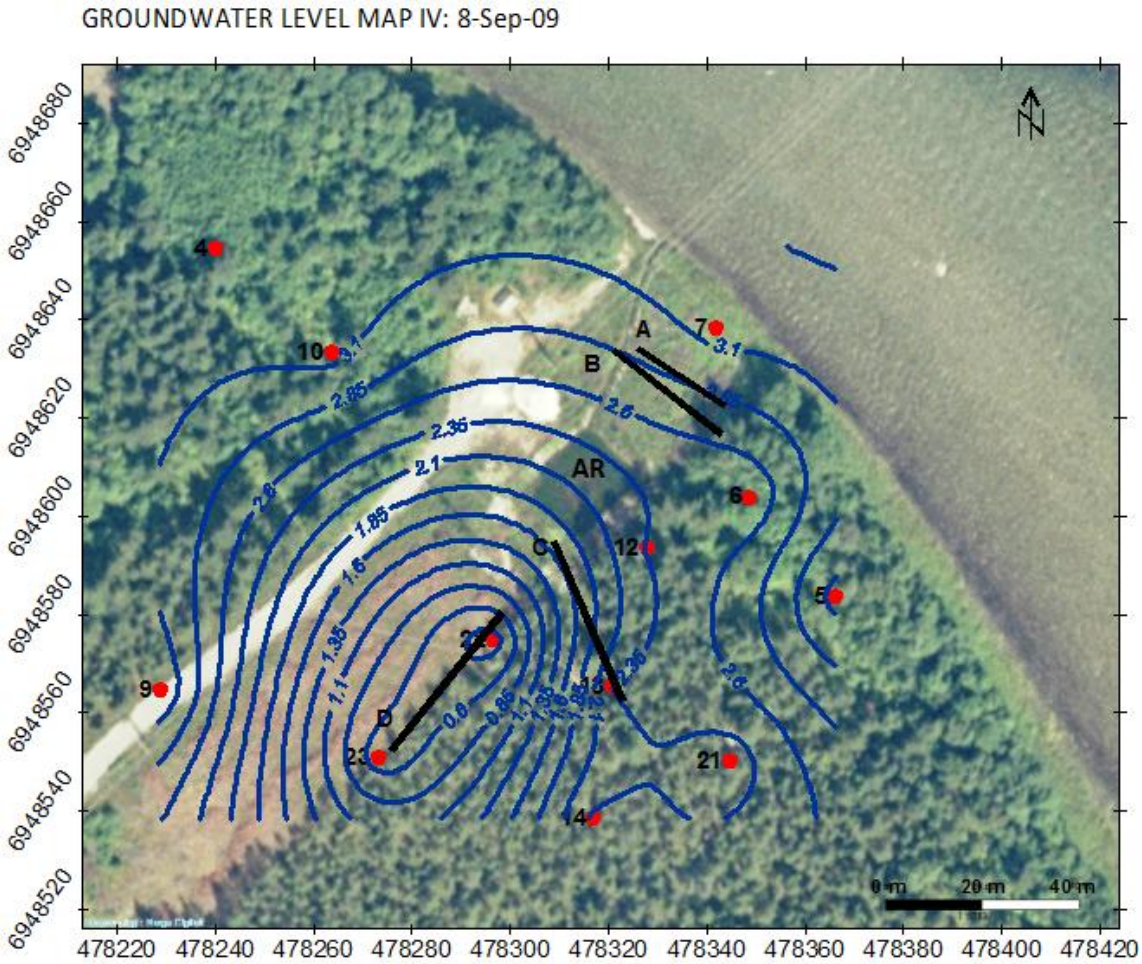

Figure 40: Groundwater level after approximately three months of continuous pumping from well D (8-Sep-09).

4.6 RESIDENCE TIME

Residence times have been estimated on the basis of a combination of flow patterns and definitions of aquifer hydraulic properties. Additional aspects of residence time are discussed in section 5.1.5.

4.6.1 RESIDENCE TIME BASED ON GROUNDWATER FLOW AND AQUIFER HYDRAULICS

Two sets of residence time data, representing two dates, is presented for a total of four flow lines of the aquifer in order to give an indication of how conditions vary temporally (seasonally) and spatially. Data from March are applied to reflect the situation during dry spells with consequently reduced river discharge and recharge through the infiltration basin. On the other hand, data from September is applied to show how the system reflects a period of increased recharge.

Hydraulic parameters and groundwater flow patterns were delineated in sections 4.1.2 and 4.5, respectively, and make the basis for estimating residence time. Tables 27-30 display data used to calculate the hydraulic gradient needed to define flow velocity, as well as the resulting residence time reported in number of days. K-values represent an average value calculated from each of the two investigation wells. *Appendix II* outlines the approach of this method.

Residence times of water flowing from the infiltration basin towards well D are apparently greater in March than later in September. Travel times are less than five and two days, respectively.

Table 27: Estimate of residence time of water infiltrating from the infiltration basin → well D.

Infiltration basin → well D							
Date	K (m/s)	*dh* (m)	*dl* (m)	i	v (m/s)	v (m/day)	t (days)
31-Mar-09	$4{,}52 \cdot 10^{-4}$	0,38	19,09	0,0199	$4{,}49 \cdot 10^{-5}$	3,9	**4,9**
8-Sep-09	$4{,}52 \cdot 10^{-4}$	0,75	16,36	0,0458	$1{,}034 \cdot 10^{-4}$	8,9	**1,8**

The temporal relationship seen in Table 27 is also valid for groundwater flowing from the gravelled road to the NE of the well area and towards well D. Table 28 is presented to imply what the travel time would be for pollution deriving at well D from the gravelled road to the NE of well D.

Table 28: Estimate of residence time of water flowing from the gravelled road NE of well D → well D.

Gravelled road to the NE of well D → well D							
Date	K (m/s)	*dh* (m)	*dl* (m)	i	v (m/s)	v (m/day)	t (days)
31-Mar-09	$4{,}52\cdot10^{-4}$	1,5	63,63	0,0236	$5{,}33\cdot10^{-5}$	4,6	**13,8**
8-Sep-09	$4{,}52\cdot10^{-4}$	2,5	56,358	0,0444	$1{,}0\cdot10^{-4}$	8,7	**6,5**

Flow from piezometer well 21 (Ub-1) was only defined for March, as the flow lines in vicinity of Pb 21 are complex in this case (Table 29). Residence time approximates the value found for groundwater flowing from the infiltration basin in direction of well D of a little less than five days.

Table 29: Estimate of residence time of water flowing from piezometer well 21 (Ub-1) → well D.

Pb 21 (Ub-1) → well C							
Date	K (m/s)	*dh* (m)	*dl* (m)	i	v (m/s)	v (m/day)	t (days)
31-Mar-09	$4{,}85\cdot10^{-4}$	1,5	19,1	0,019	$4{,}5\cdot10^{-5}$	10,6	**2,6**
8-Sep-09					*Not defined.*		

Flow from Driva in the direction of wells A and B are dominated by longer residence time than what is seen in for the other flow lines. Residence times estimated for these dates are influenced by a low and moderate discharge of Driva, respectively. In order to account for this, flow lines have been extended to show a travel time in line with the river water level 31-Mar-09. Values in Table 30 do not take into consideration the different location of screens of wells A and B. This makes these values are more representative of the situation involved with well A.

Table 30: Estimate of residence time of water infiltrating from Driva → wells A and B.

Driva → wells A + B							
Date	K (m/s)	*dh* (m)	*dl* (m)	i	v (m/s)	v (m/s)	t (days)
31-Mar-09	$4{,}52\cdot10^{-4}$	0,5	60	0,0083	$1{,}95\cdot10^{-5}$	1,7	**35,5**
8-Sep-09	$4{,}52\cdot10^{-4}$	0,5	51	0,0098	$2{,}3\cdot10^{-5}$	2,0	**25,6**

4.7 SOURCES OF GROUNDWATER RECHARGE

Recharge is the process that allows the entry of water into the saturated zone. Three occurrences of recharge have been identified as contributors to this aquifer. The most important occurs by means of induced infiltration of river water from Driva. The second is attributed water infiltrating through the basin floor of the infiltration basin. Recharge is also achieved as a result of water infiltrating the permeable flood plain entering the saturated zone. The water enters the saturated zone after residing for some time in the unsaturated zone.

4.7.1 RIVER INFILTRATION (INDUCED INFILTRATION)

Infiltration from Driva is controlled by groundwater withdrawal at the works (induced recharge) as a response to the gradient that is established from the river towards the reservoir. This setting controls the interaction between the river and the groundwater basin to a great extent. Driva is a regulated river, but still it shows clear seasonal variations in discharge. Figure 4 was used to display river discharge during the test pumping (01.01.2009-01.10-2009). The contact between the river and groundwater basin is considered good, resulting in an infiltration of river water responsible for c. 70-90 % of the natural recharge to the aquifer (NGU 1986).

River discharge reached a low during test pumping of well D in the period from March to approximately 1-May-09. Discharge was reported 40 m^3/s only. Later in May, the river showed an increased discharge and several floods dominated throughout the summer with values up to c. 250 m^3/s. A new reduction occurred during the autumn months.

The river constitutes the most important source for recharge to the groundwater basin. Contribution to recharge is highly dependent on river discharge, showing an increase as river discharge increases. This is observed as a groundwater level rise in the flood plain (Figure 36).

4.7.2 ARTIFICIAL RECHARGE (INFILTRATION BASIN)

The capacity of the infiltration basin is under normal conditions some 35-37 l/s. Water infiltrating the aquifer is to a great extent withdrawn by the wells. During periods of low river discharge this capacity is reduced, as the low water level limits the capacity of the river intake. During such dry spells, the amount of water infiltrating the aquifer through the basin is suggested from field observation to be only c. 10 l/s. As a consequence, groundwater recharge is reduced through a combination of reduction in recharge from the basin and the river.

4.7.3 RAINFALL RECHARGE (FLOOD PLAIN RECHARGE)

The contribution to recharge by means of infiltration of precipitation is considered most important when river discharge reaches a low. The process of infiltration through the flood plain contributes with c. 4,6 l/s, defined as: Q = A · (Precipitation-Evapotranspiration-Surface runoff). See also part 1.3.4.

This value is an estimate that takes into account the area of the flood plain and net precipitation. In order to delineate the area of interest for flood plain recharge, identification of the approximate recharge area, possible surface waters obstructing infiltration and sediment distribution has been of importance (Figure 41).

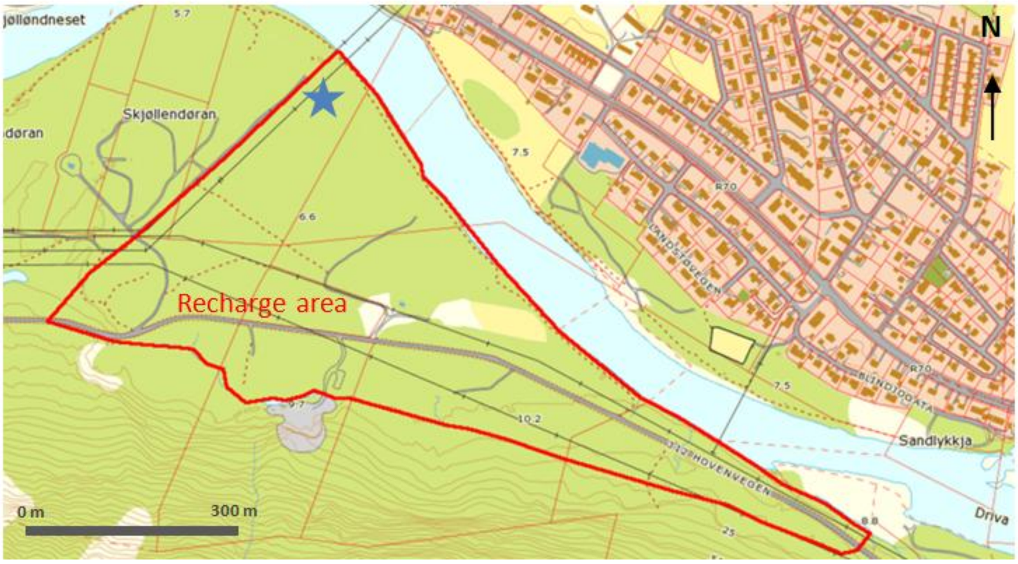

Figure 41: Delineation of the recharge area used for calculation of flood plain recharge.

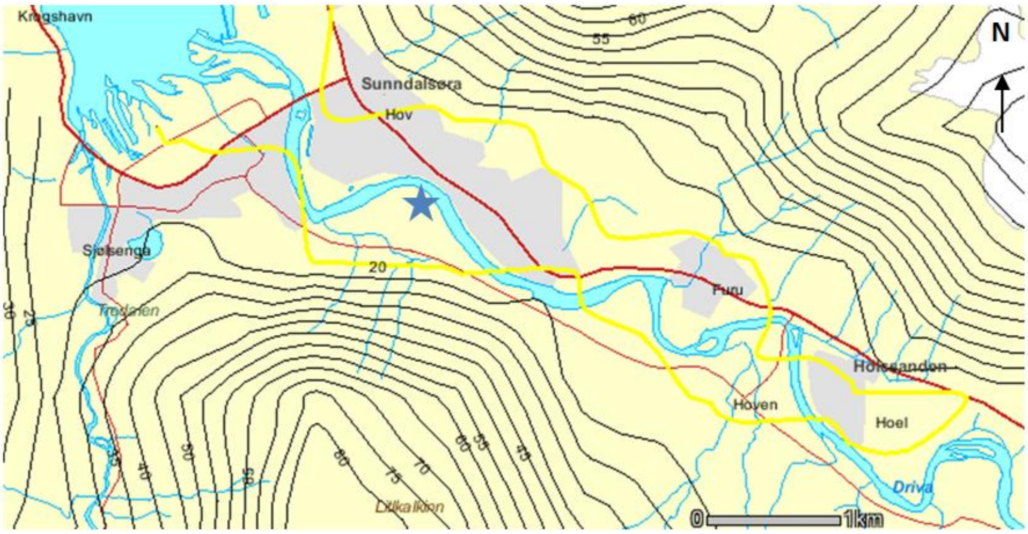

Figure 42: Annual runoff map with isohydrate lines used for the calculation of flood plain recharge (NVE 2010).

The area of recharge amount to an area of c. 0,23 km^2. The annual runoff in this area has been determined using runoff maps produced by NVE (2010), which suggests a runoff of c. 20 l/s/km^2 (Figure 42). Closest *isohydrate* is shown in yellow (20 l/s/km^2).

5. ASSESSMENTS

In order to assess the aims of the assignment, discussions of the following topics are given:

I: Sediment and aquifer properties
II: Well yield
III: Water quality
IV: Remedial/improving actions

5.1 DISCUSSION I: SEDIMENT AND AQUIFER PROPERTIES

5.1.1 SEDIMENTS AND HYDRAULIC PROPERTIES

Grain size cumulative curves confirm that the aquifer sediments were deposited in an environment dominated by fluvial processes. All curves show good sorting and have developed S-shapes accordingly, which reflects that deposition took place in a stream (Driscoll 1986). Georadar profiles and probing have indicated foreset beds of coarser material with fractions dominantly of sand and pebble in the area in vicinity of investigation wells 21, 22 and 23. These soil types were confirmed by subsequent probing (Asplan Viak 2008 b), and is reflected in the grain size analyses of this study.

The less sorted, almost chaotic, appearance of grain size curves of samples from well D compared to the investigation wells can be explained as a result of the different sampling methods that were applied in field (Figure 43):

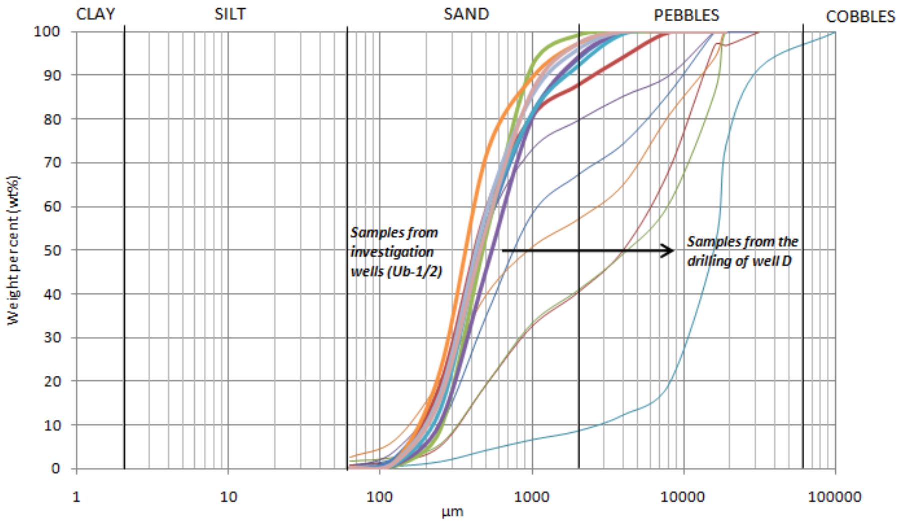

Figure 43: Dependency of sediment sampling method on grain size distribution.

Sediments from the investigation wells were collected with the aid of flushing, which tend to separate finer sediments from coarser fractions. The coarsest sediments are often retained in the aquifer resulting in the grain size distributions seen for Ub-1 and Ub-2 (5/4”-wells), which is skewed towards the finer fractions relative to well D, as can be seen in Figure 43. Well screen dimensions are also significant factors in the separation that occurs upon extraction of sediment. Well screens used in this case reduce the maximum grain size to 3-6 mm. Also, fines might be under estimated as time does not allow for a complete sedimentation before sampling is made.

Extraction of sediments during drilling of well D (ODEX-drilling), on the other hand, was performed with much heavier equipment involving more intense blowing that forced greater fractions upwards and as a consequence increased the degree of mass separation. With this method fine fractions tend to get stuck to the wall of the well and consequently fines are often underestimated from the grain size analyses. Errors of the grain size analyses are so low that the effect on the results in this case is considered to be minimal.

The cumulative curves represent quite well the soil types displayed in the lithological logs of the investigation wells. Lithologic logs are shown in Figure 44 representing the sedimentary distribution at the locations of Ub-1 and Ub-2:

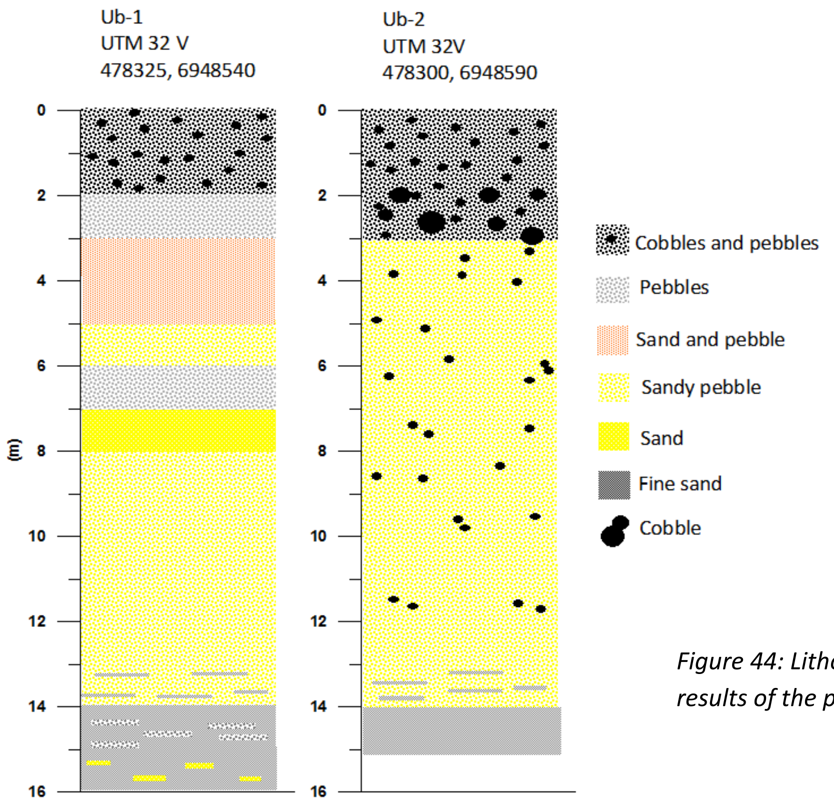

Figure 44: Lithological logs established from results of the probing of Ub-1 and Ub-2.

The lithological logs are based on data from probing conducted by AV as part of the preliminary hydrogeological investigations (Asplan Viak 2008 b). In general, both columns show a coarsening upward trend from fine sand/silt through sand and pebbles, and finally a top layer of some 2-3 meters influenced by coarser strata in the fractions of pebbles and cobbles. The Ub-1 sequence is slightly more stratified compared to Ub-2. It is possible to correlate the fine fractions of fine sand/silt that occur at some 14-16 meters below the terrain surface in both columns. Insight to the conditions in vicinity of well D was shown in e.g. Figure 21. Logs from the probing make the basis for these lithological logs. Probing was conducted for intervals of one meter, and is considered to provide good information on the vertical sedimentary distribution in this case. Georadar profiles shot in close vicinity to the investigation wells during the pre-investigations give additional support for this stratigraphy (Asplan Viak 2008 b). A layer of finer sediment in the fractions of fine sand/silt that occurs at some 14-16 meters below the terrain surface was also displayed in the logs. This layer was detected during early investigations by NGU (NGU 1986), and has later been confirmed with georadar and subsequent probing to occur at varying depths within the deposit (Asplan Viak 2008 b, Asplan Viak 2009, Asplan Viak 2010). Values of hydraulic conductivity from the grain size distribution analyses are in line with this sediment distribution.

Calculated values of hydraulic conductivity correlates well with the aquifer sediment types. The K-values are relatively high, indicating good conditions for groundwater flow (good permeability) in the masses surrounding the well and investigation wells. The values are in line with earlier investigations and studies in the area (e.g. NGU 1986). However, K-values will vary depending on location within the foreset beds. It has to be taken into consideration that values will tend to decrease slightly when groundwater flow is along paths where sedimentary boundaries have to be crossed. In the case of wells situated in foreset beds, horizontal conductivity should be expected to decrease when beds have to be crossed. The conductivity consequently increases when flow is along sediment planes (Driscoll 1986).

5.1.2 ELEMENTAL AND MINERALOGICAL COMPOSITION

The elemental composition of the aquifer sediment reflects a homogenous situation when considering the vertical distribution of elements. There is no clear tendency for any enrichment of elements for any of the analyzed depth intervals. By weight percent the relative abundance of elements are representative of sediments originating from crustal bedrock (Ottesen et.al. 2000). Total weight percent of elements does not add up to 100 % for any of the samples. This negative balance is probably not attributed loss on ignition only, meaning that other factors are causing this discrepancy. These weight loss deviations are likely due to ignition of other components than organic matter that were contained in the sediments. It is presumed this is due to the fact that trace elements were not analyzed, as well as loss of interstitial water (Nesse 2000, Beaudoin 2003).

Silicate minerals dominate the aquifer sediments. Calcite is the only calcerous mineral present, but only in minimal amounts and in discrete depth intervals. The mineral was identified for Ub-1 between 4,5 -7,5 meters and in the lowermost level of Ub-2, reported as piezometers 21 and 22, respectively, in Figure 8. It was totally absent in samples from well D. Of the silicate minerals, Ca-plagioclase is found in greatest abundance, followed by K-feldspar, quartz, amphibole, muscovite and chlorite. The picture is somewhat modified in the case of well D, in that quartz and Ca-plagioclase are found in almost equal amounts, and that there is a greater dominance of muscovite. Deviations between relative mineral percentages of investigation wells and well D might reflect the differences in grain sizes, in that somewhat greater fractions (that were collected in the case of well D) tend to be enriched in more durable minerals like quartz. The two different sediment sampling methods and their individual degree of accuracy concerning grain size distribution were viewed in the previous section 5.1.1. It is also of relevance that different instruments were used for conducting the X-ray analyses (XRD) of samples from the investigation wells contra well D. Another explanation might be that a misinterpretation has taken place. In order to eliminate or confirm the above possibilities of errors, the minerals could have been counted using a microscope.

Chlorite is present in all samples, but is slightly less prominent in well D. The presence of chlorite, as well as the absence of secondary weathering products e.g. kaolinite and gibbsite, suggests there is only a modest silicate weathering taking place in the soil profile.

5.1.3 ORGANIC MATTER

The results of LOI are highly dependent on temperature and length of time allowed for ignition of each sample (Beadouin 2003). LOI was performed with a temperature > 500 °C in this study. Ignitions with temperatures exceeding 440 °C will allow for a mobilization of interstitial water from clays and may lead to a breakdown of carbonates, (Beadouin 2003). As a consequence, the organic content might be even less than suggested with this method. All samples show low values of LOI, indicating these sediments are not a predominant source for organic matter. The highest values were detected for samples at greater depths, suggesting the decomposition may have been less prevalent in the deeper parts of the aquifer. Samples were collected as flush samples from observation wells. These wells are drilled at locations chosen with respect to georadar-profiles aiming at avoiding strata that might result in reduced water quality (Asplan Viak 2008 b), and could be represented by impermeable lenses or layers providing e.g. stagnant groundwater. This constitutes the most essential explanation for low values of LOI. Every sample represents aquifer sediments dominated by sand and gravel fractions. Based on this, and the fact that all samples are discrete point values, this is not where one would expect great amounts of organic matter. Deeper strata of even finer grain fractions, e.g. overbank- and marine sediments, are most probably a more important source. In addition to possible sources at greater depths, the river plain represents a complex fluvial system that implies variation to occur also in a lateral manner. This has been documented by e.g. Asplan Viak (2008 b).

During pre-investigations by AV in 2008 organic matter was registered in flushed samples, and especially from the location of piezometer 23 (Figure 8) where some of the organic material was found as bark fragments. It is likely that a penetration of sediments associated with e.g. abandoned river beds (ox-bow lakes) would have returned increased values of LOI in accordance with the increased amount of fine sediments likely expected to have been deposited in such a low-energetic environment.

5.1.4 GROUNDWATER FLOW AND RECHARGE

GROUNDWATER LEVELS - RESPONSE OF TEST PUMPING AND RIVER DISCHARGE

Figure 36 show that the groundwater level experiences a drawdown in relation to initiation of both aquifer tests (*Stage 2 I and II*) on the 17-Feb-09 and 30-Mar-09, respectively. The figure also indicate how river discharge influence on groundwater level for individual piezometers, in that increased periods of discharge are reflected in an increase in elevation of the groundwater table.

Disruption of the test pumping of well D shortly after start of test pumping is seen as an increase in groundwater level for all wells. Well D was closed down because the water works had problems maintaining the water supply. The closing down coincided with a period of reduced aquifer recharge. Additionally, this event also indicates that a short time is needed before a stabilization of the groundwater level occurs. On the second initiation of test pumping of well D only a slight drawdown is evident from piezometers 22 and 23, but the effect is not that evident this time. The picture could have been improved if the measurement frequency had been increased during the test pumping. Driva shows a reduced discharge from September, and this is reflected in a lowering of the groundwater level.

Registrations of drawdown on the river plain documents that a withdrawal from well D is most evident in the case of piezometer wells 23, 22, 14 and 13. Wells 4, 5, 7 and 10 represent less affected parts of the aquifer that to a greater extent is dominated by recharge of river water during the test pumping. Well 5 correspond best with the level of the river, but wells 4 and 10 also show a good correlation. The closing down of well B in May/June is not that evident.

The development described imply that the water level is more influenced by natural factors by means of river discharge from medio May. The amount of precipitation will also add up to the effect of river discharge. Because of lack of data series in the period July-August it is difficult make any description of fluctuations with river discharge for this part of the test pumping. However, the quick response to disruption of test pumping of well D suggests the groundwater flow is good, and additionally indicate only a few weeks is needed to reach equilibrium.

GROUNDWATER FLOW AND RECHARGE

The making of groundwater level maps from field data is based on mathematical approximations that to some extent are limited to the applied software. The approach used in this study relies on the interpolation of groundwater levels, and does not necessarily reflect the actual situation of groundwater flow one hundred percent. It is still considered to be a very useful tool for interpreting and understanding this aspect of the assignment when viewed in context with existing data on sediment distribution.

The groundwater flow map of **30-Mar-09** documented that groundwater is, as expected, dominantly flowing towards the well area. The prevailing components of flow into the well area derive from the river plane upstream the southwest bank of the river Driva, as well as from the mountains to the south. Infiltration of river water from Driva establishes a flow from the river towards the wells. The distance between lines of equal potential in the groundwater level maps indicate a low hydraulic gradient that increases towards the location of the wells. This is a result of the effect of extraction in the vicinity of the wells that contribute to an increased flow velocity, experienced as a drawdown. The flow upstream relative to the wells parallels to some extent the flow direction of the river. This -flow has a SE to NW-direction. A great amount of this groundwater is taken up by well C, but some is also withdrawn by well D. Well C has its screen oriented in an upstream direction that contributes to this effect. Extraction from wells A and B is evident as a drawdown established next to the river bed. This flow pattern reflects that river water infiltrating the aquifer is drawn towards these wells on the expense of the other wells. This situation is in line with the location of these wells that are closest to the river. Well A is placed at a shallower depth relative to well B, which means that infiltrated water masses from the river to a great extent is utilized by well A. This implies a longer residence time for groundwater from well B relative to well A. Support from this comes from measurements of temperature and EC of all wells, which show that well B has higher values of both parameters indicating a longer residence time (section 5.1.5). Additional water masses flows into these wells from upstream and from adjacent areas downstream.

The groundwater flow map of **31-Mar-09** reveals that the river water infiltrating through the infiltration basin to a great extent is flowing towards well D. As a result of this, and the orientation of the well screen towards the basin, it is reasonable to assume that this well receives most of the water that enters the aquifer through the basin floor (Figure 45):

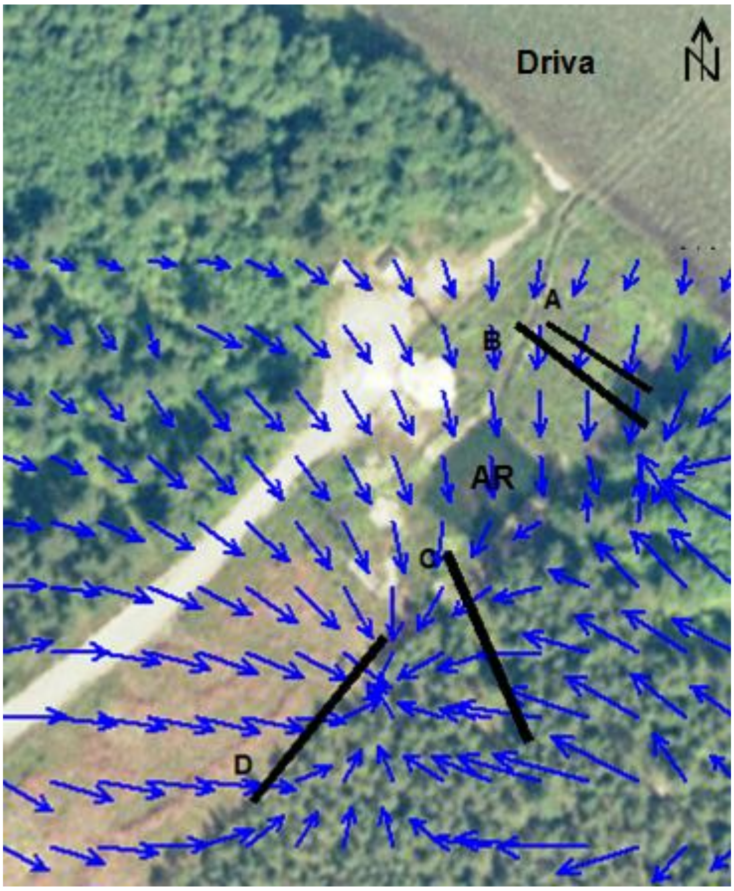

Figure 45: Groundwater flow lines indicate that well D receives water from the infiltration basin.

The observation shown in Figure 45 implies that the status of the infiltration basin as a quality improving construction is still valid, in that it most likely contributes to an increase in the capacity of the groundwater basin. It is reasonable to state that this artificial recharge is an important factor for the good quality groundwater extracted from this well. Based on these assumptions, the infiltration basin has been documented to be of importance to the composition of the groundwater in terms of quality (see section 4.3.7). Groundwater also flows to the well from adjacent areas of the aquifer in accord with lines of equal potential shown in Figures 37 to 40.

Less prominent flow towards well A is seen from the map of **2-Jun-09,** and might be an effect of the closing down of well B in the beginning of June. Another cause might be that there is an increase in river recharge at this time (e.g. Figure 4).

The location of the production wells is to some extent visible as a clear drawdown of the groundwater level. Reasons for less evident locations of wells can be due to several factors, and will be discussed in the following. One of the reasons for this is that water extraction is differentiated between the wells on the basis of production needs again regulated by the waterworks, i.e. by operational- and constructional factors. Specific capacity is also of importance, since this well parameter defines the relationship between extracted water masses and drawdown. A high specific capacity means in practice that greater amounts of water can be extracted for one meter drawdown resulting from groundwater withdrawal. The value for well D is quite low (28 l/s / 3,2 m ≈ 8,75 l/s per meter drawdown), and might explain why the drawdown in relation to extraction during test pumping of this well was greater than for production well C at the same time. For comparison, Asplan Viak (2008 a) reports that well C has a specific capacity of 13 l/s per meter drawdown, while pumping of wells A and B returned a specific capacity of 40 l/s per meter drawdown. Another factor is the location of piezometer wells in the groundwater basin, which are too scattered in the areas in vicinity of wells A and B. This makes it difficult to register any drawdown caused by these wells. Piezometer wells should ideally also have been established closer to the infiltration basin in order to make good observations of this construction, as well as its effects on the groundwater basin. Also, a water level meter should have been installed to register the water level fluctuations in the infiltration basin.

5.1.5 RESIDENCE TIME

Data of residence time presented in Tables 27-30 are based on values being read out from groundwater level maps. This makes the interpretation quite subjective. Still, several aspects can be implied from these results. Water from the infiltration basin towards well D seems to hold the lowest residence time of a little less than two to five days. Well D should consequently be considered vulnerable when it comes to pollution entering via the infiltration basin. Well A is also seemingly vulnerable with respect to its location close to Driva, but residence times indicate a somewhat longer retention time than what is the case with well D.

Additional aspects of residence time arrive from analyses of microbiology. Except from a high level of bacterial counts in the new well, all bacteriological analyses show that the groundwater basin has a good retention capacity with respect to hygienic parameters. The residence time of river water infiltrating towards well A can as a result be considered long enough to a large extent have an immobilizing effect on microbes. Water infiltrating through the infiltration basin will reside for some time in the filter sand and unsaturated zone before entering the aquifer. This time is seemingly long enough in that no detections are made from any of the wells. However, one has to bear in mind that the number of microbiological analyses is quite low, and the result has to be treated accordingly.

Field analyses of temperature and EC measured on the 30-Mar-09 and 8-Sep-09 were displayed in Figures 32 and 33, respectively. These parameters can apply as indicators of residence time. It is evident from the different physico-chemical compositions that different water qualities prevail for different wells. It has to be taken into consideration that well B was shut down in May/June and that the infiltration basin for some time received reduced amounts of river water, which in turn reduced the infiltration capacity of the basin (Figure 14). The value of EC is quite consistent with the results of the water chemistry analyses for both dates (*Appendix III*).

When it comes to temperature, well C holds the highest temperature of all wells, with well B following. Well C is now close to the mean annual air temperature of the study area, which is 6,7 °C (section 1.3.1). Groundwater from well D is slightly warmer than water from well A and the infiltration basin. In this case, water in Driva is warmer than water pumped to the infiltration basin. The values of EC suggest that water from well D is influenced by a mixing of water deriving from the aquifer and the infiltration basin. Well C reflects a situation with a slightly longer residence time, and is in line with the groundwater flow maps showing groundwater deriving mainly from areas upstream. Despite the relatively high EC of well B indicating this well has the longest residence time, well C is considered to have the longest residence time of all the production wells based on temperature measurements and groundwater flow patterns.

With respect to temperature on the 8-Sep-09, well A now produces water with the highest temperature. This data series is reflecting a seasonal heating of groundwater of well A from mixing with warmer river water. Low temperature winter time and high values during summer are an effect of low residence time, not allowing for a stabilization of temperature close to the mean annual temperature (Figure 46):

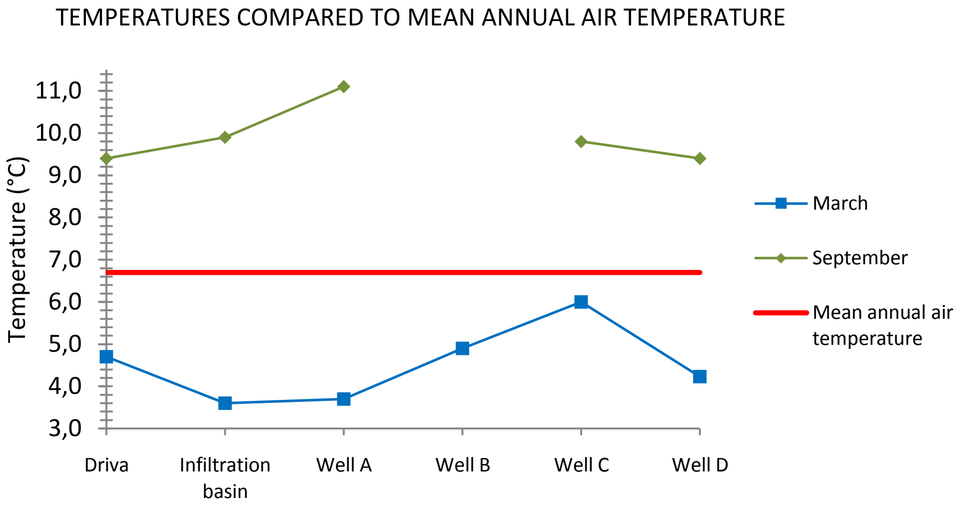

Figure 46: Temperature of all sources with comparison to the mean annual air temperature of the study area.

EC measurements of the 8-Sep-09 from the production wells on the other hand show a trend quite consistent with the first data series presented from 30-Mar-09. However, a difference in this case is linked to well D, which shows an increased value relative to well C. This could be an effect of the change in operational situation in that well B was cut off in the beginning of June, but flow patterns also change in line with natural aquifer recharge (e.g. river discharge).

To summarize the above, measurements of temperature are considered the most representative approach towards an interpretation of residence time. The following relationship between the production wells and residence time is established on the basis of temperature (BOX 8):

BOX 8: INTERPRETATION OF RESIDENCE TIME FROM TEMPERATURE

A → D → B → C → **increasing residence time**

The order of increasing residence time (BOX 8) is in conflict with the calculated residence times initially discussed in this part and presented in section 4.6.1. These values suggest water from well A has a much longer residence relative to water extracted through well D. However, this results from the distances that were applied in the calculations. The residence time for well D was only made with focus on distance from the infiltration basin and the well screen, in order to estimate the travel time of water recharged through the basin before extraction through the well. This distance is much shorter than in the case of well A, hence the discrepancy.

5.2 DISCUSSION II: WELL YIELD

Estimates of well yield of the new production well provide a short-term well capacity applicable during the test pumping period, as well as giving indications for the well yield (*Estimate I*). Suggestions for withdrawal from the well on a more permanent basis are also provided (*Estimate II)*, including aspects of maximum allowed drawdown with respect to the well screen and seasonal variations in groundwater level.

Interpretations of transmissivity make the basis for *Estimate I*. This approach relies on pump curves that were established from a step-drawdown test. The Jacob's method has been applied for the interpretation of transmissivity from pumping test data. Several conditions and assumptions underlie this and other methods used for evaluating pumping test data. These are concerned with e.g. well characteristics and aquifer properties (Krusemann and de Ridder 1990). Both well characteristics and aquifer type conflict with the terms of the Jacob's method in this case. A conflict arises with the aquifer type, in that this aquifer is unconfined while the method used is intended for confined aquifers. Additionally, the orientation of the new well in question is not in accordance with this method taking into consideration the design is that of a tilted well. Another aspect arises when pumping starts from a tilted well; the cone of depression will develop in a manner that deviates from a circular geometry standard for vertical wells. Instead, the cone will tend to develop an elliptic shape. Finally, the well screen only partially covers the aquifer thickness.

However, the method has been used in order to provide an estimate of maximum withdrawal. The value is considered good in that there is only a minor deviation on comparison with the value derived from *Estimate II*. It is also a straightforward method in that it quite efficiently provides a value for maximum well capacity (Q_{max}) and gives suggestion for and applicable well capacity for the test pumping period. The equation for calculating Q_{max} is also based on assumptions that brake down with the prevailing conditions of well design in this case. This approach suggests the maximum capacity for the new production well should be set to **c. 28,1 l/s**. The Jacobs's method was also applied on pumping data corrected for drawdown and time for all three steps of the aquifer test. This attempt of defining Q_{max} returned a value close to the initially suggested value with c. **30,6 l/s**.

The first part of *Estimate II* presents a value of Q_{max} based on observations of drawdown in well D, maximum allowed drawdown with respect to the upper rim of the well screen, as well as assumptions for when steady-state conditions are established in the aquifer. Steady-state conditions were assumed to occur after c. 160 hours and the value of drawdown was defined in line with this. Measurement of drawdown after some 16 hours of pumping with considerations of expected time to equilibrium (when the amount of withdrawn groundwater equals the amount of aquifer recharge) makes the basis here. This approach defines a value of **28,8 l/s**. The second part includes a safety margin, which includes a consideration of influence from seasonal fluctuation in groundwater level on the amount of water withdrawn. The margin amount to a drop in groundwater level of 0,5 meters that is representative of the situation on the 31st of March. This period is dominated by reduced recharge and consequently only allow for a total drop in groundwater level of 2,7 meters. This will represent a conservative value, and returns a value of **24 l/s**.

Both estimates are based on a situation where a certain amount of water infiltrates through the infiltration basin. If the amount of recharge from this source is restricted (or stops completely) Q_{max} of well D will probably be reduced to less than 20 l/s. On the other hand, a situation with maximum infiltration capacity some 30 l/s will probably be possible to extract. A reduction of artificial recharge to 10 l/s amount to reduction in basin capacity of as much as c. 73 % of maximum capacity. A withdrawal of 20 l/s from well D in this case is also likely to cause reduced capacities of well A and C.

All above suggestions for maximum well capacity of well D are conditioned by the withdrawal from production wells A, B and C at all times. A theoretical value of maximum capacity for well D only, is probably c. 32 l/s. This value is based on the specific capacity of the well (some 8 l/s) and the groundwater level equal to a hydrostatic condition (no effect from withdrawal through the other production wells). The corresponding groundwater level would have amounted to 3 meters. The value shows, however, the great dependency of well capacity on the combined effect of withdrawal from all production wells.

*On the basis of an evaluation of all the above described estimates of Q_{max} and reviews of possible scenarios of natural and artificial recharge, it is recommended that the permanent maximum withdrawal from the well is set to **24 l/s**.*

5.3 DISCUSSION III: WATER QUALITY

This section emphasizes discussions of results concerned with aspects of water quality:

- Factors of importance to iron in groundwater:
    - Influence on iron from natural conditions
    - Controls on iron from operational situations
- Review of the general water quality

Results from the characterization of sediment and aquifer properties, as well as well yield discussed in sections 5.1 and 5.2 will also be of relevance here.

5.3.1 FACTORS OF IMPORTANCE TO IRON IN GROUNDWATER

INFLUENCE ON IRON FROM NATURAL CONDITIONS

The **redox-state** of an aquifer to a great extent controls the chemical quality of groundwater (Chapelle et.al. 2009). It is evident from observations during field work and the chemical analyses that redox-species contribute to the water quality of this aquifer, by means of the presences of Fe, Mn, SO_4, NO_3-ions. In the case of iron solubility, pH is also considered a highly conditioning parameter. This groundwater has a quite low **pH-value**, defining the water accordingly as acidic. The combined effect from both low Eh and pH is important in dissolving iron from the solid phase. The **Eh-pH conditions** of the aquifer clearly set the stage for effective iron dissolution.

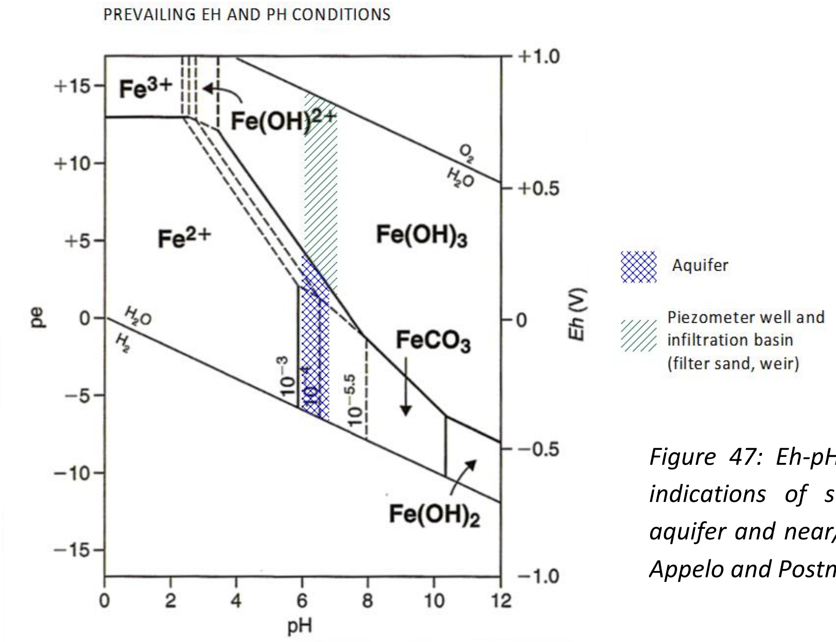
PREVAILING EH AND PH CONDITIONS

Figure 47: Eh-pH diagram with iron species and indications of stability fields prevailing in the aquifer and near/or at the surface (Modified from Appelo and Postma 2005).

The exact location of the groundwater in a stability diagram for iron is difficult to define, but areas of relevance with respect to measured pH-values is shown in blue and green for ferrous and ferric iron in Figure 47 (above), respectively.

In the aquifer, reducing conditions places the groundwater in the blue-area, while rust observed upon introduction to the atmosphere, in e.g. the weir and filter mass of the infiltration basin, represents to a greater extent the green area of ferric iron (Fe-hydroxide).

Banks et.al. (1998) report that the median pH from Quaternary sedimentary aquifers in Norway is some 7,37. The pH measured for this aquifer is approximately one order of magnitude lower, with an average pH for all samples of c. 6,6. This is not that low taking into account the aquifer is comprised by sediments derived from gneissic bedrock with only low amounts of calcerous bedrock, which was also concluded by NGU (NGU 1984 b). The acidic character is important in the context of silicate weathering processes, which depend upon ion-exchange processes, and consequently the dissolution of iron from iron-bearing minerals.

Potential mineral sources of both ferrous and ferric iron with respect to mineral-water interactions are represented by amphibole and chlorite. These are the only silicate minerals that represent possible sources for iron in this case, and are consequently interpreted as the most important sources of iron from the elemental analyses (XRF). Trace amount of sulfides are probably an additional important source for iron (Appelo and Postma 2005). The presence of sulfides might also provide an additional explanation for the low pH of the groundwater (BOX 2). However, the pH-value of the groundwater corresponds with the pH-range of most groundwater of aquifers dominated by gneissic rocks as stated above. pH can be interpreted in terms of the abrasion-pH (Brattli 2009, therein Loughnan 1969). Silicate minerals containing greater amounts of elements in the alkali and earth-alkali group are generally easier to weather and can be identified as the darker silicates, e.g. amphibole and Ca-plagioclase (Appelo and Postma 2005, therein Goldich 1938). Resistant minerals of this deposit are represented by quartz, K-feldspar and muscovite. The imprint of mafic (dark) minerals on pH in this groundwater is low. Relatively high values of more resistant minerals contribute to the explanation of the pH-value of this groundwater.

The most important factor that establishes reducing conditions is regarded as the **decomposition of organic matter** in the aquifer. Organic matter have on several occasions been documented to exist in the fine grained sediments (silt/fine sand), but also in the sand and pebble fractions of the deposit (Asplan Viak 2008 b, 2010). Analyses of LOI in this study detected some organics in the sand and pebble fraction between depths of c. 4,5 -13,5 meters.

Hartog et.al. (2005) stated that organic matter can function as very efficient reductants. The process of decomposition is probably taking place in the entire aquifer, but at some locations to greater extents as a result of greater accumulations and/or more active decomposition. The process of decomposition proceeds by means of consumption of dissolved oxygen. One of the products of this biochemical reaction is the production of carbon dioxide gas (see BOX 3). If concentrations exceed 5,5 mg CO_2/l it is probable that an active decomposition takes place (Aquanor). Accordingly, all estimates of CO_2-concentrations in this study are suggesting this is an active process in the aquifer.

Estimates of CO_2-concentrations vary with time and between wells. The temporal changes seen in the CO_2-concentrations are related to varying inputs of organic matter that are in line with episodes of increased river discharge or flooding (Table 23). It is probable that an increased recharge from river infiltration or infiltration through the river plain would result in an increased entrainment of organic matter, which subsequently is decomposed. This entrainment process contributes as a source for elevated values in that the highest values reported in September (Table 23) follows a peak in discharge of Driva (e.g. Figure 4). It was also seen that values of TOC and DOC are greatest in river water, giving additional support for Driva as an important source of organic matter to the basin (Table 24). When comparing values between all wells, it is evident from these results that decomposition of organic matter seemingly is a more active process in vicinity of well B. Groundwater from this well has suffered an oxygen depletion as a result of the relatively long residence time, and likely also because of the vicinity to a low-permeable layer containing higher concentrations of organic matter. Figure 48 shows production wells A and B to illustrate the importance of screen location and vicinity to low-permeable sediments with organic matter:

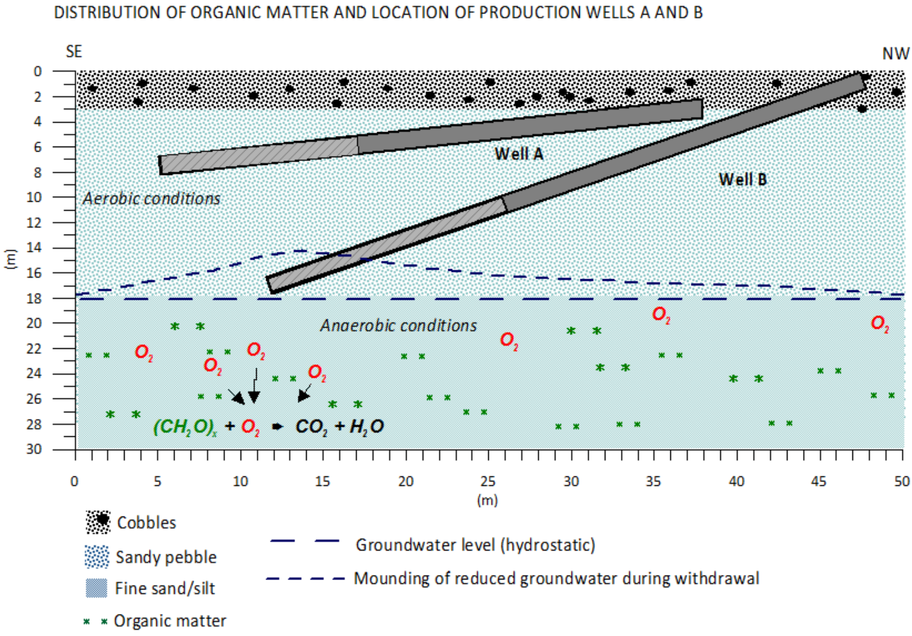

Figure 48: Distribution of organic matter in the aquifer with respect to location of wells A and B.

As indicated above, the redox-state of the groundwater varies with location in the aquifer. The low redox-potential is more evident in the deeper parts of the aquifer, and in vicinity of, or below, more impermeable layers (e.g. Asplan 1997, 2010b).

In order to account for the differences in iron concentrations between all wells, aspects of **aquifer recharge**, **groundwater flow** and **residence times** have to be included. A hydrogeological model of the aquifer is shown in Figure 49. The model illustrates some of the most important processes and aspects involved in groundwater recharge and flow. Flow vectors (arrows) to the north east indicate infiltration from Driva. Locations of wells C, B and A are only indicated as stippled lines in order to illustrate depth to well screens within the aquifer:

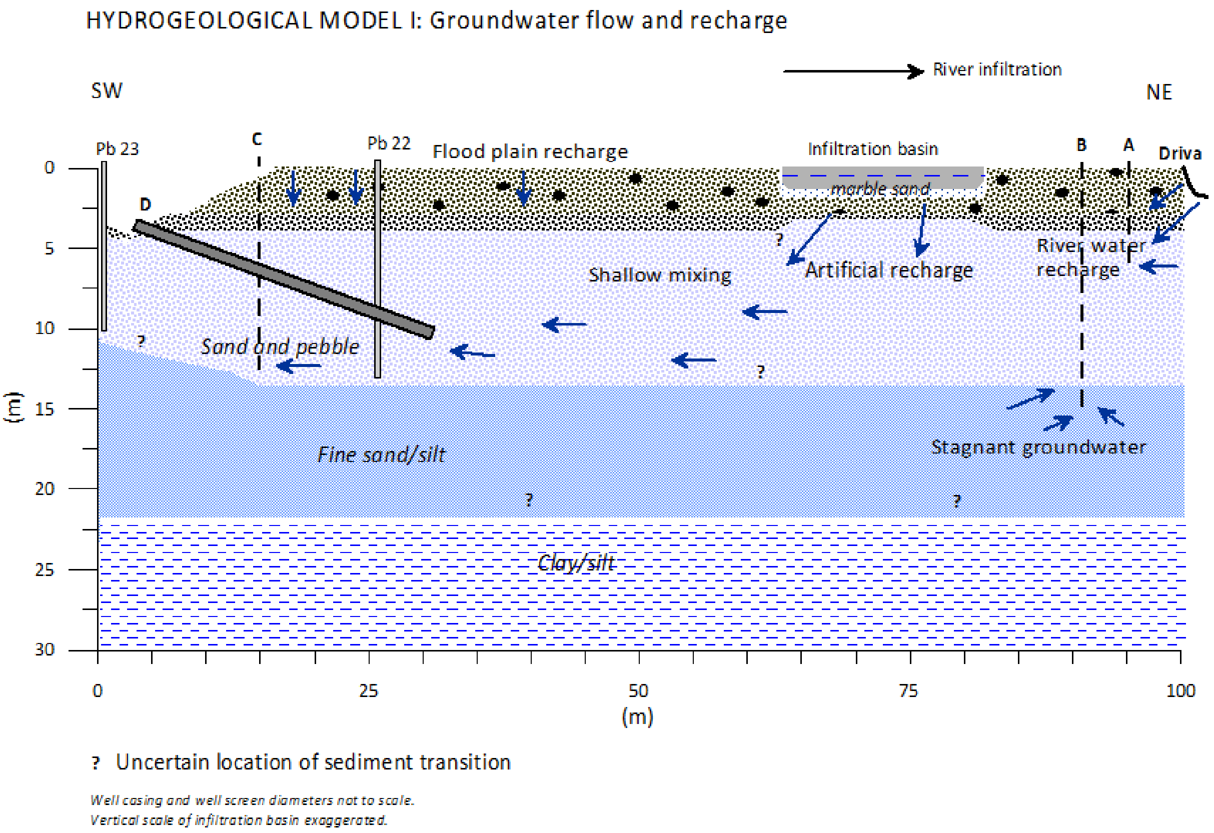

Figure 49: ***Hydrogeological model I.*** *Groundwater flow and recharge in relation to locations of production wells, the infiltration basin and piezometer wells 22 (Ub-2) and 23.*

Figure 49 illustrates which wells that are more dependent on the chemical composition of river water, and consequently river water recharge, contra wells that are more influenced by withdrawal of groundwater with slightly longer residence time (wells C and B). This situation is to a great extent supported by measurements of EC and temperature (section 4.3.4).

Groundwater from well B is ascribed a longer residence time based on depth and vicinity to sediments of lower hydraulic conductivity. This is illustrated in that this well withdraws groundwater from a location in the aquifer where stagnant groundwater resides. Well C on the other hand supplies water that initially infiltrated much further upstream from the works, and for that reason has a long residence time. Well D is much more influenced by water recharged from the infiltration basin as indicated from the groundwater level maps. It is also evident from these maps that most of the groundwater entering the well area from upstream is withdrawn through well C. Consequently, residence time of water from well D is seemingly shorter than groundwater from well C. At least, this seems to be the situation when the infiltration basin contributes to aquifer recharge. Additionally, when considering temperature well C has the longest residence time of all wells supporting this. Well A is placed in a more shallow part in proximity to the river bed, and the water chemistry of this well point towards a greater influence on water chemistry from the infiltrating river water on this well compared to the other wells.

It is not surprising that higher contents of Na and Cl are detected for well B (section 4.3.1), as this is the well with the most deeply situated well screen. The well screen is in addition placed close to a layer of fine (marine) sand and silt, which sets the stage for a withdrawal of deeper and more stagnant groundwater with more saturated pore water. The relatively low hydraulic conductivity of these sediments establishes conditions allowing less efficient flow. The other wells produce groundwater with a character generally more influenced by the water chemistry of the river. This is linked to their improved communication with infiltrated water from Driva and the more shallow situated well screens placed in aquifer sediments of greater hydraulic conductivity than well B. The residence time of groundwater from well B is consequently implied to be quite relevant to explain the water chemistry here.

Additionally, when looking at the balance between Na and Cl-ions of well B (Figure 26), it seems that there has been a depletion of sodium. This could be explained by sorption in the form of ion-exchange between Ca-ions contained in minerals and Na-ions in the aqueous phase (Appelo and Postma 2005). Intrusion of sea water as an explanation is ruled out as concentrations are quite low, and because of the great distance to the delta plain. This is supported by water analyses made by Soldal and Rye (1995), which reports that the transition towards groundwater affected by sea water occurs much further downstream from the river plain.

Samples for chemical analyses collected late August (31-Aug-09) and early September (8-Sep-09) revealed together with the river hydrograph that water quality is dependent on the **infiltration capacity of the river** (Figure 34). However, one has to bear in mind that the low number of samples collected prior to, and subsequent to, the discharge peak only bases on two sampling dates. The sampling frequency could preferably have been increased to reveal a more evident development of water quality throughout the test pumping. Still, EC, concentrations of Ca and Fe have been used as water quality parameters indicative of this a relationship. Infiltration of river water is, as stated earlier, the most important source of aquifer recharge for this groundwater basin. It has been illustrated how the different water quality parameters responded to an increase in discharge in the form of a flooding event. Well A showed a decrease of all three parameters after the discharge peak, which document the relationship between water quality of this well and river discharge. The decrease in these parameters subsequent to flooding results from the mixing of groundwater and river water with low values of TDS and EC. A similar response to discharge is seen for well C, while well D on the other hand experiences an opposite effect for EC.

The evident correlation between decreasing iron concentrations of all wells and increased discharge of Driva, indicated that the concentration of iron to some extent is controlled by the water level of Driva, and hence the infiltration capacity of the river. Well A is presumably the most vulnerable well towards a decline in discharge since this well starts to extract deeper situated groundwater. This was experienced in the case of well B, as reduced aquifer recharge resulted in withdrawal of deeper and consequently more reduced groundwater. The combined effect from groundwater flow, recharge, residence time, **sediment distribution** and **contents of organic matter** is illustrated in Figure 50:

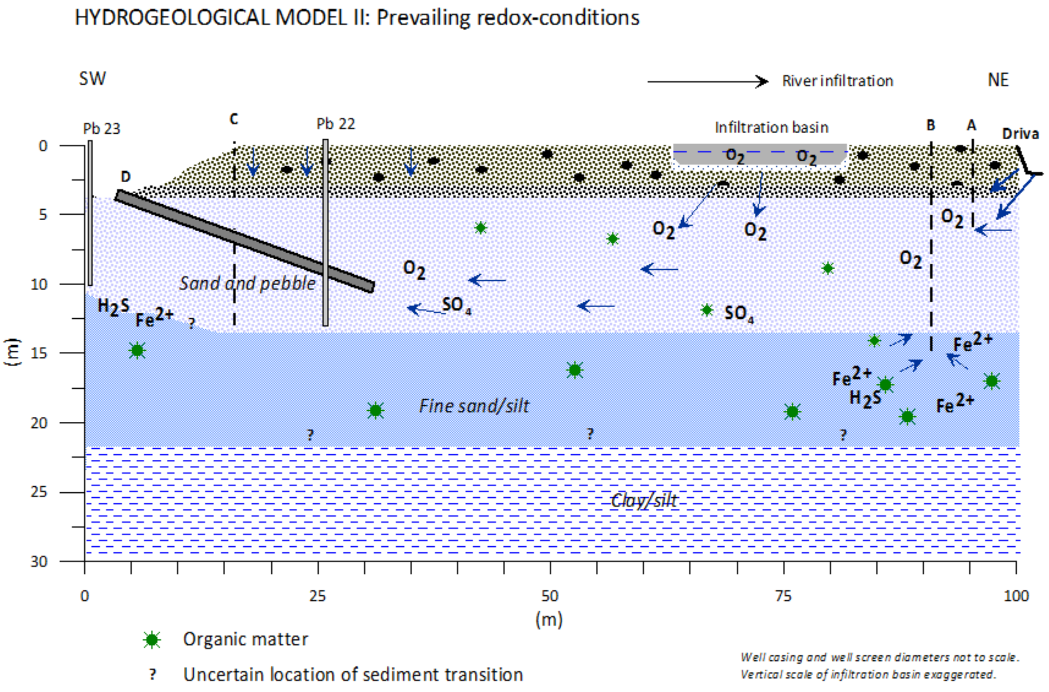

Figure 50: ***Hydrogeological model II.*** *Prevailing redox-conditions (here: oxygen, iron, sulphate species) with respect to location in the aquifer and sediment characteristic.*

Oxygen concentrations usually decrease rapidly from the groundwater table and downwards, and additional depletion derives from the fact that O_2-reduction is favored thermodynamically relative to the reduction of e.g. nitrate and sulphate (Appelo and Postma 2005). The decomposition of organics through the consumption of oxygen is considered the most important biochemical reaction in the aquifer involved in setting the stage for reducing conditions. Consequently, the wells influenced by shallow mixing with more oxygen-rich water have relatively low concentrations of iron. Well B is dominated by more stagnant groundwater, and as a result more depleted in oxygen. In vicinity of investigation well 23, sulphate-reduction is indicted as an ongoing biochemical process with the detection of hydrogen sulfide gas. Sulphate ions in the sand/pebble strata above indicate the effect of more oxygen-rich water on redox-state and hence water chemistry.

Low concentrations of nitrate in well B relative to the other wells might be a result of reduced influence from farmland runoff and/or that a more efficient reduction of this redox-species takes place here. Figure 51 displays the average values of nitrate for all sampled sources:

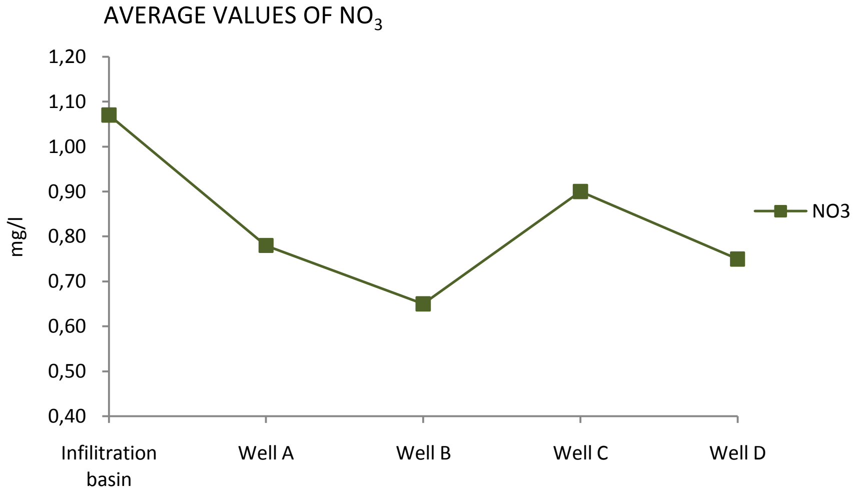

Figure 51: Average values of nitrate of all wells and the infiltration basin suggesting nitrate reduction is prevalent in vicinity of well B.

CONTROLS ON IRON BY OPERATIONAL SITUATIONS

Subsequent to **the closing down of well B** in the beginning of June results of the water analyses of samples from well A of 31-Aug-09 showed an increase in iron values that exceeded the guide value (0,214 mg Fe/l). The most prominent change in water quality from the test pumping period is represented by this particular increase in iron content that was experienced with well A. However, it has to be kept in mind that the number of water samples from this period is low.

Water samples collected by the water works after the closing of well B also reports elevated iron concentrations. This increase is related to changes in the groundwater flow pattern as well B no longer withdraws the deeper, reduced groundwater in this part of the aquifer. Groundwater from the deeper parts of the aquifer is to a much greater extent withdrawn through the overlying well A. A principle drawing is shown in Figure 52:

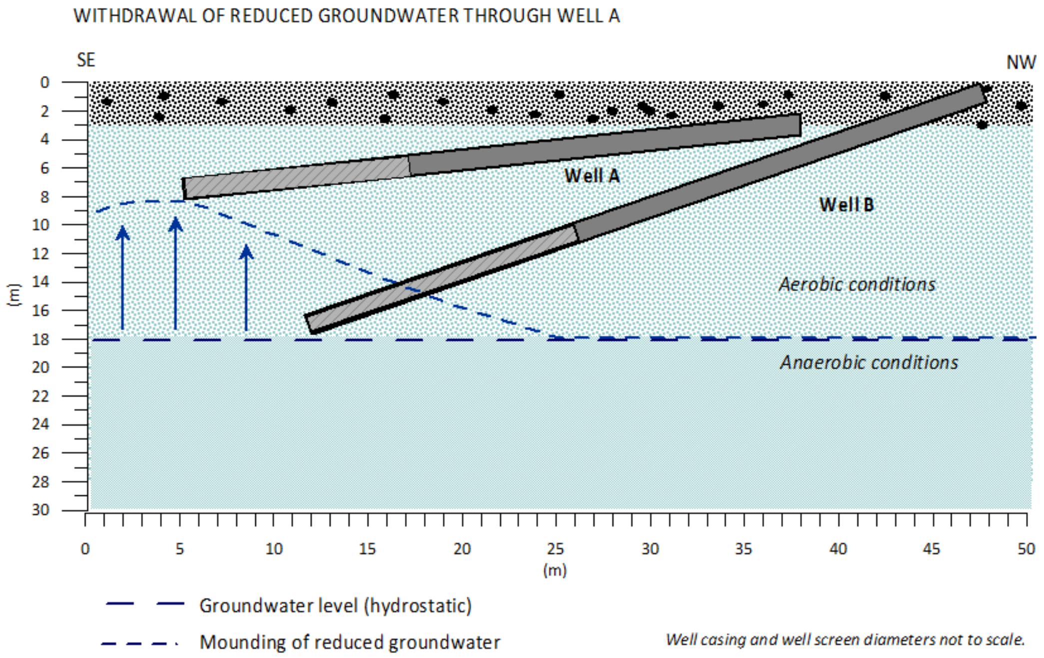

Figure 52: The effect on well A from the closing down of well B is experienced as an enhanced withdrawal of reduced groundwater through the overlying well A, illustrated with a mounding of groundwater from the deeper and more anaerobic parts of the aquifer.

Elevated levels of iron were also measured in water from well C in the same time period. This elevation is not considered to be controlled by the same factors governing the water chemistry of well A, but instead ascribed as a consequence of increased turbidity. Turbidity is evident as an explanation for elevated iron concentrations of water from well C as the value of turbidity represents the highest of all wells from this period (Figure 53):

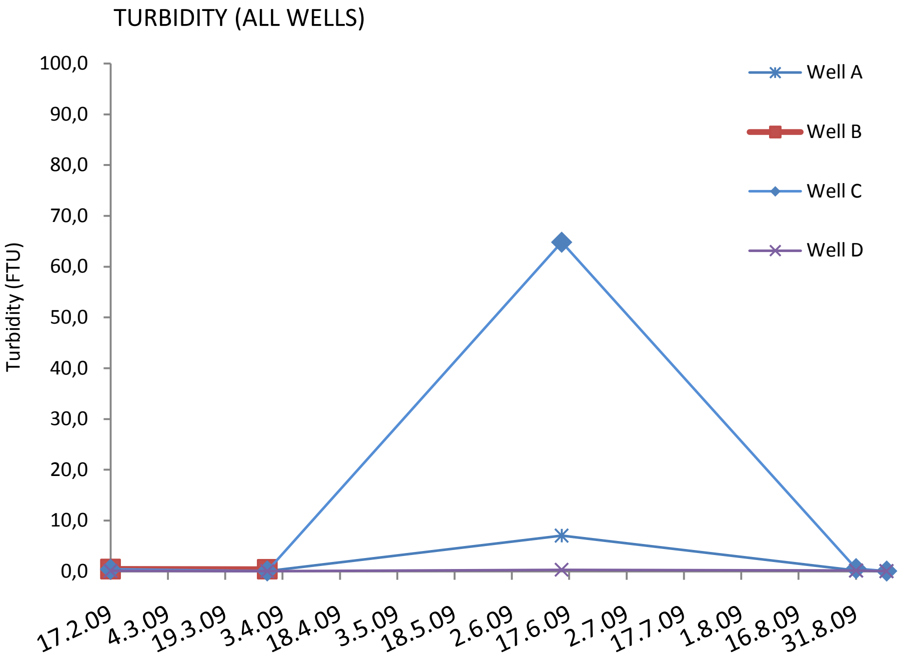

Figure 53: Turbidity in samples from well C increases in a period coinciding with an iron increase in samples from well A, consequently providing the best explanation for elevated values of iron in well C.

The most probable explanation for turbid water is that too little time was allowed between time of sampling and closing of the other wells. This situation results in increased admission velocities that have the capability to tear off rust from the well and transport sand particles through the screen.

Well D produces water of generally good water quality and shows a good development throughout the test pumping in that none of the water quality parameters crosses the guide values (except for the pH-value on the 2-Jun-09). This documentation reveals that well D, and well C, shows very limited signs of the closing of well B with respect to well A.

A quantification of **the effect on water quality from infiltration through the infiltration basin** is somewhat more difficult to assess. The effect on water quality is somewhat difficult to document, because the amount of infiltrated water through the basin is closely conditioned by the river discharge at all times. Periods of decreased infiltration capacity coincides with episodes of reduced discharge, meaning that the hydrological character of the river camouflages any effects of infiltration through the basin floor. This is due to the dependency on water supply from the river intake that is situated in the river. Based on this, there is little documentation is scarce in order to provide a comprehensive assessment of the dependency of reduced artificial recharge on water quality. Still, one should assume that infiltration of water from the infiltration basin functions as a barrier towards extraction of deeper groundwater with high values of dissolved iron. Support arrives from the groundwater flow maps (Figures 37-40), which shows the basin is feeding well D.

5.3.2 GENERAL REVIEW OF THE WATER QUALITY

A general review of the water quality is provided with focus on both physico-chemical and microbiological parameters.

Relatively low concentrations of **major ions** are reflected by the relatively low values of EC and TDS of the groundwater. **Calcium (Ca)** and **magnesium (Mg)** are cations predominantly present as a result of dissolution of dark silicate minerals, in this case plagioclase, amphibole and chlorite. Some of the calcium probably also derive from calcite ($CaCO_3$). The concentrations of these ions are relatively low as a result of the absence of calcerous minerals in the deposit, and are further reflected in the soft character of the water (dH° < 2-3,5). The occurrence of the respective major cations represented by **potassium (K)** and **sodium (Na)** are not that clear to interpret. Both Mg and K usually behave quite conservative (Banwart 1999). K-ions only occur in minor amounts and is most likely attributed the silicate weathering of muscovite, and possibly K-feldspar. K-ions can also be present as a result of runoff of fertilizers from e.g. farmland areas. However, taking into account field measurements of ammonium concentrations and laboratory analyses of **nitrate** and **phosphate,** it seems the best explanation is the weathering of K-bearing minerals. Na-ions are present in small amounts, and are usually derived from salt (NaCl) in relation to sedimentary deposits found below the highest coastline and/or as a constituent of precipitation (Folkehelseintituttet 2008 b). Na-ions, like K-ions, can be introduced by **anthropogenic factors**, e.g. pollution from waste disposals and manure, but with the same argument as for K-ions this is excluded as a relevant explanation. Both the natural origins could be relevant for this element in this case, but when it comes to marine origin this seems to be more relevant in the case of well B. The other wells are probably more controlled by the chemical constituents dissolved in precipitation.

The high residence time of water entering well B is evident from its higher content of Na and Cl, as well as the values of **EC** and **TDS** that represent the highest values of all wells. The differences in these parameters for different samples document a linear relationship between EC and TDS of all wells and the infiltration basin (Figure 54):

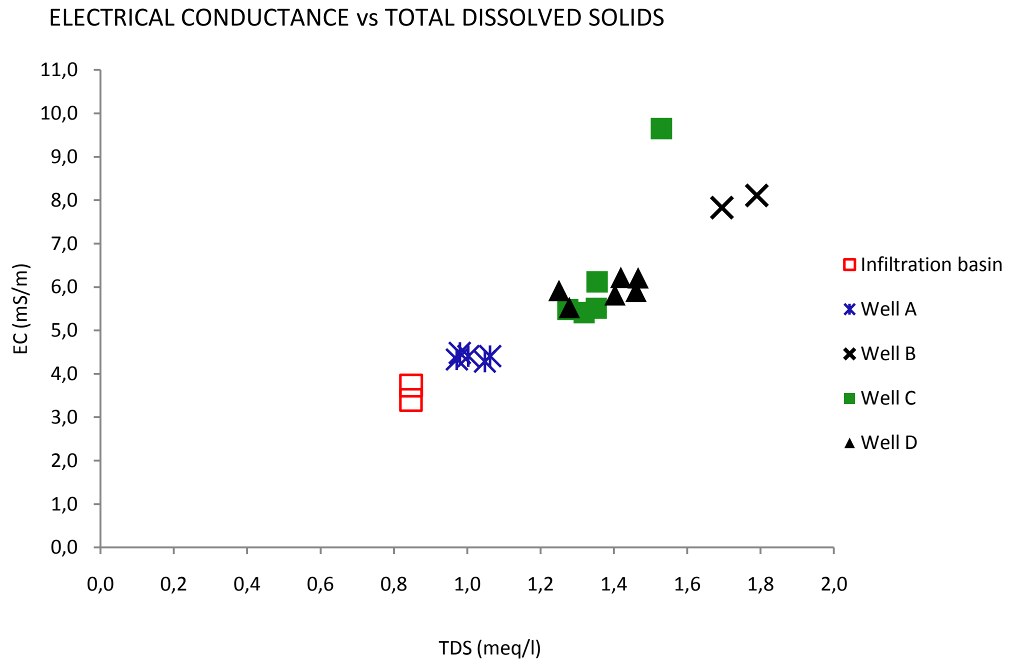

Figure 54: TDS and EC of the infiltration basin and the production wells display a linear relationship.

Lowest values are representative of the infiltration basin, while values increase from well A to wells D and C. Wells C and D display a cluster of data points located between wells A and B. Well B is represented by only two values. Well C shows one value that deviates from the linear trend.

Samples analyzed in this study reports quite low values of EC and TDS. According to Banks et.al. (1998), this is a common aspect for groundwater derived from unconsolidated Quaternary deposits, considering the sediments already have been subject to more intense weathering of the bedrock source and relatively resistant minerals are the once left behind in the deposit. Analyses of water chemistry in advance of this study, e.g. NGU 1984 b, report locally higher values of EC than during the test pumping. This is also the case with pH, which has been measured lower than 5 (e.g. Asplan Viak 2010). Combined, these results are reflecting local variations. Additional aspects pointing towards an immature groundwater characteristic is the absence of plain hydrochemical facies implied by the Piper-diagrams (Figures 30 and 31).

Contents of **bicarbonate** (HCO_3) are interpreted in terms of **total alkalinity** in this case, since the pH is in the range of 6-7 alkalinity is defined as: Alk ~ [HCO_3] (Benjamin 2002). The trend is that acidic water show higher levels of dissolved CO_2 and consequently low values of alkalinity (HCO_3) (Folkehelseinstituttet 2008 b). This makes the groundwater aggressive and further results in a low buffering capacity. In practice, this means that one can expect the groundwater to be vulnerable when it comes to fluctuations in pH upon introduction of acids, e.g. acidic precipitation. Low alkalinity can further be connected to the low content of Ca-ions and consequently lack of calcerous minerals in the aquifer sediments.

Turbidity and **color** are factors indicating the amount of particulate and organic matter in water, respectively. Both parameters are generally kept at a low level during the test pumping. Turbidity and color of river water and the infiltration basin is generally low, reflecting limited influence from erosion and flooding along the river course. This also reflects the time of year the samples were taken. Driva and the infiltration basin were analyzed in advance of a period of increased river discharge. The second sample from the infiltration basin also represents water influenced by a more quiescent period (e.g. Figure 4). Elevated values of turbidity of the well samples reflect in most cases too short time between closing down of pumps (of wells that were not sampled) and subsequent sampling, which is associated with a short period with turbulence as the admission velocity increases. This results in the mobilization and suspension of iron-hydroxide colloids (rust) and possibly also sand particles withdrawn from the sediments outside the well screen. Seemingly, turbidity varies in line with iron concentrations. This was also the conclusion by NGU (NGU 1984 b).

Iron and **manganese** are present in varying concentrations. Manganese often occurs in high concentrations in association with high values of iron (Appelo and Postma 2000, NGU 1991 a). This is a result of the Eh-pH condition of iron also being favorable of Mn-dissolution. From these conditions one would expect manganese to be a problem in addition to iron. The explanation to the low concentrations of manganese in the groundwater is the low contents of this element in the aquifer sediment (Tables 10-11). Fe constitutes some 2,7-3,6 wt. % of the elemental composition, which corresponds to a content approximately three times greater than Mn. This consequently makes iron a more dominating redox-specie, as shown in Figure 55:

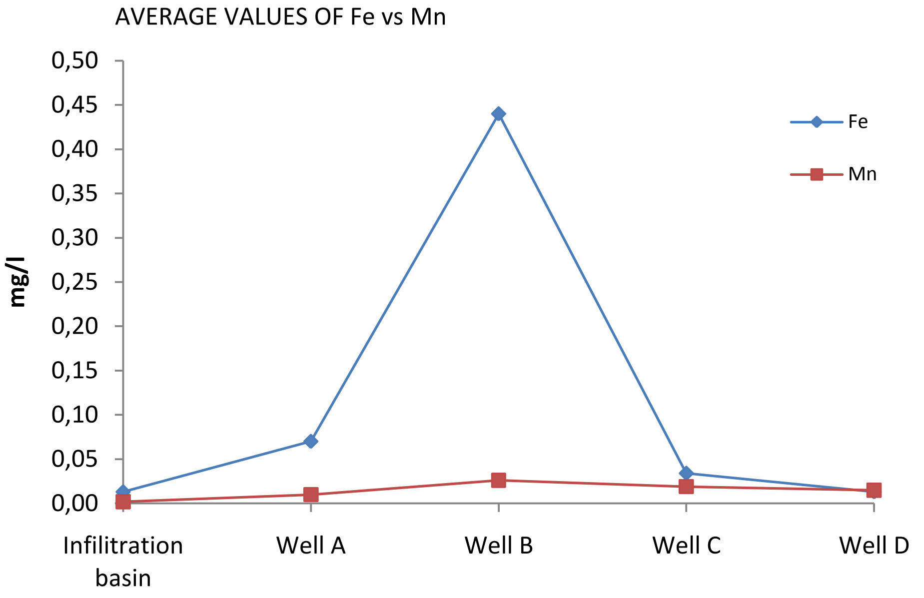

Figure 55: Average values of iron and manganese of samples from the infiltration basin and production wells illustrating the dominance of iron with respect to manganese.

Both elements dominantly derive from the minerals constituting the aquifer sediments (Ottesen et.al. 2000). Field analyses generally show higher concentrations of iron than what results from the laboratory analysis. Some of the differences in concentration might be linked to the different analyzing techniques that were highlighted in section 5.1.2, but also that some iron precipitates before the conduction of laboratory analyses.

Sulphate probably derives from trace amounts of sulfide or sulphate-minerals (e.g. pyrite, gypsum). The anion could also derive from (acidic) precipitation. The slight increase in concentrations of the wells relative to the infiltration basin could be explained by increased residence times. The occurrence of sulfide-minerals cannot be ruled out as a contributing factor in that the sediment samples only represent spot tests. It also has to be taken into account that several minerals might occur in trace amounts and would not have been detected from the analyses. Sulphate is considered a corrosive factor, but is present in too low concentration to cause any substantial problems in this case. The emission of sulphur dioxide from the aluminum works at Sunndalsøra (Hydro Aluminium Sunndal) is also a possible contributor (KLIF). Sulphate can be reduced to hydrogen sulfide (H_2S). This gas has been reported from the test pumping of investigation well 23 (Asplan Viak 2008 b), as well as from a new investigation well (Ub-25) constructed in relation to hydrogeological investigations by AV in February 2010 (Asplan Viak 2010). The latter investigation at the same time revealed artesian conditions in relation to the investigation well, which means layers of low-permeability is present in this parts, which might restrict mixing with oxygen-rich water.

Nitrate is found only in minor concentrations. No **nitrite** was detected above the detection limit during test pumping. Concentrations of other indicators of fertilizers and manure besides nitrate are also low (ammonium, potassium and phosphate), and with that exclude organic pollution and influence from farmland activities as a threat towards the groundwater quality.

Decomposition of organic matter may also cause the reduction of nitrate:

Denitrification involves the oxidation of organic matter by reduction of nitrate to N_2-gas, and is one of the reactions involved in the cycling of nitrogen (Appelo and Postma 2005). There were made no detections of nitrite in this study (below detection limit), which often is used as an indicator for this reaction. Nor ammonium was detected from the field analysis. Even if no immediate forms of denitrification were found, the decomposition of organic matter can be reflected from the concentrations of HCO_3 and pH. The reaction is well documented (Appelo and Postma 2005, therein Bradley et.al. 1992, Bragan et.al. 1997).

$5CH_2O + \mathbf{4NO_3^-} \rightarrow 2N_2 + 4HCO_3^- + H_2CO_3 + 2H_2O$

Fluoride is detected with low concentrations and usually derives from bedrock, or in this case, from the minerals contained in the aquifer sediments. Mineral analyses indicate that fluoride is contained in riebeckite, and dissolution of this mineral is probably an important source for this anion (*Appendix III*). Additionally, fluoride might also be introduced from the emission of fluorine from the aluminum works at Sunndalsøra (Hydro Aluminium Sunndal), which is a by-product of the production of aluminum from bauxite (KLIF).

Heavy metals are present, but in so low concentrations that influence on water quality should be minor. Concentrations of copper (Cu) have been seen to respond to variations in pH, in that concentrations of Cu shows a (faint) increase for reduced pH-values (*Appendix III*). Dissolution of Cu and other heavy metals (e.g. Zn, Pb, Ni) increases under oxidizing and acidic conditions (Brattli 2009). This can be seen for well C, which has the lowest average pH and most often Cu-concentrations exceeding the detection limit. The absence of potential sources of pollution in the area (e.g. mine pit runoff) makes dissolution from sediments the most reasonable source (Ottesen et.al. 2000). The activity at the shooting range (section 1.2) seems to be low with respect to the low lead concentrations detected in this study.

Microbiological parameters of the groundwater are generally good, but more samples should have been collected to draw a more detailed picture of the situation. Taking into account that the data only represents spot tests, greater concentrations should be expected to occur. This is expected to occur in times of flooding when there is a potential for increased runoff from e.g. farmland. A detection of bacterial counts in well D is expected as this normally is the case with newly established wells, and the problem can be solved with disinfection of the well (Asplan Viak 2009).

5.4 DISCUSSION IV: REMEDIAL/IMPROVING ACTIONS AT THE GROUNDWATER WORKS

The discussion below concentrates on different aspects of remedial actions in order to reduce iron contents, and additionally presents other improving actions that might increase the general water quality at the groundwater works.

5.4.1 REMEDIAL ACTIONS WITH RESPECT TO IRON REDUCTION/REMOVAL

ESTABLISHMENT OF WATER SUPPLY FROM WELL D

The new production well produces groundwater of good quality throughout the test pumping. This is reflected from both field and laboratory analyses, including analyses of microbiology. Field analyses conducted as part of *Stage 2 I and II* indicate a stable chemistry over a short time scale (*Appendix III*). Only one value of pH was not approved according to *the drinking water regulations*. Iron values are generally low during the test pumping, and the well should be expected to continue production of groundwater with low iron values (as long as no unexpected problems occur at the works). Neither a reduced infiltration capacity of the infiltration basin, nor the closing down of well B caused any alarming increases in iron concentrations of groundwater extracted from this well. These results reflect that the new well meets all requirements for groundwater quality, as well as well capacity. This also confirms that the establishment of a new production well was an important remedial action in setting the stage for reduced iron contents. An establishment of water supply from well D will be feasible in that it fulfills all initial intentions.

Dependant on what type of pump is chosen to operate well D, one have to consider the allowed withdrawal with respect to the location of the pump relative to the well screen. Calculation of well capacity is based on a drawdown for a situation relevant for suction pumps. If a submersed borehole pump is installed instead, the maximum allowed drawdown, and consequently the well capacity, is reduced as this pump is situated just above the well screen.

IMPROVING THE RIVER INTAKE TO SUSTAIN ARTIFICAL RECHARGE

An improvement of the river intake is necessary to secure a steady recharge to the aquifer from the infiltration basin on an annual basis. It was seen in section 4.5.2 that the infiltration basin is important in feeding the new production well. Despite the lack of direct documentation of how the infiltration basin influences the groundwater quality, it is likely that there is an effect. It has also been reported that periods with reduced river discharge causes a decrease in infiltration capacity of the infiltration basin resulting in elevated values of iron (Personal comment # 1). In order to improve the intake, two solutions are considered feasible and presented in the following:

1. Extension of the existing river intake to the deeper parts of the river bed.
2. Establishment of a tilted well beneath the river bed.

Extension of existing river intake

This action will involve an extension of the existing intake into the deeper parts of the river bed with the aims of stabilizing and maximizing the water supply, which consequently becomes independent of river discharge (seasonal changes). This is an attempt to secure a more stable supply for the infiltration basin, aiming at reaching an effective infiltration that amount to c. 35-37 l/s. This value equals the present capacity of the basin. In order to secure a steady supply, investigations should be conducted to find a location in the river bed where the intake is placed safely and secured from water level fluctuations. To accomplish this, the river bed morphology should be investigated more closely.

The need for this action has to be viewed in light of the longer term effects of actions involving modifications on the infiltration basin (basin 1 and 2), as well as the effects from digging in the river course to increase the river intake capacity to c. 37 l/s. Digging was conducted by the water works during spring 2010 to increase the infiltration basin recharge (Asplan Viak 2010).

Establishment of tilted well beneath the river bed

This is the second of a total of two attempts to secure the supply for a steady artificial recharge. In order to solve the problem with a construction of a new well, it will involve the establishment of a tilted well beneath the river bed. Because of the limited capacity of the groundwater basin, an establishment of a well in the aquifer is ruled out (Asplan Viak 2009). Placing a tilted well with its screen underneath the river bed will most probably not have any reducing effect on the capacity of the other production wells. A disadvantage is the residence time of the water from this well, which is expected to be quite short.

Investigations with the purpose of fulfilling this action were made by AV (Asplan Viak 2010) with the conduction of hydrogeological investigations (GPR, probing and test pumping of investigation wells). Results from these investigations show that conditions are feasible for establishment of a tilted well beneath the river bed.

Establishment of the well and subsequent test pumping will be the next steps in order to establish a fundament for assessing the water quality and the yield of the sediments underlying the river bed. Conditioned by how water quality develops with time, the different areas of applicability for this well might be extended. Good quality water could be used as an additional source at the groundwater works. Connection could be made directly to the water treatment facility or the pump house to substitute well B. A direct connection to the treatment facility was suggested by Asplan Viak (2009). If the quality is not approved with respect to e.g. bacteriological parameters as a result of too low residence time, water should mainly be pumped to the infiltration basin. Another option is to utilize the water from the well mainly with the aim of supplying the infiltration basin, but to during dry spells if water quality is approved.

THE FATE OF WELL B

The test pumping period and the effects from closing down this well during this period have provided important documentation on the functionality of well B. The well produces groundwater with values of iron that exceeds the guide values for this element. This situation has previously been documented by Asplan Viak (2008 a, 2008 b, 2009). This study has documented that the closing down of well B results in an increase in iron concentration of the other production wells. Well A is the most vulnerable well as it has its screen placed above the screen of well B. As a consequence of the closing, well A produced groundwater with clearly elevated levels of iron. Samples of 31-Aug-09 documented iron concentrations that exceeded the guide value. This increase can be correlated with reduced river discharge (Figure 34). It seems that if well B is permanently cut off, the problem might be transferred to a well that functions quite well, and that produces good quality drinking water.

These observations suggest that well B should be cut off from the groundwater works as a production well. Instead, an amount of water corresponding to some 10-15 l/s should be withdrawn, meaning that the operative level is such that the withdrawal equals an amount necessary to avoid extraction of reduced groundwater into the overlying well (A). Well B could preferably be operated with a submersible borehole pump. The exact amount necessary for extraction will vary in line with the aquifer recharge (seasonal variations in groundwater level).

Groundwater from well B could be pumped by means of diversion to Driva or function as an additional water supply for the infiltration basin. If the water is transferred to the infiltration basin, clogging of the filter mass might be a result (Schuh 1990, Helmisaari et.al. 2005). Usually, color and turbidity are responsible for issues of clogging when river water is infiltrated, but this is not considered a relevant problem when dealing with groundwater. Dissolved ferrous iron in the groundwater will on the other hand to an unknown extent precipitate on/in the filter sand (marble sand) as ferric iron in the form of iron-hydroxides (rust). This situation might occur as a result of the change in redox-state upon contact with an oxygen-rich environment:

$$Fe(OH)_2 \rightarrow Fe^{2+} \rightarrow Fe^{3+} \rightarrow \mathbf{Fe(OH)_3}$$

If precipitation is extensive this will demand regeneration of the filter mass on a regular basis and associated operational routines. The demands for regeneration will again depend on the extent of clogging and how this develops with time. In order to prevent too frequent clogging, a second basin (basin 2) should be constructed (section 6.1.4). The advantages of re-infiltrating groundwater relatively rich in CO_2 into basin 2 are the positive effects this will have in increasing both pH and alkalinity as a response to an enhanced dissolution of the marble sand (NGU 1991 b). At the same time, river water infiltrates through basin 1 making regeneration and operation of the basins independent.

The solution involving modifications of the original infiltration basin (basin 1) should preferably be subject to testing in order to observe the functionality of this solution with time prior to establishment of a second basin (basin 2).

INCREASE THE INFILTRATING SURFACE OF THE INFILTRATION BASIN

The infiltration basin was extended in the winter/spring of 2010 amounting to a 50 % areal increase. Further increases in surface area should be viewed in connection with the establishment of a second infiltration basin. This basin could be supplied with water re-infiltrated from well B as mentioned in the previous section, but it considered preferable to do this in a second basin (basin 2). See the previous section.

If further extensions of the basin are needed restrictions are put on the direction to which an extension can take place. An expansion is restricted to an easterly direction in line with locations of adequate sediments with good hydraulic conductivity, the river and the location of a gravelled road NE of well D leading to the pump house. Investigations with GPR detected sediments consisting of too fine fractions in vicinity of the gravelled road to the NE of the well area, but sediments with good hydraulic properties on the other hand were found in the area east of the existing location (Asplan Viak 2008 b). In other words, this operation has to take place upstream relative to the present location of the (extended) infiltration basin.

The effect on iron content is uncertain because of lack of data from the period prior to construction of the basin. There is also lack of evidences for the effect of marble sand on pH (Asplan Viak 2008 a). This study has not provided additional information that documents the effect of the basin on water chemistry, and thus makes it difficult to propose a qualitative outcome of this action (see comments in sections 4.3.7 and 5.3.1). However, an increase in surface area will contribute to an increase in the infiltration capacity of the basin and consequently increase the aquifer recharge to an amount that might represent periods of increased river discharge. This will have an impact on the water chemistry by means of reducing the risk for elevated values of iron during dry spells through the mixing with oxygen-rich water.

Overview map: Remedial actions with respect to reduction of iron contents

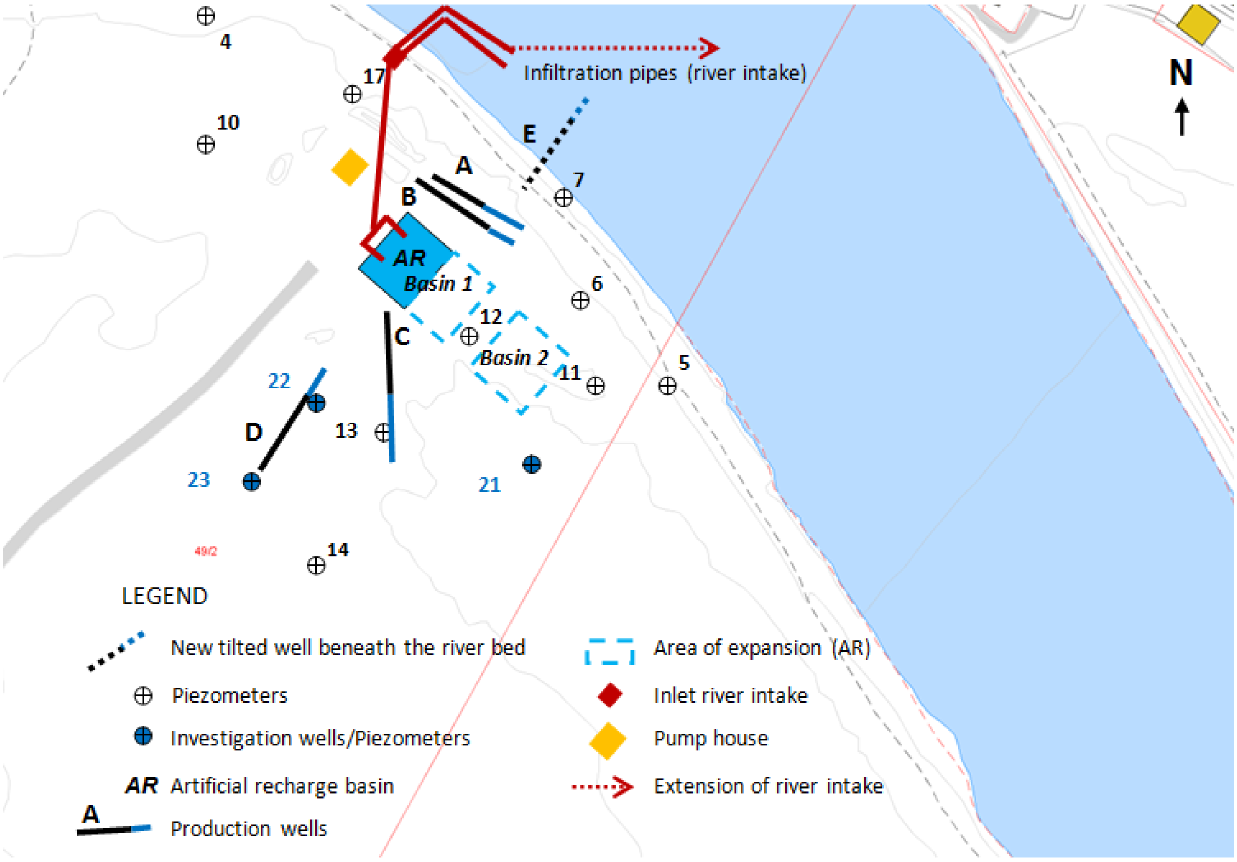

Figure 56: Overview map showing (suggested) remedial actions with respect to reduction of iron.

5.4.2 OTHER IMPROVING ACTIONS AT THE GROUNDWATER WORKS

The different actions described in this section are additional suggestions for improvement of the groundwater works. These actions does not aim directly at reducing the content of iron, but are considered important to implement because they are of great importance for the general water quality at the works.

INCREASED GROUNDWATER PROTECTION

Issues of land use and potential sources of pollution were viewed in the introductory sections, and shows there are no immediate threats towards the water quality (section 1.2). Still, the groundwater basin should be protected to an extent that by all means secures a good water quality in a future perspective. This can be implemented by means of avoiding pollution from present land use and activities in vicinity of the works. In addition, the outcome will be restrictions on future activities in the area that might have a negative effect on water quality in a way that puts pressure on this freshwater resource. In the following, several suggestion and advices will be outlined and presented that aims at increasing the safety at the works, and at securing a good quality raw water with emphasize on physico-chemical and bacteriological aspects.

Groundwater vulnerability

The location of the groundwater works and surrounding areas are regulated through *the Planning and Building Act*, as an area intended for *agricultural areas* and *public outdoor recreation areas* as described introductory in section 1.1. All constructions and facilities of the groundwater works are located on the flood plain on the west side of Driva. The area of interest is dominated by woodland with pine trees, and is mainly used for recreational purposes. The river Driva is used for fishing. Observations of present land use during field investigations and collection of existing information seem to be in line with these demands.

The evaluation of possible sources of pollution is based on present land use and the municipal master plan, which can be viewed in greater detail in *Appendix VIII (CD)*. There has not been registered any considerable threats towards the groundwater quality, but still a few possible sources of pollution have been identified, and are commented in the following:

Fertilizers and salts are contaminants that might affect groundwater quality through **farmland runoff**. Bacteriological and microbiological analyses do not indicate any pollution of the groundwater related to such a runoff. E.coli was detected in samples from Driva, but the source has not been traced in this case. No detections were made in samples of groundwater of the same date. Chemical analyses of the groundwater show no signs of nitrate- or phosphorous contamination. Levels of potassium are also low, and a field analysis of ammonium returned values below the detection limit. Combined, these results show that contamination from farming activities does not represents any harm to the groundwater quality (Folkehelseinstituttet 2008 b). With emphasize to these registrations, pollution from farmland activities has very limited influence on groundwater quality leading up to the conclusion that there is no need for special regulations of existing farming activity. **Pollution from vehicles** passing in vicinity of the water works or parked nearby might cause problems in terms of episodes with e.g. fuel leak. Such episodes are mainly associated with groundwater contamination from a polluting point source (NGU 1991 d). The level of activity at the **shooting range** is seemingly low with respect to concentrations of heavy metals in the groundwater, in this case lead. The form and degree of **recreational activity** (fishing, hiking) in the area is no direct threat to the groundwater quality.

Degree of natural protection

Assessments of the degree of vulnerability of an aquifer involve several aspects. The following will be in focus (after NGU 1992 a):

- Characteristics of the unsaturated zone.
- Lithology, e.g. the occurrence of low permeable clay/silt.
- Aquifer extent and thickness.
- Character of infiltrating surface towards a water course (in this case a river bed).

The **unsaturated zone** is only some 2-3 meters thick and consists of relatively coarse sediments in the fractions of pebbles and cobbles. This is consequently an open aquifer. Water recharged through the flood plain will likely have a short residence time in this relatively thin and permeable zone. Potential contaminants contained in this recharge water will probably not be retained to any great extent in this zone, but instead be transported into the saturated zone. Clearly, the aquifer and the water works are vulnerable when it comes to pollution entering via this zone, and especially if the contaminating source is located close to the well area.

Sediments of the unsaturated zone are associated with hydraulic properties equal to a relatively high **permeability**. There are no confining strata, e.g. clay/silt layers, in close vicinity to the well area that might function as a hydraulic barrier towards contamination entering the groundwater from the unsaturated zone. On the other hand, barriers of fine sand/silt to the W and SE reduces groundwater flow into the well area from these parts (downstream) of the plain. Such strata have also been detected at varying aquifer depths as seen from georadar profiles (Asplan Viak 2008 b), but none of the well screens fully penetrates this strata. Although not representing a confining layer, well B has its screen located in close vicinity to a low-permeable layer at some 14-16 meters depth (Figure 48).

The aquifer has a low **thickness** and limited **extent**. All wells are as a result constructed as tilted wells in order to meet capacity needs. Well screens are as a consequence of this design located at relatively shallow depths. This, in addition to the low thickness of the unsaturated zone, might contribute to a reduction of the natural protection. Flooding events takes place, but the well design of well D reduces the risk for a short circuit during floods, because the well is located slightly higher in the terrain relative to the other wells.

Residence time in the groundwater basin is relatively short as discussed earlier, and much lower than 60 days. The good hydraulic conductivity of the sediments, a good hydraulic contact to the river and a thin unsaturated zone are responsible for this. It was seen that water withdrawn from well A has the most river-dominated signature, but still no detections have been made of any microbiological sort. This suggests that water infiltrating through the river bed and directly into the saturated zone, to a great extent is attenuated in the finer grained sediment fractions here. This was the conclusion also in the report by Asplan Viak (2009), which states that the experience with wells placed in infiltrating groundwater basins to a great extent receives protection from the fine sediments that cover the river bed. This is also mentioned in NGU (1992 a). Estimates suggest that an incident with a fuel leak on the gravelled road NE of well D needs less than a week to reach this well, and that the residence time of water arriving from the infiltration basin would (in a worst case scenario) reach well D in less than two days (section 4.6.1). NGU (1991 b) states that water infiltrating through an infiltration basin ought to have a residence time of at least 50 days, but this is not required since the water is treated at the treatment facility with UV-radiation.

Sediments in the fractions of pebble and cobbles represent fractions less feasible for an efficient retardation compared to the clay fraction (Fetter 2001). The permeability is too high and so the residence time is quite low in order to give an effective retardation (adsorption). Where sand occur in greater amounts, on the other hand, the degree of adsorption will increase as a response to increased surface area and retention time (Fetter 2001). Relatively low contents of organic matter were reported from the analyses of loss on ignition of the sand and pebble fraction. This suggests natural organic matter to a low degree is available for the sorption and immobilization of contaminants, e.g. heavy metals like Cu, Zn and Pb, that potentially might enter the deposit. Still, there might be local zones in the aquifer that comprises greater accumulations of organic matter. The extent of such zones should be linked to the finer grained parts of the deposit as discussed in section 5.3.1. Clay minerals were not detected in the analyses this study, and consequently another sorbent of e.g. heavy metals can be excluded as playing important role for retardation (Fetter 2001).

To summarize, the most probable source of pollution is still considered to be represented by Driva, which serves as the most prominent contributor to recharge for this aquifer. If the river is polluted to a sufficient degree, this will affect the groundwater quality as the river water infiltrates through the river bed. The same scenario is valid if polluted water infiltrates through the infiltration basin.

Area of influence and zoning

In practice, zoning will involve increased protection of the area of recharge and an optimal protection of wells. The definition of groundwater protection zones is an important tool to reduce the risk of negative influence on water quality, and is required by *the drinking water regulation*. By defining these zones and implementing them, one can secure the groundwater basin from potential contamination. In line with this, the rest of this part will deal with how the area of influence and the borders of protection zones 0-3 have been defined. Identifying the area of influence able to say whether the existing borders of this area are still valid, or not. Two maps are provided with suggestions for location of these borders (Figures 57 and 58) and might provide a supplement to the clausal plan of the area. General restrictions towards e.g. land utilization involved for each zone are based on guidelines given in NGU (1992 a), and can be viewed in detail in *Appendix VII (Norwegian)*.

The area of influence is defined as those parts of the aquifer where the groundwater level is affected by the withdrawal from (one or more) pumping wells (NGU 1992 a). The resulting effect on drawdown from the pumping of well D was quantified in section 4.5 as changes in groundwater level on the river plain during the test pumping period (Figure 36). From these registrations, it was found that the greatest change in drawdown was in the direction of piezometer wells 23, 14 and 13. The area of influence is also evident from the groundwater flow patterns (Figures 37-40). On comparison with the previous border of the influential area (NGU 1986), a new border should be extended in an E-SE to S-direction relative to well D.

Zone 0 aims at securing the well area by means of establishing a physical boundary towards others than the operational and/or technical staff at the groundwater works. In practice, this can be achieved by constructing a lockable fence in some 10-30 meters radius from the production wells, as suggested by NGU (1992 a). It is strongly recommended that the infiltration basin is included in this zone. The fence surrounding the basin at present should also be elevated to c. 1,8 meters. Figure 57 displays the suggestion for the area occupied by zone 0, and is estimated to amount to c. 3500 m^2 (35x100 m).

The border of **zone 1** is defined on the basis of where one assume infiltrating water to have a travel time in the saturated zone that equals 60 days for a situation where the groundwater basin is subject to maximum strain (NGU 1992 a). This is in practice a situation where one reaches daily maximum of production and when a maximum withdrawal from the pumps is achieved.

The border is defined with the aid of two methods, but also of importance for drawing these lines is the distribution of sediments and expected hydraulic properties. The first method bases on flow velocities, while the second is based on the cylinder method. More information on how values were computed in each case can be found in *Appendix II*.

In order to calculate **flow velocities**, hydraulic gradients for two different situations were determined. One gradient equals the hydrostatic (original) groundwater level with a gradient of c. 0,4 % and the second with a gradient of c. 8 % representing the level during test pumping. Initial hydraulic gradient is low as one would expect from the flat topography of the river plain. K-values used were found in section 4.1.2 and the value of efficient porosity is defined in accordance with the dominance of sand and pebble of the deposit (0,20). Flow velocity towards the well area during test pumping is estimated c. 18 m/day, while flow under hydrostatic conditions is c. 1,1 m/day. These flow velocities make it possible to suggest where one should locate the influential radii that correspond to a residence time of 60 days. Hence, the zone is placed c. **69 and 97 meters** inwards in direction of wells D and C, respectively. The test pumping indicated a withdrawal from the well area of maximum c. 100 meters (and is the influential radii, R_0, used in section 4.1.1 for calculation of Q_{max}).

Calculation with the aid of the **cylinder method** returns the following:

V_{60days} = 78 l/s x 3600 s x 24 t x 60 days = 404 352 m^3

Defining effective porosity to 0,20 will amount to a volume of saturated sand/pebble of:

404 352/0,20 = 2 021 760 m^3

With an average aquifer thickness of 10 meters the area occupied by zone 1 is:

A = 2 021 760/10 = 202 176 m^2

The area equals a circle with a radius of 253 m.

The cylinder method is, however, not that applicable in this case. The method does not take into account the infiltration that takes place through the infiltration basin and river that have residence times of less than 60 days. On the basis of both methods of calculation, as well as emphasizing the hydrogeological properties of the aquifer (Asplan Viak 2008 b, NGU 1986), zone 1 is drawn in vicinity to the fine grained sediments to the NW at piezometer well 10, and not further south than where one would expect to find finer grained sediments. This border is indicated in Figures 57 and 58:

Figure 57: Suggestion for location of protection zones 0 (well area) and 1 (area of infiltration).

Zone 2 is defined mainly in line with the location of zone 1. The distance to the wells is in the range of 100-300 meters. Since no piezometer wells are placed outside zone 1, it is impossible to exactly decide where withdrawal no longer has an effect on the groundwater table.

The border of **zone 3** is highly dependent on the characteristics of the specific location in question and the distribution of existing boundaries related to real estate/private property. Both aspects are considered and the border is placed accordingly. The border is drawn to the east from a part of the river bank where one would expect infiltration of river water to take place. This is also the area that would be responsible of a runoff into zone 2. The border is further placed along the transition between the mountain foot and unconsolidated sediments in the south. The subsequent location is mainly controlled by established borders leading towards the rim of the river plain to the north. This is also where one would suggest the infiltration of river water is no longer of importance for the water quality.

All descriptions above are based on estimates, and the locations of zones are depending on factors that are outside the scope of this assignment. Still, Figures 57 and 58 can provide a useful base for future decision-making and work with the clausal plan of the area.

Figure 58: Suggestion for location of groundwater protections zones 0-3.

Restrictions in line with suggested zoning

The well area is found in a relatively calm part of the valley that is subject to little pressure from land utilization. Possible sources of pollution are identified as outdoor activities and runoff from farmland. It has to be taken into account that the aquifer is open, meaning there is a direct contact with the groundwater table and the atmosphere via pores of the unsaturated zone. The top layer is comprised of mainly cobbles and pebbles, and thus represents an unsaturated zone with relatively low retention capacity for pollution arriving from the surface. General suggestions for restrictions linked to each zone can be viewed in more detail in *Appendix VII (Norwegian)*. The general rule is that restrictions linked to e.g. zone 2 must comply with the requirements of zone 3. Consequently, the restrictions are stricter as one move progressively closer to the well area. In the following, different concerns for each zone are highlighted:

The only sources of pollution identified inside **zones 1 and 2** are linked to forestry/wood cutting and outdoor activities. Results from the water analyses during the test pumping do not give any indications for an influence from these sources on the groundwater quality. On the other hand, zone 1 could be vulnerable towards point source pollution from e.g. vehicles parked on the gravelled road that leads to the pump house. This suggests that a barrier should be put up on the road that leads up to the water treatment facility and the groundwater works. A signboard should also be put up so that the users can be informed that the river plain comprises a public drinking water supply. It is recommended that one fences in a greater area (35x100) in order to avoid all unnecessary traffic in the well area, which then correlates to zone 0. The fence should be lockable and have a minimum height of 1,8 meters. Margins of the fence already surrounding the infiltration basin can be used. In order to avoid conflict with users of the area, the fence does not have to be extended all the way to the pump house.

ASPECTS OF REGENERATION

Well regeneration

The bacteriological analysis returned a detection of bacterial count in well D that by far exceeded the guide value (Table 26). The well should be disinfected with chlorine before it is connected to the water works. A disinfection of all production wells and well systems should also be considered. An additional contributor to the problems related to iron can then be reduced, as events causing turbidity in wells systems often releases iron encrustations/precipitates.

Filter sand regeneration

Marble sand is used as filter medium in the infiltration basin, and constitutes the basin floor. To ensure a more effective dissolution of calcite, regeneration of the sand by means of refilling and/or replacement should be conducted. It is expected that more calcite (crushed marble/limestone) will dissolve as a response to supply of groundwater from well B that holds greater concentrations of CO_2. This can be expressed with the following equation:

$CaCO_3$ (crushed marble/limestone) + $CO_2 \rightarrow \mathbf{HCO_3} + \mathbf{Ca^{2+}}$

The equation shows the dependency of calcite dissolution on the CO_2-concentration, and consequently the water chemistry of water from well B should be feasible for an efficient dissolution since this groundwater has a relatively high content of CO_2. The result is an increased concentration of HCO_3 and Ca-ions, which increases pH and alkalinity. This was earlier commented in BOX 7. Estimates have suggested that at least an average concentration of 8,9 mg CO_2/l should be expected (Table 19) if raw water from well B is utilized. A marble consumption of approximately 30 g/m^3 is expected for river water, but greater values up to c. 50 g/m^3 should be expected for water added CO_2 (groundwater) according to Asplan Viak (1997).

At present, the groundwater works does not consider clogging of the filter sand to be a problem, in that the filter sand is regenerated with a low frequency (Personal comment # 2). This is a result of a good river water quality, and reflects low amounts of organic matter (e.g. algae), particulate matter and temperature of the water supplying the basin (Helmisaari et.al. 2005, NGU 1991 b). What the effect on the filter sand will be with respect to iron precipitation is not straightforward to predict, but it is likely that an increased formation of iron hydroxide will increase the clogging rate and consequently require more frequent regeneration of the masses (a few times a year, or more often). This is an important argument to construct a second basin where groundwater from well B is infiltrated. With this arrangement operating, the basins can e.g. run by turn instead of closing down during regeneration. Consequently, there will be a continuous artificial recharge.

INVESTIGATE POSSIBILITES FOR A WATER RESERVE

Experiences at the works concerning operational aspects and the test pumping of well D, have made clear that the water supply form Sunndal water works is vulnerable towards dry spells of longer duration (increase in iron concentrations as infiltration capacity of the infiltration basin is reduced). This is especially pronounced late winter before the initiation of snow melting, as the decline in discharge of Driva results in a minimum aquifer recharge. During this time period it is reported that the Kalken water works only contribute with a minimal amount to the water supply. On the other hand, in times of the year when the pressure on the groundwater basin increases, the supply from Kalken reaches its optimal level (Asplan Viak 2008 a, 2009). This situation highlights the need for a new or improved solution that can provide the water works with a more stable water reserve on an annual basis. This would secure the water supply from Sunndal water works.

Several approaches are listed:

- Improve the existing supply from Kalken water works.
- Investigate other locations in Sunndalen relevant for groundwater exploitation (earlier investigations were presented in the introductory part, and distribution of sediments in section 1.4.2).
- Initiate a co-operation with a private water works, e.g. with Nofima in Litldalen (Asplan Viak 2009).

6. CONCLUSIONS

Assessments of results and observations from the test pumping period prove that improving actions should be implemented at the groundwater works. The new production well (well D) meets all requirements in terms of both capacity and quality. A comprehensive review of the water chemistry of all wells, as well as a review of microbiological aspects, reports a generally good quality throughout the test pumping. The only problem is related to high values of iron in production well B. Values have on occasion risen to a little less than 0,5 mg Fe/l.

High values of iron of groundwater from well B results from the extraction of reduced groundwater. The Eh-pH condition of the aquifer in vicinity of well B is favorable of efficient iron dissolution from iron-bearing minerals, e.g. amphibole, chlorite and pyrite. The dissolution process is enhanced in this part of the aquifer as a result of a limited mixing with oxygen-rich water combined with a somewhat longer residence time. Depletion of dissolved oxygen in this part of the aquifer is mainly caused by an enhanced decomposition of organic matter in the sediments. Evidences for active decomposition processes are reported from elevated levels of CO_2, with well B displaying the highest values of all wells. This might reflect a higher concentration of organic matter here and/or conditions more favorable of increased decomposition. Detections of organic matter in analyzed samples of sand and pebble fractions are low, but not insignificant. With respect to sedimentary processes, the finer grained sediments close to well B could be expected to have entrained even greater amounts of organic matter. This well has the deepest situated screen in parts of the aquifer that sets the stage for an effective dissolution and accumulation of dissolved iron. Location of the well and screen close to low-permeable strata with organic matter is also relevant for explaining the elevated concentrations of iron in well B.

The closing down of well B have been documented to have a negative effect on well A by means of increasing iron concentrations that crosses the guide value for this element. Wells C and D were not affected to the same extent by the change in groundwater flow from the closing of well B, reflecting the importance of other factors on the water chemistry of these wells. The distance to well B is relevant, but factors linked to different groundwater flow patterns and the location of wells/well screens in more permeable sediments is also important. Groundwater flow patterns suggest that the infiltration basin is an important source of recharge. Hence, the basin has proven to have some control on the water chemistry and especially with respect to the chemistry of well D. The water chemistry also fluctuate with river discharge, showing a decrease in e.g. iron concentrations of wells A, C and D in periods of increased discharge of Driva.

Key actions at the works in order to reduce iron concentrations are considered to be a modification of the present operation of well B combined with the establishment of water supply from well D. Suggestions for improving actions at the groundwater works is presented below in line with the aims of the assignment and focuses on reducing the iron contents. In addition, comprehensive actions that are considered to secure the water quality are also included. With aspects of the functionality of well D accounted for, actions aiming at improving water quality with respect to reducing iron contents are preferably implemented in the following order:

1. Establish a water supply from the newly established well (D), which then replaces well B as a production well of drinking water.
2. Well B should still be operative such that the withdrawal of groundwater equals an amount necessary to avoid extraction of reduced groundwater into the overlying well (A). Groundwater from well B (c. 10-15 l/s) could be pumped by means of diversion to Driva or function as an additional water supply for the infiltration basin (basin 2). The latter option is feasible in that it will contribute to an increase in pH and alkalinity from re-infiltration of groundwater relatively rich in CO_2.
3. Improve the river intake in order to secure a more predictable water supply on an annual basis:
    i. Establish a new tilted well beneath the river bed. Decisions on areas of use (supply for the infiltration basin (basin 1) and/or additional production well) are dependent on water quality.
    ii. Relocate the already existing river intake into the deeper parts of the river.

4. Further increases in surface area of the basin should be viewed in connection with the establishment of a second infiltration basin (basin 2). This will enhance water supply during periods of dry spells through an increase in aquifer recharge.

Other improving action that should be conducted at the groundwater works:

5. It is strongly recommended that the degree of groundwater protection is increased, to better protect the aquifer from future activities and land use. This action should be accomplished in line with the clausal plan of the area with the definition of four protection zones (0-3), as well as an extension of the protection zone south of well D. The well area (zone 0) must be secured with a lockable fence.
6. Consider possible water supply reserves, as well as an improvement of the Kalken water works to ensure a stable water supply on an annual basis.

7. Reduce the amount of bacterial counts in well D by means of disinfection with e.g. chlorine, prior to operation of the well. Regeneration (disinfection) should also be considered conducted in relation to all wells and transmission lines.
8. Regenerate (refill) the filter mass (crushed limestone/marble) of the infiltration basin in order to increase pH and alkalinity. If feasible, one can replace the present treatment step with addition of $Ca(OH)_2$ at the water treatment facility with the infiltration of CO_2-rich groundwater from well B through infiltration basin 2.

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Additional sources

Cover photo (left): Hydro Aluminium Sunndal
Personal comment # 1: Sunndal municipality represented by Bjørn Røkkum (30-Mar-09).
Personal comment # 2: Sunndal municipality represented by Eilif Lervik (13-Jan-10).

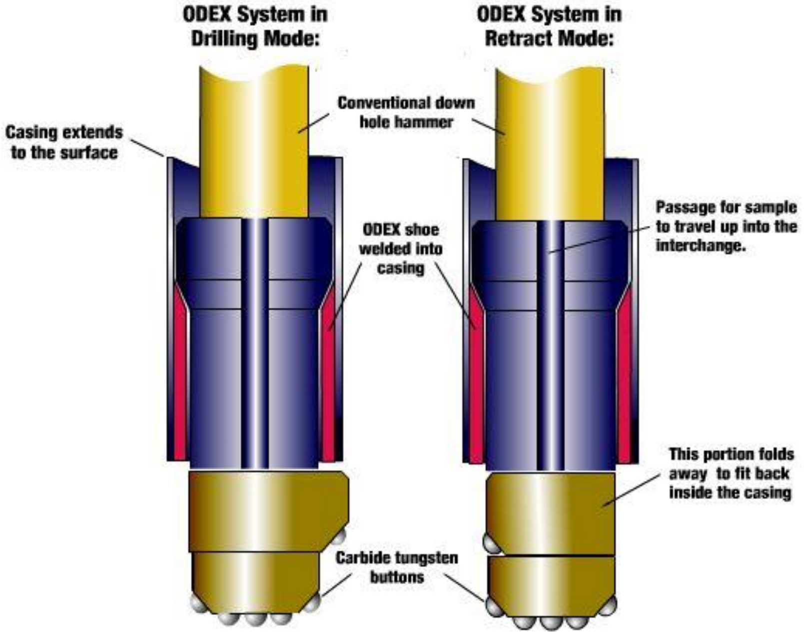
APPENDIX

CONTENTS

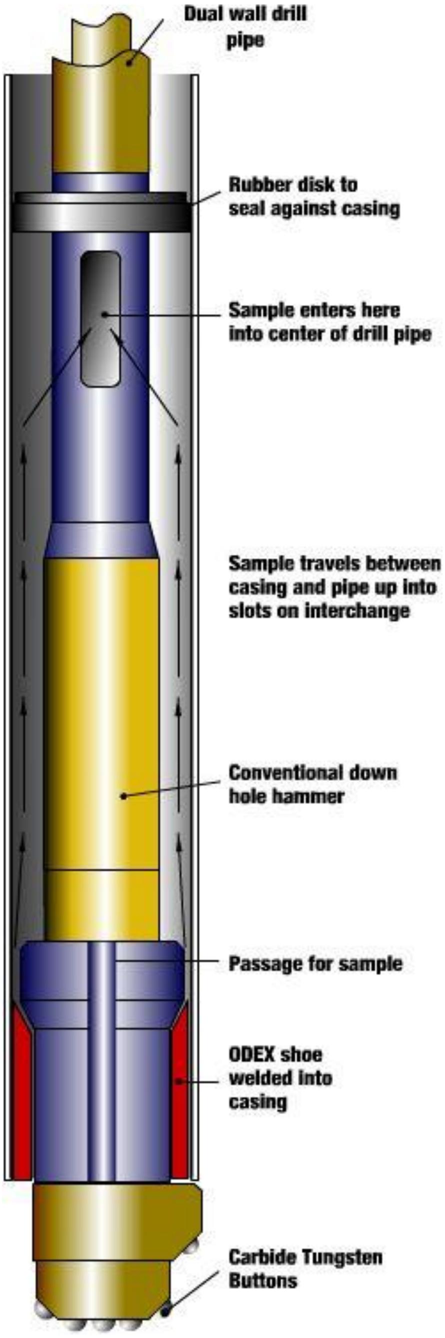
APPENDIX I: ODEX-DRILLING

Figure A (Midnight Sun Drilling) shows how the ODEX-method functions in drilling (left) and retract (right) mode. Figure B display how extraction of sediment samples are conducted and how samples are transported inside casing and drill pipe.

A



B



APPENDIX II: METHODS OF CALCULATION

THE THORNTHWAITE ESTIMATION TECHNIQUE

Davie (2008) gives as outline of the steps to be followed for estimating potential evaporation using the Thornthwaite estimation technique, i.e. obtained from measurements. The technique was originally calibrated for humid temperate regions, and consequently different climates will require alternative calibrations. This empirical model links average air temperature and potential evaporation as follows:

A monthly heat index, *i*, is derived from the mean monthly air temperature, *t* (°C), of a region:

$$i = \left(\frac{t}{5}\right)^{1{,}514}$$

The annual heat index, *I*, is defined as the sum of all monthly heat indexes:

$$I = \sum_{j=1}^{12} i$$

The equation applied for estimating evaporation is based on a number of evaporation measurements:

$$PE = 16\left(\frac{10t}{I}\right)^{a}$$

a: Calibrated as a cubic function from the annual heat index, *I*.

$6{,}7 \times 10^{-7} I^{3} - 7{,}7 \times 10^{-5} I^{2} + 0{,}018\, I + 0{,}49$

HYDRAULIC PARAMETERS DERIVED FROM GRAIN SIZE DISTRIBUTION CURVES

GRADING (CU, UNIFORMITY COEFFICIENT)

Grading, often known as the uniformity coefficient, is defined as the ratio that is 60 % (d_{60}) finer by weight to the grain size that is 10 % finer by weight (d_{10}) (after Fetter 2001). d_{60} and d_{10} are found graphically by reading of the respective values on the grain size distribution curve. The value reports the degree of sorting, bus is also defined in order to estimate the hydraulic conductivity of the aquifer:

$$\mathrm{Cu} = \frac{d_{60}}{d_{10}}$$

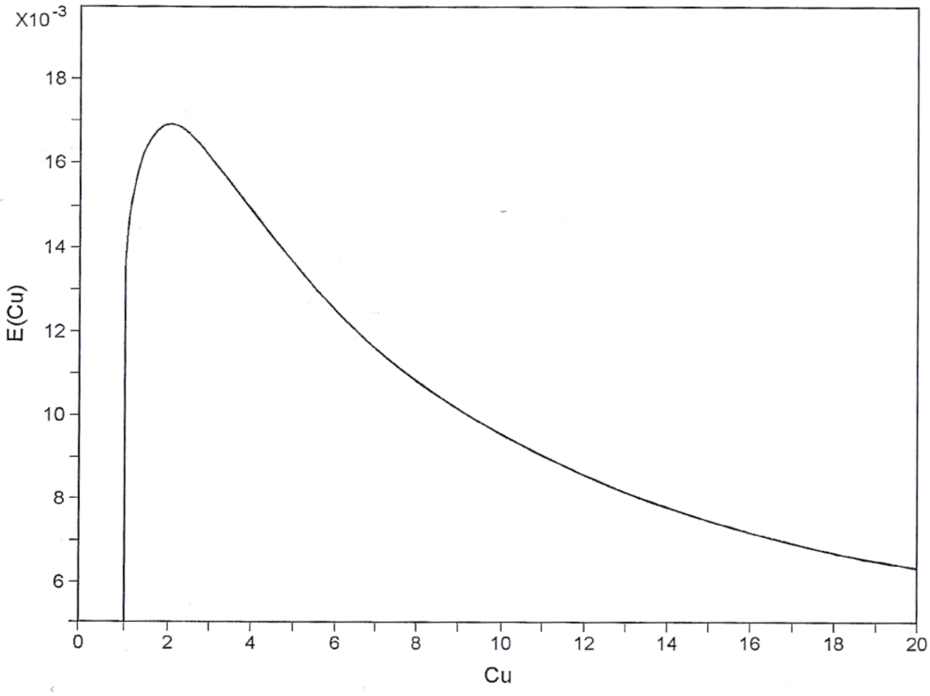
HYDRAULIC CONDUCTIVITY (K)

Two methods have been applied for the calculation of hydraulic conductivity in order to accommodate for different grading ranges. *Hazens method* provides a good estimate for values of $C_u < 5$, while the *Gustafsson formula* is more applicable for values greater than $C_u > 5$. The Gustafssons formula is based on a graphical solution in order to define K. A value of C_u and d_{10} derived from e.g. a grain size analyses forms the basis for this method. A curve displaying the relationship between Cu and the dimension less value of E(Cu) is used. This curve is empirical and has been developed in line with test pumping data and analyses of grains sizes (Brattli 2009, therein Gustafsson 1982).

Hazen method (Cu < 5): $K = 0{,}0116 \cdot (d_{10})^2$

Gustafssons formula (Cu > 5): $K = E(Cu) \cdot (d_{10})^2$

Empirical curve for reading off values of E(Cu) :



TRANSMISSIVITY (T)

A value representative of the aquifer transmissivity can subsequently be found from the value of K taking into account that T is proportional to the product of K and the aquifer thickness (Krusemann and de Ridder 1990).

$$T = K \cdot b$$

HYDRAULIC PARAMETERS FROM STEP-DRAWDOWN TEST

TIME AND DRAWDOWN CORRECTION FACTORS

Field data from *Stage 2 Step-drawdown test II* was used to calculate hydraulic parameters. Drawdown corrections are preformed in order to take into account that this is an open aquifer. After Haug (2003):

$$s' = s_{\mathrm{corr}} = s - \frac{s^2}{2d} = s_{\mathrm{measured}} - \frac{s_{\mathrm{measured}}^2}{2 \cdot 10} = s_{\mathrm{measured}} - \frac{s_{measured}^2}{20}$$

Time correction factors:

$\beta_{t(1)} = 1$

$$\beta_{t(2)} = \Pi_{i=1}^{n-1} \left(\frac{t-t_i}{t-t_i'}\right)^{\frac{Q_i}{Q_n}} = \left(\frac{t}{t-3600}\right)^{0{,}650}$$

$$\beta_{t(3)} = \Pi_{i=1}^{n-1} \left(\frac{t-t_i}{t-t_i'}\right)^{\frac{Q_i}{Q_n}} = \Pi_{i=1}^{2} \left(\frac{t-t_i}{t-t_i'}\right)^{\frac{Q_i}{Q_n}} = \left(\frac{t}{t-3600}\right)^{0{,}517} \cdot \left(\frac{t-3600}{t-7200}\right)^{0{,}794}$$

MAXIMUM WELL CAPACITY

ESTIMATE I: JACOB'S METHOD

Inserting values for discharge and drawdown defines a value of transmissivity (T):

$$T = 0{,}183 \cdot \frac{Q}{\Delta s}$$

Q: Discharge (m^3/s)
Δs: Drawdown with time equal to 1-log unit

Maximum well capacity (after Asplan Viak 2009):

$$Q_{max} = \frac{s_{allowed} \cdot 2\pi \cdot T}{ln \frac{R_0}{r}}$$

T: Transmissivity (m^2/s)
$s_{allowed}$: Maximum allowed drawdown with respect to e.g. top of well screen (m)
R_0: Influential radius (m)
r: Well diameter (mm)

ESTIMATE II: CRITICAL VERSUS OBSERVED DRAWDOWN

This estimate takes advantage of the proportional relationship between maximum capacity and drawdown, and was applied in sections 4.2.4 and 4.2.5. After Asplan Viak (2009):

$$Q_{max} = Q \cdot \frac{\mathrm{s_{allowed}}}{\mathrm{s_{measured}}}$$

RESIDENCE TIME

Residence time estimations have two different purposes in this assignment. First of all to define the groundwater residence time of groundwater extracted through the production wells. Secondly, it was applied to define groundwater protection zones (e.g. defining the influential area due to the additional pumping of well D).

A brief description of the method used:

1. The values of hydraulic conductivity (K) that were used were presented in section 4.1.2. Discrimination between K-values was made with respect to proximity to the respective sampling point (investigation wells in this case).
2. Flow directions were found from studying the groundwater level maps in Figures 38 and 40. Directions were determined from the arrangement of equipotential lines of the map.
3. Hydraulic conductivity (i) was estimated from the change in groundwater level (dh) between equipotential lines along a flow line and the horizontal distance between them (dl). Map scale is defined to M = 1:1880.
4. A value of effective porosity (n_{eff}) was chosen with respect to the dominating soil types (sand and pebble), and accordingly assigned a value of 0,20.
5. Groundwater flow velocity was then found from (after Brattli 2009):

$$v = \frac{K}{n_{eff}} \cdot \frac{dh}{dl}$$

6. Residence time was subsequently defined from the ratio between estimated distance and groundwater flow velocity.

DEFINING GROUNDWATER PROTECTION ZONE 1

Two methods have been applied to define protection zone 1 (the area of influence) at the groundwater works. One of the methods is based on calculations of groundwater flow velocities for two different settings, while the second approach is known as the *cylinder method*.

GROUNDWATER FLOW VELOCITY CALCULATIONS

1. Groundwater level maps representative of both a hydrostatic groundwater level and a groundwater level influenced by pumping makes the basis for this solution. Asplan Viak (1997) can provide a hydrostatic groundwater level map. The second map was displayed in Figure 38.
2. Flow velocities were estimated for both situations using the equation of flow velocity (see previous section: *Residence time).* Distances equal to travel times of 60 days were calculated in the directions toward well C and D, respectively.
3. Distances towards well C and D have been estimated from an imaginary "main point" from the groundwater level map that is supposed to be representative of the situation.

CYLINDER METHOD

The cylinder method can be used to define the distance from a pumping well to the border equal to a residence time of 60 days. In this case, the picture is much more complex in that withdrawal of groundwater is a factor dependent on a combined extraction from several production wells, as well as recharge taking place form the river. An overview of how the method was used in this case:

1. The basis for defining discharge was set equal to the value of maximum production at the groundwater works of c. 280 m^3/h (Asplan Viak 2008 a).
2. Amount of groundwater withdrawn in 60 days:

$$V_{60days\,(water)} = Q_{\max\,prod.} \cdot t_{60days}\ (m^3)$$

3. The total volume of the groundwater basin (saturated sand and pebble) was found as the ratio between volume of withdrawn water (*2*) and effective porosity of 0,20.
4. The corresponding area was found using an aquifer thickness of b = 10 m.
5. Taking into account the area of the model, in this case a circle, the radii was found accordingly:

$$r_{60days} = \sqrt{\frac{A_{60days}}{\pi}}\,(m)$$

CHEMICAL PARAMETERS

TOTAL HARDNESS

Total hardness is calculated from the following relationship (e.g. Fitts (2002) and Hiscock (2008)):

Total hardness = 2,5(Ca^{2+}) + 4,1(Mg^{2+})

Indexes 2,5 and 4,1 are the ratio of $CaCO_3$ formula mass to Ca atomic mass and Mg atomic mass, respectively.

SATURATION INDEX FOR CALCITE (SIC)

Calculations of saturation indexes for calcite ($\log_{10}$ IAP/Q) were made as follows, after e.g. Fitts (2002) and Hiscock (2008):

1. Ionic strength of solution (molality, mol/kg):

$$I = \frac{1}{2} \cdot \sum m_i \cdot z_i^2$$

2. The extended Debye-Hückel equation was applied as this relationship can be used for ions of low to moderately concentrated solutions ($I \leq 0{,}1$):

$$-log\gamma_i = \frac{Az_i^2\sqrt{I}}{1 + a_i B\sqrt{I}}$$

In this case, coefficients are: A = 0,5085, B = 0,3281, a_{HCO3-} = 4 , a_{CO32-} = 4,5 and a_{Ca2+} = 6 (Appelo and Postma 2005).

3. Molal CO_3^{2-}-concentrations were estimated from the equilibrium equation:

$$K_{HCO_3^-} = \frac{[H^+] \cdot [CO_3^{2-}]}{[HCO_3^-]} = 10^{-10{,}33}$$

4. Activities for the relevant species were calculated from the relationship:

$$\alpha = \gamma \cdot m$$

5. Activities of calcium and carbonate were subsequently inserted into the calcite solubility product, and IAP was compared with K_{sp} (or Q):

$$[Ca^{2+}] \cdot [CO_3^{2-}] = K_{sp}$$

IAP/Q < 1 : Undersaturated
IAP/Q = 1 : Equilibrium
IAP/Q > 1 : Saturated. Respective log-values (SIC) are negative, zero and positive.

APPENDIX III: WATER ANALYSES SCHEMES

Guide values listed in the tables contained in this part are defined by *the drinking water regulations*. An ion balance error < ± 5% is indicative of a chemical analysis of good quality and is defined as: *∑(cations-anions)/∑(cations+anions)]·100%*. Concentrations below the detection limit are assigned a value equal to half the detection limit when performing calculations of ion balance error.

Water chemistry I: 17-Feb-09

Location	**Unit**	**Infiltration basin**		**Well A**		**Well B**		**Well C**		**Well D**		
Date		17-Feb-09		17-Feb-09		17-Feb-09		17-Feb-09		17-Feb-09		
Construction		Artificial recharge		Production well		Production well		Production well		Production well		
Analysis nr. (NGU)		2009.0072		2009.0072		2009.0072		2009.0072		2009.0072		
Well dimension	mm			140		168		168		168		
Coordinates		478314.0, 6948607.0		478299.289, 6948640.869		478310.161, 6948637.120		478317.697, 6948633.818		478284.440, 6948563.835		
Physical/chemical		Field	Lab	Field	Lab	Field	Lab	Field	Lab	Field	Lab	Guide value
pH	pH		7,06		6,80		6,66		6,57		6,83	6,5-9,5
Electrical conductance	mS/m	3,88	3,73		4,48		7,83		5,48	5,90	5,92	250
Temperature	°C	0,7								3,8		
Alkalinity	mmol/l	0,18		0,16		0,23		0,17		0,22		
Color	mg Pt /l	<2		<2		<2		<2		<2		20
Turbidity	F.T.U.	< 0.05		0,24		0,47		0,40		0,11		4
Anions												
Fluoride	mg F/l	0,15		0,14		0,12		0,09		0,08		1,5
Chloride	mg Cl/l	2,24		3,26		9,13		3,79		2,85		200
Nitrite	mg NO_2/l	< 0.05		< 0.05		< 0.05		< 0.05		< 0.05		0,16
Bromide	mg Br/l	< 0.1		< 0.1		< 0.1		< 0.1		< 0.1		
Nitrate	mg NO_3/l	1,58		1,04		0,65		0,90		0,66		44
Phosphate	mgPO_4/l	0,39		0,41		0,39		< 0.2		0,47		
Sulphate	mgSO_4/l	3,68		5,11		7,31		8,10		8,97		100
Sum ALK and anions	meq/l	0,32		0,36		0,64		0,44		0,49		
Cations												
Silicon	mg Si /l	1,32		1,78		3,01		2,80		2,01		
Aluminum	mg Al/l	<0.02		<0.02		<0.02		<0.02		<0.02		0,2
Iron	mg Fe/l	<0.002		0,0166		**0,463**		0,0212		0,0095		0,2
Magnesium	mg Mg/l	0,442		0,553		1,06		0,66		0,563		
Calcium	mg Ca/l	3,31		3,54		6,04		4,30		5,73		
Sodium	mg Na/l	1,61		1,86		3,50		2,29		1,84		200
Potassium	mg K/l	0,66		0,81		1,28		1,21		1,13		
Manganese	mg Mn/l	<0.001		0,0019		0,0258		0,0110		0,0051		0,05
Copper	mg Cu/l	<0.005		0,0059		<0.005		0,0261		<0.005		0,1
Zink	mg Zn/l	<0.002		0,0430		0,0387		0,0468		0,0232		0,3
Lead	mg Pb/l	<0.005		<0.005		<0.005		<0.005		<0.005		0,01
Nickel	mg Ni/l	<0.005		<0.005		<0.005		<0.005		<0.005		0,02
Cadmium	mg Cd/l	<0.0005		<0.0005		<0.0005		<0.0005		<0.0005		0,005
Cromium	mg Cr/l	<0.002		<0.002		<0.002		<0.002		<0.002		0,05
Phosphorous	mg P/l	<0.05		<0.05		<0.05		<0.05		<0.05		1,1
Sum cations	meq/l	0,29		0,32		0,59		0,40		0,44		
Ion balance error	**%**	**-5,1**		**-5,0**		**-4,0**		**-5,4**		**-4,9**		

Water chemistry II: 30-Mar-09

Location	**Units**	**Well A**		**Well B**		**Well C**		**Well D**		
Date		30-Mar-09		30-Mar-09		30-Mar-09		30-Mar-09		
Construction		Production well		Production well		Production well		Production well		
Analysis nr. (NGU)		2009.0129		2009.0129		2009.0129		2009.0129		
Well dimension	mm	140		168		168		168		
Coordinates		478299.289, 6948640.869		478310.161, 6948637.120		478317.697, 6948633.818		478284.440, 6948563.835		
Physical/chemical		Field	Lab	Field	Lab	Field	Lab	Field	Lab	Guide value
pH	pH	6,70	6,73	6,48	6,59	6,10	**6,40**	6,80	6,57	6,5-9,5
Electrical conductance	mS/m	4,20	4,33	8,00	8,11	6,10	9,65	5,81	5,82	250
Temperature	°C	3,7		4,9		6,0		4,23		
Alkalinity	mmol/l	0,16		0,21		0,17		0,18		
Color	mg Pt /l	2,60		16,20		<2		<2		20
Turbidity	F.T.U.	< 0.05		0,42		0,07		< 0.05		4
Anions										
Fluoride	mg F/l	< 0.05		< 0.05		< 0.05		< 0.05		1,5
Chloride	mg Cl/l	3,11		9,24		5,48		3,27		200
Nitrite	mg NO_2/l	< 0.05		< 0.05		< 0.05		< 0.05		0,16
Bromide	mg Br/l	< 0.1		< 0.1		< 0.1		< 0.1		
Nitrate	mg NO_3/l	0,97		0,65		0,61		0,58		44
Phosphate	mg PO_4/l	< 0.2		< 0.2		< 0.2		< 0.2		
Sulphate	mg SO_4/l	4,74		8,17		8,18		9,32		100
Sum ALK and anions	meq/l	0,35		0,64		0,50		0,46		
Cations										
Silicon	mg Si /l	1,76		3,28		3,59		3,23		
Aluminum	mg Al/l	<0.02		<0.02		<0.02		<0.02		0,2
Iron	mg Fe/l	<0.002		**0,416**		0,0188		0,0042		0,2
Magnesium	mg Mg/l	0,544		1,13		0,806		0,622		
Calcium	mg Ca/l	3,81		6,91		5,08		5,36		
Sodium	mg Na/l	2,11		3,86		3,32		2,43		200
Potassium	mg K/l	0,86		1,54		1,43		1,25		
Manganese	mg Mn/l	<0.001		0,0262		0,0357		0,0134		0,05
Copper	mg Cu/l	<0.005		0,0057		0,0074		<0.005		0,1
Zink	mg Zn/l	0,0319		0,0352		0,0362		0,0280		0,3
Lead	mg Pb/l	<0.005		<0.005		<0.005		<0.005		0,01
Nickel	mg Ni/l	<0.005		<0.005		<0.005		<0.005		0,02
Cadmium	mg Cd/l	<0.0005		<0.0005		<0.0005		<0.0005		0,005
Cromium	mg Cr/l	<0.002		<0.002		<0.002		<0.002		0,05
Phosphorous	mg P/l	<0.05		<0.05		<0.05		<0.05		1,1
Sum cations	meq/l	0,35		0,66		0,50		0,46		
Ion balance error	**%**	**0,3**		**-1,6**		**-0,7**		**-1,1**		

Water chemistry III: 02-Jun-09 and 15-Jun-09

Location	**Unit**	**Well A**		**Well C**		**Well D**		**Well D**		
Date		15-Jun-09		15-Jun-09		02-Jun-09		15-Jun-09		
Construction		Production well		Production well		Production well		production		
Analysis nr. (NGU)		2009.0211		2009.0211		2009.0211		2009.0211		
Well dimension	mm	140		168		168		168		
Coordinates		478299.289, 6948640.869		478317.697, 6948633.818		478284.440, 6948563.835		478284.440, 6948563.835		
Physical/chemical		Field	Lab	Field	Lab	Field	Lab	Field	Lab	Guide value
pH	pH		6,87		6,44		**6,39**		6,58	6,5-9,5
Electrical conductance	mS/m		4,41		6,12		6,21		6,22	250
Temperature	°C									
Alkalinity	mmol/l	0,16		0,12		0,16		0,16		
Color	mg Pt /l	2,74		<2		<2		<2		20
Turbidity	F.T.U.	**7,06**		**64,8**		0,17		0,33		4
Anions										
Fluoride	mg F/l	<0.05		0,11		<0.05		<0.05		1,5
Chloride	mg Cl/l	2,39		5,13		4,33		3,98		200
Nitrite	mg NO2/l	< 0.05		< 0.05		< 0.05		< 0.05		0,16
Bromide	mg Br/l	< 0.1		< 0.1		< 0.1		< 0.1		
Nitrate	mg NO3/l	0,76		0,76		1,00		0,99		44
Phosphate	mg PO4/l	< 0.2		< 0.2		< 0.2		< 0.2		
Sulphate	mg SO4/l	4,68		8,34		10,20		8,47		100
Sum ALK and anions	meq/l	0,33		0,44		0,49		0,45		
Cations										
Silicon	mg Si /l	2,10		2,93		3,17		3,20		
Aluminum	mg Al/l	<0.02		<0.02		0,022		<0.02		0,2
Iron	mg Fe/l	0,0052		0,0041		0,0028		<0.002		0,2
Magnesium	mg Mg/l	0,611		0,838		0,768		0,768		
Calcium	mg Ca/l	3,90		4,70		5,57		5,50		
Sodium	mg Na/l	2,12		3,18		2,77		2,76		200
Potassium	mg K/l	0,78		1,16		1,18		1,18		
Manganese	mg Mn/l	0,0026		0,0200		0,0204		0,0223		0,05
Copper	mg Cu/l	<0.005		<0.005		<0.005		<0.005		0,1
Zink	mg Zn/l	<0.002		0,0029		0,0123		0,0142		0,3
Lead	mg Pb/l	<0.005		<0.005		<0.005		<0.005		0,01
Nickel	mg Ni/l	<0.005		<0.005		<0.005		<0.005		0,02
Cadmium	mg Cd/l	<0.0005		<0.0005		<0.0005		<0.0005		0,005
Cromium	mg Cr/l	<0.002		<0.002		<0.002		<0.002		0,05
Phosphorous	mg P/l	<0.05		<0.05		<0.05		<0.05		1,1
Sum cations	meq/l	0,36		0,47		0,49		0,49		
Ion balance error	**%**	**4,7**		**3,7**		**4,2**		**-0,3**		

Water chemistry IV: 31-Aug-09

Location	**Unit**	**Well A**		**Well C**		**Well D**		
Date		31.08.2009		31.08.2009		31.08.2009		
Construction		Production well		Production well		Production well		
Analysis nr. (NGU)		2009.0289		2009.0289		2009.0289		
Well dimension (mm)		140		168		168		
Coordinates		478299.289, 6948640.869		478317.697, 6948633.818		478284.440, 6948563.835		
Physical/chemical		Field	Lab	Field	Lab	Field	Lab	Guide value
pH	pH		6,92		6,62		6,92	6,5-9,5
Electrical conductance	mS/m		4,41		5,51		5,53	250
Temperature	°C							
Alkalinity	mmol/l	0,20		0,15		0,23		
Color	mg Pt /l	5,38		9,89		<2		20
Turbidity	F.T.U.	0,15		0,53		0,18		4
Anions								
Fluoride	mg F/l	0,07		0,13		0,07		1,5
Chloride	mg Cl/l	2,35		4,03		1,35		200
Nitrite	mg NO2/l	< 0.05		< 0.05		< 0.05		0,16
Bromide	mg Br/l	< 0.1		< 0.1		< 0.1		
Nitrate	mg NO3/l	0,48		0,66		0,65		44
Phosphate	mg PO4/l	< 0.2		< 0.2		0,28		
Sulphate	mg SO4/l	4,61		8,20		8,03		100
Sum ALK and anions	meq/l	0,37		0,43		0,43		
Cations								
Silicon	mg Si /l	2,03		3,00		2,38		
Aluminum	mg Al/l	<0.02		0,036		0,056		0,2
Iron	mg Fe/l	**0,214**		0,107		0,0414		0,2
Magnesium	mg Mg/l	0,596		0,795		0,577		
Calcium	mg Ca/l	4,55		4,68		6,02		
Sodium	mg Na/l	1,88		2,86		2,09		200
Potassium	mg K/l	1,07		1,41		1,51		
Manganese	mg Mn/l	0,0101		0,0146		0,0117		0,05
Copper	mg Cu/l	<0.005		0,0066		<0.005		0,1
Zink	mg Zn/l	0,0042		0,0045		0,0707		0,3
Lead	mg Pb/l	<0.005		<0.005		<0.005		0,01
Nickel	mg Ni/l	<0.005		<0.005		<0.005		0,02
Cadmium	mg Cd/l	<0.0005		<0.0005		<0.0005		0,005
Cromium	mg Cr/l	<0.002		<0.002		<0.002		0,05
Phosphorous	mg P/l	<0.05		<0.05		<0.05		1,1
Sum cations	meq/l	0,39		0,46		0,48		
Ion balance error	**%**	**4,1**		**3,2**		**4,7**		

Water chemistry V: 08-Sep-09

Location	**Unit**	**Infiltration basin**		**Well A**		**Well C**		**Well D**		
Date		08-Sep-09		08-Sep-09		08-Sep-09		08-Sep-09		
Construction		Artificial recharge		Production well		Production well		Production well		
Analysis nr. (NGU)		2009.0305		2009.0305		2009.0305		2009.0305		
Well dimension	mm			140		168		168		
Coordinates		478314.0, 6948607.0		478299.289, 6948640.869		478317.697, 6948633.818		478284.440, 6948563.835		
Physical/chemical		Field	Lab	Field	Lab	Field	Lab	Field	Lab	Guide value
pH	pH	6,50	6,79	6,41	6,66	6,10	6,32	6,24	6,52	6,5-9,5
Electrical conductance	mS/m	3,70	3,40	4,50	4,28	5,30	5,41	5,80	5,90	250
Temperature	°C	9,9		11,1		9,8		9,4		
Alkalinity	mmol/l	0,18		0,20		0,15		0,18		
Color	mg Pt /l	3,48		3,81		<2		<2		20
Turbidity	F.T.U.	<0.05		<0.05		<0.05		<0.05		4
Anions										
Fluoride	mg F/l	< 0.05		0,05		0,11		0,08		1,5
Chloride	mg Cl/l	1,50		2,28		4,94		4,32		200
Nitrite	mg NO2/l	< 0.05		< 0.05		< 0.05		< 0.05		0,16
Bromide	mg Br/l	< 0.1		< 0.1		< 0.1		< 0.1		
Nitrate	mg NO3/l	0,55		0,66		0,76		0,63		44
Phosphate	mg PO4/l	< 0.2		< 0.2		< 0.2		< 0.2		
Sulphate	mg SO4/l	3,96		4,57		7,48		8,80		100
Sum ALK and anions	meq/l	0,30		0,36		0,44		0,48		
Cations										
Silicon	mg Si /l	1,60		2,12		2,90		3,20		
Aluminum	mg Al/l	<0.02		<0.02		0,025		0,026		0,2
Iron	mg Fe/l	0,0245		0,0971		0,0206		0,0166		0,2
Magnesium	mg Mg/l	0,478		0,581		0,776		0,761		
Calcium	mg Ca/l	3,51		4,21		4,33		5,62		
Sodium	mg Na/l	1,32		1,72		2,55		2,48		200
Potassium	mg K/l	0,82		1,06		1,630		1,620		
Manganese	mg Mn/l	0,0032		0,0082		0,0123		0,0189		0,05
Copper	mg Cu/l	0,0050		0,0058		0,0054		<0.005		0,1
Zink	mg Zn/l	0,0508		0,0596		0,0694		0,0485		0,3
Lead	mg Pb/l	<0.005		<0.005		<0.005		<0.005		0,01
Nickel	mg Ni/l	<0.005		<0.005		<0.005		<0.005		0,02
Cadmium	mg Cd/l	<0.0005		<0.0005		<0.0005		<0.0005		0,005
Cromium	mg Cr/l	<0.002		<0.002		<0.002		<0.002		0,05
Phosphorous	mg P/l	<0.05		<0.05		<0.05		<0.05		1,1
Sum cations	meq/l	0,29		0,36		0,43		0,49		
Ion balance error	%	**-1,8**		**0,5**		**-1,35**		**0,8**		

APPENDIX IV: GROUNDWATER LEVEL SCHEMES

Groundwater level below well top (m)

Date	Remarks	Piezometer well												
		1	4	5	6	7	9	10	12	13	14	21	22	23
16.02.2009		-*	3,880	4,310	4,980	4,510	3,880	3,950	4,840	4,970	5,105	3,860	3,890	3,750
17.02.2009		-*	3,890	4,330	5,160	3,300	3,880	3,970	4,940	5,210	5,250	3,910	4,190	3,840
30.03.2009	Early	-*	3,88	4,38	5,31	3,17	3,89	3,99	5,26	5,38	5,48	4,07	4,15	3,945
30.03.2009	Late	-*	3,88	4,44	5,45	3,43	3,89	3,99	5,25	5,59	5,62	4,13	4,660	4,120
31.03.2009		-*	3,91	4,54	5,69	3,51	3,9	4,05	5,44	5,95	5,97	4,34	5,16	4,49
02.04.2009		3,05	3,92	4,61	5,85	3,42	3,92	4,07	5,94	6,12	6,22		6,60	6,03
03.04.2009		-*	3,94	4,58	5,76	3,54	3,92	4,07	5,67	5,94	6,04	5,94	6,05	5,87
07.04.2009		-*	3,87	4,44	5,40	3,47	3,84	3,98	5,19	5,48	5,60	5,60	5,63	5,44
29.04.2009		-*	3,53	3,94	4,57	2,82	3,51	3,55	4,39	4,62	4,72	4,93	4,85	4,72
02.06.2009		-*	2,97	3,51	4,32	2,31	3,00	3,05	4,30	4,57	4,69	4,69	5,37	4,90
08.09.2009		-*	3,39	3,85	4,55	2,44	3,32	3,41	4,47	4,65	4,80	4,96	5,47	5,05

Groundwater level above sea water level (m a.s.l.)

Date	Remarks	Piezometer well												
		1	4	5	6	7	9	10	12	13	14	21	22	23
16.02.2009		-*	2,690	2,974	1,962	1,115	2,773	2,646	1,990	2,041	2,085	3,307	1,791	1,770
17.02.2009		-*	2,680	2,954	1,782	2,325	2,773	2,626	1,890	1,801	1,940	3,257	1,491	1,680
30.03.2009	Early	-*	2,690	2,904	1,632	2,455	2,763	2,606	1,570	1,631	1,710	3,097	1,531	1,575
30.03.2009	Late	-*	2,690	2,844	1,492	2,195	2,763	2,606	1,580	1,421	1,570	3,037	1,021	1,400
31.03.2009		-*	2,660	2,744	1,252	2,115	2,753	2,546	1,390	1,061	1,220	2,827	0,521	1,030
02.04.2009		3,05	2,65	2,67	1,09	2,21	2,73	2,53	0,89	0,89	0,97	-	-0,92	-0,51
03.04.2009		-*	2,63	2,70	1,18	2,09	2,73	2,53	1,16	1,07	1,15	1,23	-0,37	-0,35
07.04.2009		-*	2,70	2,84	1,54	2,16	2,81	2,62	1,64	1,53	1,59	1,57	0,05	0,08
29.04.2009		-*	3,04	3,34	2,37	2,81	3,14	3,05	2,44	2,39	2,47	2,24	0,83	0,80
02.06.2009		-*	3,60	3,77	2,62	3,32	3,65	3,55	2,53	2,44	2,50	2,48	0,31	0,62
08.09.2009		-*	3,18	3,44	2,39	3,19	3,33	3,19	2,37	2,36	2,39	2,21	0,21	0,47

* Obstruction in piezometer made it impossible to measure.

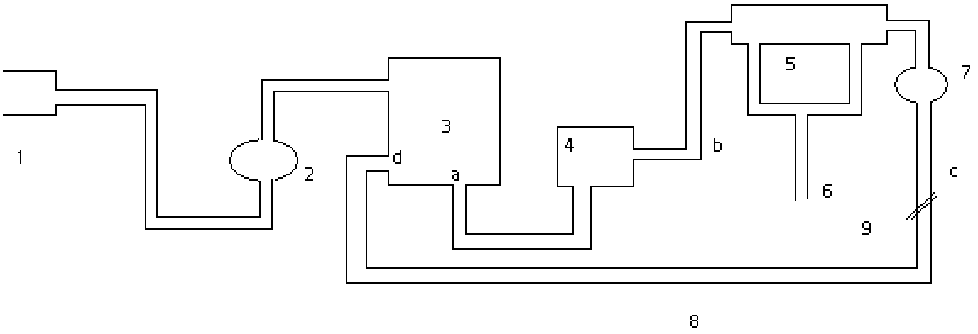
- Not measured.

APPENDIX V: MICROBIOLOGY

SETUP OF THE HEMOFLOW INSTALLATION

The illustration shows the setup of equipment used for the concentration of large volumes of water with the Hemoflow-method (Veenendaal, H. R. and Brouwer-Hanzens, A. J. 2007). Different components are listed by number:

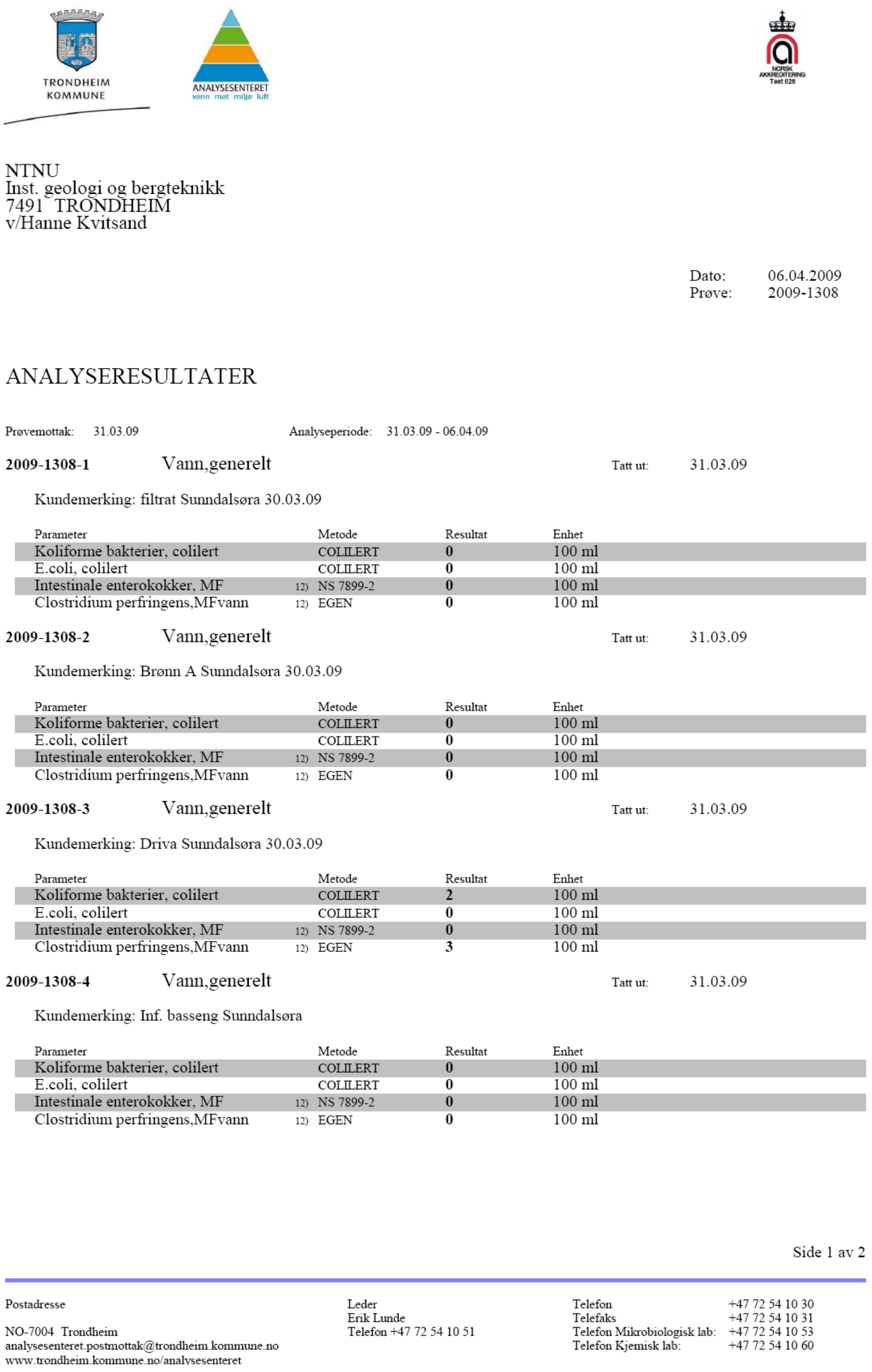
1. Sample point
2. Water meter
3. Tank with floater
4. Pump
5. Hemoflow filter
6. Filtrate
7. Pressure meter
8. Return hose
9. Tube clamp (a, b, c and d are connecting point for tubes)



MICROBIOLOGICAL GUIDE VALUES (DRINKING WATER REGULATIONS)

Microbiological guide values (Drinking water regulations)	
Bacterial counts, at 22 °C (pr. ml.)	100
E.coli (pr. 100 ml)	0
Intestinal enterococci (pr. 100 ml)	0
Coliforms (pr. 100 ml)	0
Clostridium perfringens (pr. 100 ml)	0

MICROBIOLOGICAL ANALYSES I



TRONDHEIM
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ANALYSESENTERET
vann mat miljø luft

NORSK
AKKREDITERING
Test 028

NTNU
Inst. geologi og bergteknikk
7491 TRONDHEIM
v/Hanne Kvitsand

Dato: 06.04.2009
Prøve: 2009-1308

ANALYSERESULTATER

Prøvemottak: 31.03.09 Analyseperiode: 31.03.09 - 06.04.09

2009-1308-1 Vann,generelt Tatt ut: 31.03.09

Kundemerking: filtrat Sunndalsøra 30.03.09

Parameter		Metode	Resultat	Enhet
Koliforme bakterier, colilert		COLILERT	**0**	100 ml
E.coli, colilert		COLILERT	**0**	100 ml
Intestinale enterokokker, MF	12)	NS 7899-2	**0**	100 ml
Clostridium perfringens,MFvann	12)	EGEN	**0**	100 ml

2009-1308-2 Vann,generelt Tatt ut: 31.03.09

Kundemerking: Brønn A Sunndalsøra 30.03.09

Parameter		Metode	Resultat	Enhet
Koliforme bakterier, colilert		COLILERT	**0**	100 ml
E.coli, colilert		COLILERT	**0**	100 ml
Intestinale enterokokker, MF	12)	NS 7899-2	**0**	100 ml
Clostridium perfringens,MFvann	12)	EGEN	**0**	100 ml

2009-1308-3 Vann,generelt Tatt ut: 31.03.09

Kundemerking: Driva Sunndalsøra 30.03.09

Parameter		Metode	Resultat	Enhet
Koliforme bakterier, colilert		COLILERT	**2**	100 ml
E.coli, colilert		COLILERT	**0**	100 ml
Intestinale enterokokker, MF	12)	NS 7899-2	**0**	100 ml
Clostridium perfringens,MFvann	12)	EGEN	**3**	100 ml

2009-1308-4 Vann,generelt Tatt ut: 31.03.09

Kundemerking: Inf. basseng Sunndalsøra

Parameter		Metode	Resultat	Enhet
Koliforme bakterier, colilert		COLILERT	**0**	100 ml
E.coli, colilert		COLILERT	**0**	100 ml
Intestinale enterokokker, MF	12)	NS 7899-2	**0**	100 ml
Clostridium perfringens,MFvann	12)	EGEN	**0**	100 ml

Side 1 av 2

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MICROBIOLOGICAL ANALYSES II

TRONDHEIM
KOMMUNE

ANALYSESENTERET
vann mat miljø luft

NORSK
AKKREDITERING
Test 028

Dato: 06.04.2009
Prøve: 2009-1308

2009-1308-5 Vann,generelt Tatt ut: 31.03.09

Kundemerking: konsentrat Brønn A

Parameter	Metode	Resultat	Enhet
E.coli, colilert	COLILERT	**0**	100 ml

2009-1308-6 Vann,generelt Tatt ut: 31.03.09

Kundemerking: konsentrat inf basseng

Parameter	Metode	Resultat	Enhet
E.coli, colilert	COLILERT	**0**	100 ml

2009-1308-7 Vann,generelt Tatt ut: 31.03.09

Kundemerking: konsentrat Driva

Parameter	Metode	Resultat	Enhet
E.coli, colilert	COLILERT	**3**	100 ml

12) Måleusikkerhet er ikke beregnet. Resultatene er oppgitt som cfu

Måleusikkerhet: Vi henviser til www.trondheim.kommune.no/analysesenteret eller ved henvendelse til laboratoriet.
Resultatene gjelder bare for prøvene i rapporten. Rapporten må ikke gjengis i utdrag uten skriftlig tillatelse.

Med hilsen

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Astrid Lian
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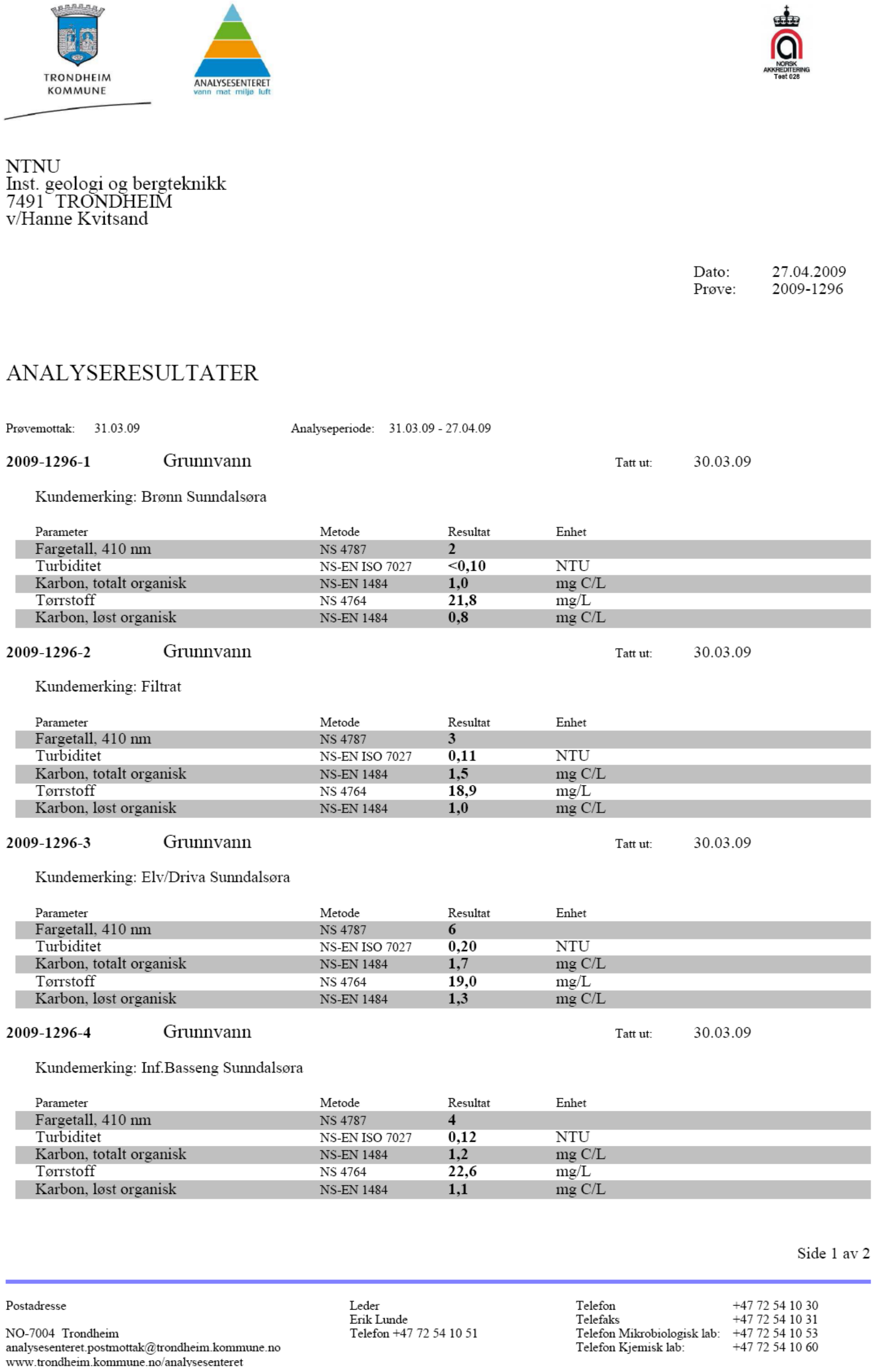
MICORBIOLOGICAL ANALYSES III



TRONDHEIM KOMMUNE

ANALYSESENTERET
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NORSK AKKREDITERING Test 028

NTNU
Inst. geologi og bergteknikk
7491 TRONDHEIM
v/Hanne Kvitsand

Dato: 27.04.2009
Prøve: 2009-1296

ANALYSERESULTATER

Prøvemottak: 31.03.09 Analyseperiode: 31.03.09 - 27.04.09

2009-1296-1 Grunnvann Tatt ut: 30.03.09

Kundemerking: Brønn Sunndalsøra

Parameter	Metode	Resultat	Enhet
Fargetall, 410 nm	NS 4787	**2**	
Turbiditet	NS-EN ISO 7027	**<0,10**	NTU
Karbon, totalt organisk	NS-EN 1484	**1,0**	mg C/L
Tørrstoff	NS 4764	**21,8**	mg/L
Karbon, løst organisk	NS-EN 1484	**0,8**	mg C/L

2009-1296-2 Grunnvann Tatt ut: 30.03.09

Kundemerking: Filtrat

Parameter	Metode	Resultat	Enhet
Fargetall, 410 nm	NS 4787	**3**	
Turbiditet	NS-EN ISO 7027	**0,11**	NTU
Karbon, totalt organisk	NS-EN 1484	**1,5**	mg C/L
Tørrstoff	NS 4764	**18,9**	mg/L
Karbon, løst organisk	NS-EN 1484	**1,0**	mg C/L

2009-1296-3 Grunnvann Tatt ut: 30.03.09

Kundemerking: Elv/Driva Sunndalsøra

Parameter	Metode	Resultat	Enhet
Fargetall, 410 nm	NS 4787	**6**	
Turbiditet	NS-EN ISO 7027	**0,20**	NTU
Karbon, totalt organisk	NS-EN 1484	**1,7**	mg C/L
Tørrstoff	NS 4764	**19,0**	mg/L
Karbon, løst organisk	NS-EN 1484	**1,3**	mg C/L

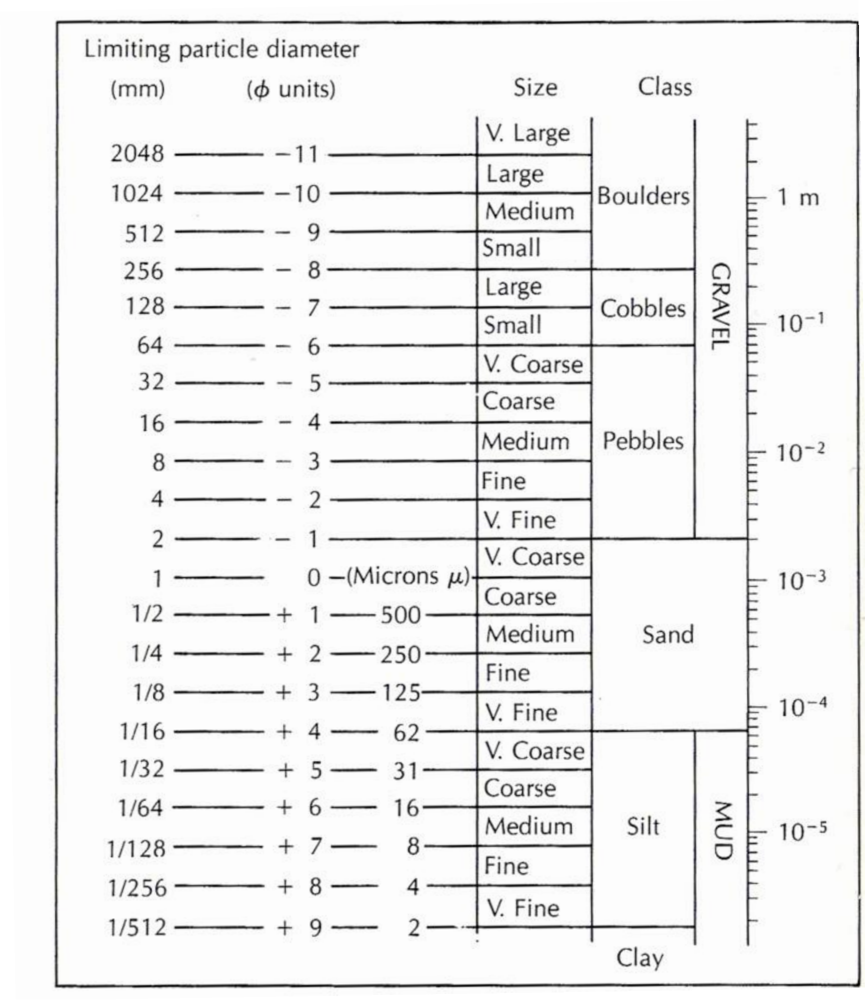
2009-1296-4 Grunnvann Tatt ut: 30.03.09

Kundemerking: Inf.Basseng Sunndalsøra

Parameter	Metode	Resultat	Enhet
Fargetall, 410 nm	NS 4787	**4**	
Turbiditet	NS-EN ISO 7027	**0,12**	NTU
Karbon, totalt organisk	NS-EN 1484	**1,2**	mg C/L
Tørrstoff	NS 4764	**22,6**	mg/L
Karbon, løst organisk	NS-EN 1484	**1,1**	mg C/L

APPENDIX VI: CLASSIFICATION OF SEDIMENTS

Sediment size is reported with respect to the grain size classification scheme after Fetter (2001, therein Friedman and Sanders 1978):



APPENDIX VII: GROUNDWATER PROTECTION ZONES (NORWEGIAN)

According to NGU (1992 a) and Asplan Viak (2009):

Sone 3. *Foreslåtte restriksjoner og arealbruk innenfor denne sonen styres i stor grad av kommunenes arealplaner (kommuneplan og kommunedelplan) og annet lovverk som Forurensningsloven, Vannressursloven og Plan- og bygningsloven.* ***Terrenginngrep og aktivitetsendringer som ikke er nevnt spesielt i det etterfølgende skal forelegges drikkevannsmyndighetene før gjennomføring og eventuelt konsekvensvurderes i forhold til grunnvannet.*** *Følgende restriksjoner gjelder for sone 3:*

1. *Forbud mot etablering av bedrifter, virksomheter eller anlegg som kan utgjøre en potensiell forurensningstrussel mot grunnvannet. Bestemmelsene omfatter også bedrifter som bruker slike stoffer som råstoff i produksjonen.*

2. *Forbud mot deponering av husholdningsavfall, kloakkslam og annet organisk avfall (gjelder også mellomlagring av kloakkslam til bruk på dyrket mark).*

3. *Forbud mot ubeskyttet lagring utendørs av stoffer med utvaskbare kjemikalier. Kunstgjødsel kan lagres ute på pall dersom emballasjen hindrer utvasking.*

4. *Forbud mot lagring av drivstoff, plantevernmidler eller kjemikalier i større mengder enn det som er nødvendig for gårdsdrift og husholdning (maksimum 1 års forbruk).*

5. *Forbud mot nedgravde olje- eller kjemikalietanker. Maksimum lagertank for olje og oljeprodukter er 3m*^{3}*. Alle lagertanker må stå på tett underlag med kanter støpt høye nok til å samle opp hele tankens innhold. Tankene skal være lette å inspisere for lekkasjer. Makimalt 1 tank pr. eiendom innenfor sone 3.*

6. *Forbud mot større/felles separate avløpsanlegg.*

7. *Større masseuttak ut over grunneiers naturlige forbruk kan bare skje etter godkjent reguleringsplan.*

Det ***tillates*** *bruk av gjødsel og godkjente sprøytemidler i jordbruksområder i henhold til godkjent gjødselplan.*

Sone 2. *Innenfor sone 2 anbefales det forbud mot følgende aktiviteter som kan medføre økt forurensningsfare:*

8. *Forbud mot oppføring av bygninger med unntak av enkle bygninger (uten innlagt vann) tilknyttet skogbruk (vedlager, flyttbare skogsbrakker etc.).*

9. *Forbud mot infiltrasjon av avløpsvann i grunnen.*

10. *Forbud mot lagring av kjemikalier, olje og oljeprodukter.*

11. *Forbud mot lagring og deponering av fór (silo, lutet halm, rundballer etc).*
12. *Forbud mot bruk av sprøytemidler (skog).*
13. *Forbud mot masseuttak.*
14. *Forbud mot omfattende gravearbeider i elveløp. Eventuelle tiltak for å hindre videre erosjon fra elv bør konsekvensutredes og skal forelegges drikkevannsmyndigheter og vassdragsmyndigheter for endelig godkjenning.*
15. *Forbud mot camping m/overnatting.*
16. *Forbud mot etablering av nye konkurrerende brønner med unntak av grunnvannsuttak av grunneier til eget forbruk i henhold til avtale.*

Sone 1. *Følgende restriksjoner foreslås:*

17. *Forbud mot oppstilling/parkering av kjøretøy med unntak av for drift av vannverket.*
18. *Forbud mot lagring/opplasting av tømmer.*
19. *Forbud mot lagring eller fylling av drivstoff eller annen form for forurensende aktivitet, bortsett fra kjøring.*
20. *Forbud mot bruk av hogstmaskiner. Vedhogst og skogsdrift med bruk av motorsag og traktor tillates.*
21. *Forbud mot leirslagning.*
22. *Forbud mot etablering av andre grunnvannsbrønner enn i tilknytning til vannverket.*
23. *Utmarksbeite kan tillates.*

Sone 0 (Brønnområdet):

24. *Sonen skal kun benyttes til aktivitet knyttet til driften av vannverket.*

APPENDIX VIII: CD