

Mobilisation of geogenic arsenic into groundwater in Västerbotten County, Sweden

Mobilisering av naturligt förekommande arsenik
till grundvatten i Västerbotten, Sverige

Marcus Svensson

ABSTRACT

Mobilisation of geogenic arsenic into groundwater in Västerbotten County, Sweden

Arsenic is a naturally - occurring element in the earth's crust, bedrock and soil. Hydrogeochemical processes mobilise geogenic arsenic into groundwater. This has caused worldwide problems as groundwater with elevated concentrations of dissolved arsenic is the main potable water source for millions of people. In Swedish groundwater, the arsenic concentration is often low and well below the National Food administration's guideline value of 10 µg/l. In areas with sulphur-rich rock and glacial till, however, the arsenic concentration can be significantly higher.

In this study hydrogeological and hydrochemical conditions favourable for elevated concentrations of arsenic were delineated through the correlation between groundwater chemistry, geology and arsenic concentrations.

The study was done in the Skellefteå field in northern Sweden where glacial till with high arsenic concentrations has been encountered. 44 wells, both tube wells and dug wells, were sampled and analysed for arsenic and major anions and cations. In addition to these, 96 analyses from domestic wells were obtained from municipalities within the study area. Geological maps were obtained from the Västerbotten County Administration Board, which were imported into a GIS-system.

Arsenic concentrations varied between below the detection limit (<0.5 µg/l) and 300 µg/l. The arsenic concentrations in the tubewells were linked to their respective bedrock group and statistical data was calculated for the arsenic concentration in each group. The statistical analysis suggests that high arsenic groundwater and alkaline volcanic rock correlate while low arsenic concentration correlates with sedimentary rocks. Groundwater samples were classified into redox-classes based on the concentration of dissolved iron, manganese and sulphate. This was done for the samples where these constituents had been analysed (n=51). It was revealed that a large number of wells were "mixed waters", thus it was hard to conclude whether they were reduced or oxidised. Arsenic was plotted versus other analysed parameters. High nitrate concentration correlated with low dissolved iron and arsenic concentration. It is suggested that nitrate may lead to the oxidation of Fe²⁺ resulting in the formation of iron oxyhydroxides to which arsenic may adsorb; thus, arsenic and iron concentrations are low in high nitrate waters.

Based on the study two hypotheses are proposed as responsible for the high arsenic ground waters in the study area. Hypothesis I suggests reduction of iron oxyhydroxides in glacial till as the main source, whereas hypothesis II suggests oxidation of primary minerals in the crystalline rock as the main source of arsenic.

Each of the high arsenic wells was linked to a respective hypothesis based on the groundwater composition. The process described in hypothesis II is responsible for the main part of arsenic in the ground water even though hypothesis I cannot be disregarded completely as two samples were identified as potential hypothesis I wells.

Keywords: Arsenic, groundwater, adsorption, desorption, iron oxyhydroxide, redox, Västerbotten, Skellefteå field.

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REFERAT

Mobilisering av naturligt förekommande arsenik till grundvatten i Västerbotten, Sverige

Arsenik är ett naturligt förekommande element i jordskorpan, i berggrunden och i jord. Hydrogeokemiska processer mobiliserar arsenik till grundvattnet vilket orsakar problem världen över då grundvatten med förhöjda koncentrationer av löst arsenik används som huvudsaklig källa för dricksvatten. I Svenska grundvatten är arsenik koncentrationen ofta låg och betydligt lägre än Livsmedelsverkets gränsvärde på 10 µg/l. I områden med sulfidrikt berg och morän kan dock arsenik koncentrationen vara betydligt högre.

I denna studie har hydrogeologiska/kemiska förhållanden gynnsamma för förhöjda arsenik koncentrationer beskrivits genom korrelationen mellan grundvattenkemin, geologin och arsenik koncentrationen. Studien genomfördes på Skelleftefältet i norra Sverige där morän med hög arsenik halt har påträffats. 44 brunnar, både borrade och grävda provtogs och analyserades för arsenik och vanliga anjoner och katjoner. Förutom dessa har 96 analyser från privata brunnar erhållits av kommuner inom studie området. Geologiska kartor erhöles från länsstyrelsen i Västerbotten för att importeras till ett GIS-system.

Arsenik koncentrationen varierade från under detektionsgränsen (<5 µg/l) till 300 µg/l. Arsenik i bergborrade brunnar kopplades till sin respektive berggrundstyp och statistisk data beräknades för arsenik koncentrationen i respektive grupp. Den statistiska analysen antyder att grundvatten med hög arsenik koncentrationen korrelerar med basisk vulkanit medan låg arsenik koncentration korrelerar med sedimentär berggrund. Grundvattenproverna klassificerades i redox-klasser baserat på koncentrationen löst järn, mangan och sulfat. Detta gjordes för de prover där dessa element blivit analyserade (n=51). Det visade sig att ett stort antal brunnar var "mixat vatten", det var således svårt att avgöra om dessa var reducerade eller oxiderade. Arsenik plottades mot analyserade parametrar. Hög nitrat koncentration korrelerade med låg löst järn och arsenik koncentration. Det föreslås att nitrat kan leda till oxidation av Fe²⁺ som resulterar i bildandet av järn-oxyhydroxider till vilka arsenik kan adsorberas, alltså är arsenik och järn koncentrationerna låga i vatten med hög nitrat koncentration.

Baserat på studien föreslås två hypoteser som skyldiga till de höga arsenik koncentrationer som påträffats i studie området. Hypotes I föreslår reduktion av järn-oxyhydroxider i morän som den huvudsakliga källan medan hypotes II föreslår oxidation av primära mineral i den kristallina berggrunden som den huvudsakliga källan till arsenik.

Var och en av de brunnar med hög arsenik koncentration kopplades till sin respektive hypotes baserat på grundvatten sammansättningen. Processen som beskrivs i hypotes II står för huvuddelen av arseniken i grundvattnet även om hypotes I inte helt kan avfärdas då två brunnar identifierades som potentiella hypotes I brunnar.

Nyckelord: Arsenik, grundvatten, adsorption, desorption, järn-oxyhydroxid, redox, Västerbotten, Skelleftefältet.

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PREFACE

This masters thesis has been carried out as a part of the Aquatic and Environmental Engineering program at Uppsala University and has been conducted as a co-operation between Ramböll, The Royal Institute of Technology (KTH- International Groundwater Arsenic Research Group (Prof. Gunnar Jakcs)) and Uppsala University. The master thesis is a pre-study to a SGU financed project: "*Arsenic in Swedish groundwater- mobility and risk for elevated concentrations*" that will be conducted by KTH (Prosun Bhattacharya a.o.) during 2007-2008.

This project has been enabled thanks to my supervisor, Mattias von Brömssen, at Ramböll and Gunnar Jacks (KTH) who has helped me with the water sampling. I would also like to thank Dr. Roger Herbert at Uppsala University who has been functioning as subject reviewer and all the nice people who I met in Västerbotten.

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1. Introduction

Arsenic (As) is a metalloid and a naturally - occurring element in all soil, rock and sediments. Both natural and anthropogenic activities result in an input of arsenic on the environment. Natural processes like erosion, weathering and emissions from volcanoes contribute to the accessibility of arsenic. A number of anthropogenic processes are also, indirectly and directly, responsible for the accumulation of arsenic in nature, for example, mining and metallurgy industry: In addition, arsenic is used as wood preservatives and in waste management (Sarkar 2002; Welch and Stollenwerk 2003). Natural contents in Sweden are generally ranging from a few to 10 mg kg⁻¹ in the soil. However in some areas of Sweden with sulphur-rich rocks and in areas with arsenic-rich sediments the content in soil can be as high as 30 mg kg⁻¹ (SGU, 2005). Such areas are found in Västerbotten (the Skellefteå ore-field area), on Öland, in the region of Enköping-Västerås and around Sollefteå (Berglund et al. 2005, SGU, 2005). In these areas, elevated concentrations of geogenic arsenic have been found also in the groundwater.

Arsenic in groundwater has in recent years been given more attention both internationally and nationally, due to various health effects caused by inorganic dissolved arsenic (chapter 1.2). Many surveys studying the origin and processes responsible for elevated concentrations of geogenic (i.e. naturally - occurring) arsenic in groundwater have been conducted (Smedley and Kinniburgh. 2002, Lipfert et al. 2006, Backman et al. 2006, Geological survey of Finland 2006, Peters et al 2003). However, few consider geogenic arsenic in waters of Sweden. Thus, knowledge of conditions and processes responsible for elevated arsenic concentration in Swedish groundwater is very limited. A few feasibility studies have been performed by the Geological Survey of Sweden (SGU) and the municipality of Norsjö, Malå and Skellefteå. Those studies have been limited in the number of samples and have focused on mapping.

1.1 Objective

The objective of this study is to delineate geological conditions favourable for, and to understand the processes responsible for, high levels of geogenic arsenic in Swedish groundwaters in the Västerbotten region.

To achieve the objective, the correlations between groundwater chemistry, geology and elevated arsenic concentrations have been studied. Analysed samples complement analyses conducted by municipalities and SGU in the Västerbotten region and therefore contribute to a better view of the problem. The study may also provide information for the designing a tool for identifying low arsenic aquifers as suggested by von Brömssen et al. (2006).

1.2 Arsenic Toxicity

Humans are exposed to arsenic through food, water and air. Arsenic levels in food are often low with the exception of fish and shellfish, which adsorb arsenic from water. These are, however, organic forms of arsenic that are less harmful to humans.

Inorganic forms of arsenic are much more toxic and can cause various health effects such as irritation of stomach and intestines, decreased production of red and white blood cells, skin abnormalities and lung irritation. Arsenic is also carcinogenic and long term exposure can cause tumours in skin, lung, kidney, and urine bladder (IMM, 2005). The main source for exposure of inorganic forms of arsenic to humans is potable water and, due to the health effects caused by dissolved arsenic, knowledge of the behaviour and occurrence of arsenic in groundwater is of importance. The guideline value for arsenic in potable water is based on the lifetime risk for cancer and was in 2003 lowered in Sweden from $50 \mu\text{g l}^{-1}$ to $10 \mu\text{g l}^{-1}$. The state of New Jersey, USA, lowered in January 2006 the guideline value from 10 to $5 \mu\text{g/l}$ in order to protect the public health (NJGS, 2006). Countries with a widespread arsenic problem have a guideline value still set at $50 \mu\text{g l}^{-1}$ and WHO has set the recommended Provisional Tolerably Weekly Intake (PTWI) to $15 \mu\text{g/kg}$ body mass, which translates to about $150 \mu\text{g/day}$ for an adult.

2. Arsenic in natural waters

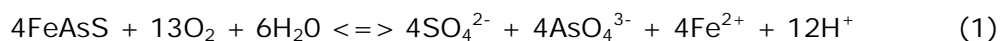
2.1 Source

Arsenic is a natural constituent of the earth's crust and occurs as a major constituent in more than 200 minerals.

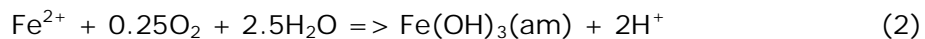
Arsenic can enter the environment through natural or anthropogenic processes. The most common anthropogenic sources are mining industry, agriculture, wood preservation activities and coal combustion (Bhattacharya, 2002). One of the most important factors influencing the concentration of geogenic arsenic is the arsenic content in the bedrock or the parent materials from which the soil is derived. Arsenic commonly associates with sulfide minerals of which pyrite (FeS_2) is the most abundant. Pyrite is formed in a low temperature sedimentary environment under reducing conditions. During formation some of the soluble arsenic will be incorporated in the crystal structure as a substitute for S resulting in an arsenic - rich mineral such as arsenopyrite (FeAsS) (Smedley and Kinniburgh, 2002).

2.1.1 Weathering

Arsenic is first mobilised from the soil environment due to chemical weathering of arsenopyrite or other arsenic- rich primary minerals. Weathering of the minerals depends on several different factors, i.e. pH, the presence of water, dissolved oxygen, temperature and the redox potential. In an oxidising environment, oxidation of arsenopyrite results in the production of SO_4^{2-} and H^+ ions resulting in decrease in pH as the following reaction shows.



In the reaction arsenic is oxidised from As^0 to As(V) forming oxyanions. Depending on the level of dissolved O_2 Fe^{2+} can be oxidized to Fe^{3+} and precipitate as Fe^{3+} -oxyhydroxides.



In groundwater where the source of aqueous arsenic is weathered sulphides, a correlation between sulphate and arsenic might be expected. This may however not be true as the formation of secondary minerals including sulphate (e.g. $\text{CaSO}_4 \cdot 2\text{H}_2\text{O}$ in the presence of high Ca^{2+}) and/or adsorption of arsenic may occur thus disrupting the relationship.

2.2 Factors affecting arsenic mobility in natural waters

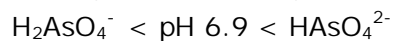
Arsenic is present in the environment in several oxidation states but occurs mainly in groundwater as oxyanions of trivalent arsenite (As(III)) or pentavalent arsenate (As(V)). The transformation between different oxidation states is complex and depends not only

on the redox environment but also on biological processes, ligand exchange, and sorption processes, which makes the speciation of arsenic in natural waters difficult to predict (Korte and Fernando, 1991). The mobility of arsenic, where arsenic speciation has an important role, is controlled by adsorption/desorption processes in a natural soil-rock-water system. The sorption processes are in turn dependent on pH and the redox environment.

2.2.1 pH

When arsenic is mobilised into water, through e.g. weathering, it forms oxyanions (reaction 1). The oxyanions behave like acids releasing protons in steps (so called deprotonation), thus the charge of the anions is governed by the pH making it an important factor controlling the mobilisation of arsenic. Figure 1 shows the stepwise deprotonation of H_3AsO_4 and H_3AsO_3 as a function of pH.

As(V) is present either as $HAsO_4^{2-}$ or as $H_2AsO_4^-$ with a shift at pH 6.9;



For As(III) the shift occurs at pH 9.0;

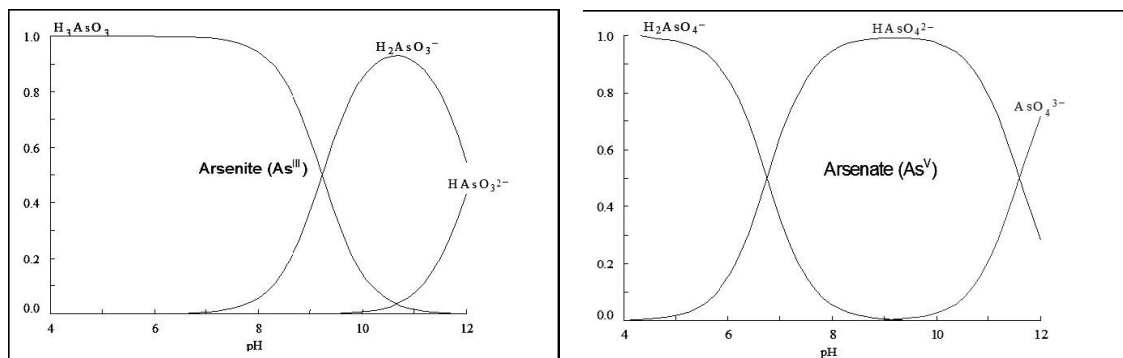
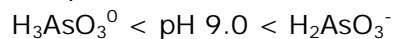


Figure 1. Distribution diagram of As(V) and As(III) as function of pH (Smedley and Kinniburgh, 2002).

Arsenic oxyanions form surface complexes with metal oxyhydroxides (e.g. Al, Mn or Fe oxyhydroxides) in natural waters. As the positive charge of the metal oxyhydroxide surface increases with decreasing pH the affinity of anions for the positively charged adsorption sites on the metal hydroxide increases. This will lead to an increase in adsorption of arsenic with decreasing pH. The larger negative charge of the As(V) oxyanion compared to the As(III) oxyanion results in a larger As(V) affinity for the positively charged metal oxyhydroxide surface, leading to a reduced mobility of the As(V) species compared to As(III). Besides governing the charge of the anion, a decrease in pH will lead to an increase in the proportion of adsorption sites favouring the complex formation with As anions (Gustafsson et al. 2005). Inversely, an increase in pH will lead to desorption of adsorbed As resulting in As mobilization. Figure 1 shows the surface charge of iron oxyhydroxide as a function of pH and figure 3 shows the solubility of amorphous $Fe(OH)_3$ and goethite [α - $FeOOH$] as a function of pH.

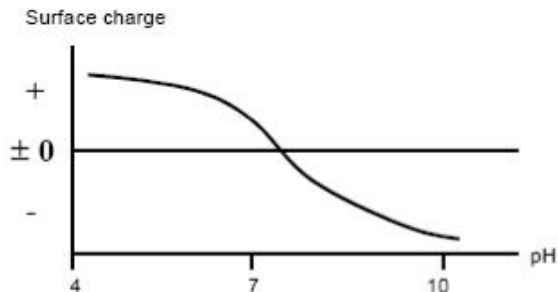


Figure 2. Example of surface charge on iron oxyhydroxide as a function of pH.

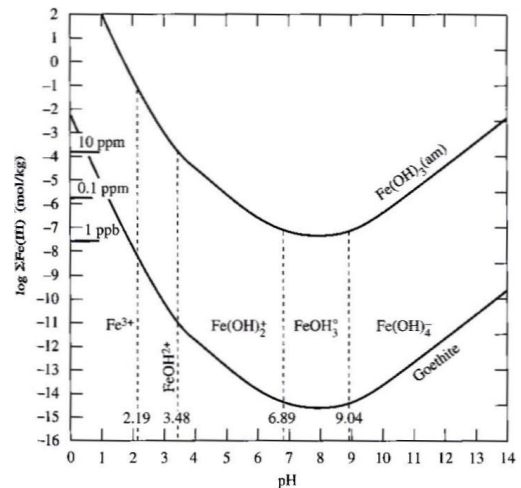


Figure 3. Solubility of amorphous $\text{Fe}(\text{OH})_3$ and goethite [$\alpha\text{-FeOOH}$] as a function of pH. Also shown are fields with, at the present pH, dominant Fe^{3+} -hydroxy-complexes (Langmuir, 1997).

2.2.2 Redox environment

In the environment a series of redox reactions occurs that are of importance for arsenic speciation and for the behaviour of metal oxyhydroxides, which both control the mobility of arsenic. The redox potential (E_h ; measured in volts) determines the tendency for electron transfer in a system depending on the availability of electron acceptors. The sequence of redox-reactions given in table 1 for a soil-water system begins with the consumption of O_2 as bacteria degrade organic matter and release dissolved carbon dioxide. When the oxygen is depleted, nitrate (NO_3^-) is used as electron acceptor, if available, and so on. The sequence ends with the production of methane from fermentation and methanogenesis. Reduction of $\text{As}(\text{V})$ normally occurs after Fe^{3+} reduction but before the sulphate reduction (Smedley and Kinniburgh, 2002). This means that $\text{As}(\text{V})$ is the dominant species in an oxidizing to slightly reducing condition. The reduction rate of $\text{As}(\text{V})$ or the oxidation of $\text{As}(\text{III})$ is however relatively slow. This is due to biochemical processes in the environment which disrupts the redox equilibrium. This means that $\text{As}(\text{V})$ can also be found in some anoxic waters and $\text{As}(\text{III})$ can be found in an oxic environment (Ali and Ahmed, 2003). The behaviour of arsenic in terms of mobility can be described in three different redox zones (see Table 2).

Table 1. Sequence of principle redox reactions with decreasing Eh(V).The <> symbols mean that organic carbon is oxidised

Reaction	Redox couples	Equation
Oxygen reduction	O_2/H_2O < CH_2O >/ CO_2	$CH_2O + O_2 \rightarrow CO_2 + H_2O$
Denitrification		
Heterotrophic	NO_3^-/N_2 < CH_2O >/ CO_2	$CH_2O + 4/5 NO_3^- + 4/5 H^+ \rightarrow CO_2 + 2/5 N_2 + 7/5 H_2O$
Autotrophic	NO_3^-/N_2 FeS_2/Fe^{2+}	$5FeS_2 + 14NO_3^- + 4H^+ \rightarrow 7N_2 + 10SO_4^{2-} + 5Fe^{2+} + 2H_2O$
Manganese reduction	MnO_2/Mn^{2+} < CH_2O >/ CO_2	$CH_2O + 2MnO_2 + 4H^+ \rightarrow 2Mn^{2+} + 3H_2O + CO_2$
Iron reduction	$Fe(OH)_3/Fe^{2+}$ < FeS_2 >/ SO_4	$FeS_2 + 14Fe(OH)_3 + 26H^+ \rightarrow 15Fe^{2+} + 2SO_4^{2-} + 34H_2O$
Iron reduction	$Fe(OH)_3/Fe^{2+}$ < CH_2O >/ CO_2	$CH_2O + 4Fe(OH)_3 + 8H^+ \rightarrow 4Fe^{2+} + 11H_2O + CO_2$
Arsenic reduction	$H_2AsO_4^-/H_3AsO_3$ < CH_2O >/ CO_2	$CH_2O + 2H_2AsO_4^- + 4H^+ \rightarrow 2H_3AsO_3 + H_2O + CO_2$
Sulfate reduction	SO_4/HS^- < CH_2O >/ CO_2	$CH_2O + 1/2 SO_4^{2-} + 1/2 H^+ \rightarrow 1/2 HS^- + H_2O + CO_2$
Methanogenesis	CO_2/CH_4 < CH_2O >/ CO_2	$CH_2O + 1/2 CO_2 \rightarrow 1/2 CH_4 + CO_2$

Table 2. The behaviour of arsenic conceptualised in three redox zones (from Sracek et al., 2004)

1) Oxidizing environment in which the dominating iron species is Fe^{3+} , arsenic concentrations are low due to adsorption processes.
2) Slightly reducing environment where iron reduction takes place, arsenic concentrations might be elevated as adsorbed arsenic is released when Fe^{2+} goes to the liquid phase.
3) Strong reducing environment where sulphate reduction takes place, arsenic concentration is low as the reduced sulphate precipitates as H_2S and arsenic co-precipitates as secondary arsenopyrite. This however presupposes that SO_4^{2-} is present in enough quantity.

2.2.3 Adsorption / desorption and the mobility of arsenic

The adsorption/ desorption processes are among the most important factors controlling the concentration of arsenic in groundwater. In general, arsenic can be adsorbed to three different surfaces: clay particles, metal hydroxides and humus substances. The adsorption leads to immobilisation of arsenic. According to Smedley and Kinniburgh (2002) the interaction between organic matter and arsenic is subordinate the interaction between mineral-arsenic. Because of the low proportion of organic matter in till, it is assumed that the interaction between organic matter and arsenic in Västerbotten is of minor importance in controlling the mobility of arsenic.

The main adsorption sites for arsenic in general are those on Fe, Al and Mn-oxyhydroxides. Ferric oxyhydroxides are often the most important surfaces controlling As due to their greater abundance (Smedley and Kinniburgh, 2002). This is expected also for Västerbotten as the primary source is weathering of sulphide minerals generating $Fe^{2+}(aq)$ that will be oxidised forming Fe^{3+} -oxyhydroxides. Nevertheless, Al and Mn oxides may also contribute to the adsorption and immobilisation of As when present in large quantities. At pH below 7.5, Al-oxyhydroxides are as effective as Fe-oxyhydroxides for adsorbing As(V), at higher pH however Fe-oxyhydroxides are more effective in adsorbing As(III) (Smedley and Kinniburgh, 2002). The adsorption of As to Mn, dependent on the charge difference, is negligible in soils with $pH > 4$ as the Mn-

oxyhydroxide surface has a pH_{pzc} (point of zero charge) around 2 and has a net negative charge in soils with pH greater than pH 5.

The Fe^{3+} - oxyhydroxides have a strong binding affinity for As and readily form so-called inner-sphere and outer-sphere surface complexes with arsenic (Dzombak and Morel 1990). In the outer-sphere surface complexation the arsenic oxyanion is adsorbed to the surface by weak electrostatic attractions whereas in inner-sphere surface complexation the anion is adsorbed close to the surface through strong covalent bonds.

Figure 4a illustrates the adsorption site of the metal hydroxide and Figure 4b shows an inner-sphere surface complex. Iron oxyhydroxides have a low solubility and thus precipitate resulting in an immobilisation of the adsorbed As. Figure 3 illustrates the solubility of Fe-oxides as a function of pH. If sulphate is present in quantity and the environment is strongly reducing (table 2, point 3), the mobility of arsenic is low because of the formation of secondary arsenic-rich sulphur minerals such as pyrite (Sracek et al. 2004). In an environment with a $\text{pH} < 9$, carbonate surface are positively charged and thus may adsorb the arsenic oxy-anion in an alkaline soil with pH between 7 and 9, this may reduce arsenic mobility in a limestone aquifer (Sadiq 1995, Smedley and Kinniburgh, 2002).

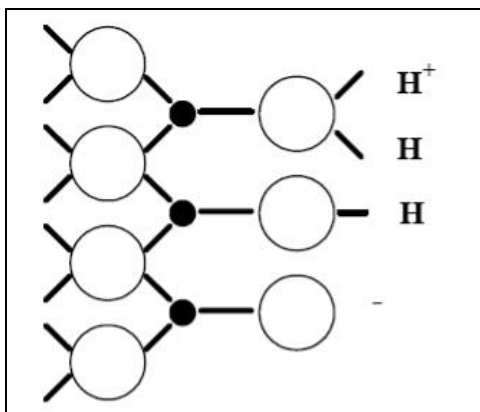


Figure 4a. Surface of a metal oxide, the metal is represented as a solid circle and oxygen as open circles (Welch and Stollenwerk 2003).

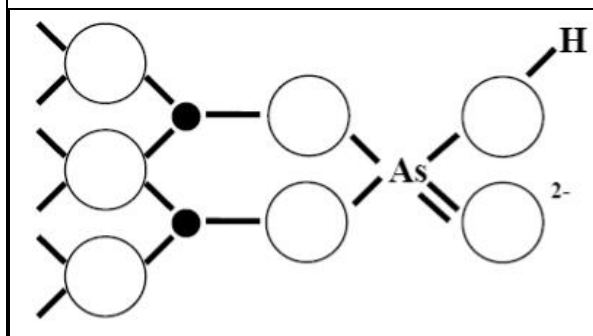


Figure 4b . Inner-sphere complexation of arsenic onto a metal oxide (Drever 1997; Welch and Stollenwerk 2003)

Reductive dissolution and desorption processes are responsible for the mobilisation of arsenic. Arsenic is released through reductive dissolution when Fe-oxyhydroxide is subjected to a reducing environment. Fe will be reduced and as the adsorption media of arsenic is dissolved, arsenic is mobilised.

Both phosphate and carbonate may affect the mobility of arsenic through ion exchange. Phosphate has a stronger affinity for Fe-oxyhydroxide than arsenic and readily replaces arsenic in the solid phase in a competitive ligand exchange reaction (Purnendu and Archana, 2002; Appelo et al. 2002; Reynolds et al. 1999). A crystallisation of the adsorption media will also contribute to the mobilisation of arsenic. Freshly precipitated Fe-oxyhydroxides are amorphous and thus extremely fine grained resulting in a very large specific particle surface area. With time the oxyhydroxides gradually crystallise into more ordered forms such as goethite. The crystallisation reduces the specific particle

surface of the Fe-oxyhydroxide thus decreasing available adsorption sites for arsenic leading to a desorption and mobilisation of adsorbed arsenic (Smedley and Kinniburgh, 2002). In addition to these processes arsenic will be desorbed from the iron-oxyhydroxide surface as a result from a decrease in pH as the affinity for anions will decrease.

3. Study Area

3.1 Location

The study area is located in Västerbotten County in the northern parts of Sweden (Figure 5). The study area includes the municipalities of Norsjö, Malå and Skellefteå covering an area of approximately $150 \times 50 \text{ km}^2$.

The area was selected because of its location in the Skellefte ore-field which consists of volcanic rocks hosting massive sulphide ores rich in As, Zn and Cu (Weiher and Mäki, 1997). The predominant ore minerals in the Skellefte orefield are chalcopyrite (CuFeS_2), galena (PbS), sphalerite (ZnS), pyrite (FeS_2), pyrrhotite (Fe_{1-x}S) and arsenopyrite (FeAsS) (Martin, 2001). Here, elevated concentrations of geogenic arsenic are found both in till and in the groundwater.



Figure 5. Map showing the location of the Västerbotten county and the study area in Sweden.

3.2 Natural occurring arsenic in Västerbotten, Sweden

The primary source of arsenic in water in the Västerbotten region are sulphide minerals associated with metasedimentary rocks such as black schist (SGU, 2005). SGU published in 2003 a map over the arsenic content in till in Västerbotten (Lax, 2003) which showed elevated arsenic concentrations over the Skellefte field (Figure 6). Normally the content of arsenic in bedrock aquifers or soil aquifers is low ($<2 \mu\text{g/l}$) but in the areas of sulphide rich rock and arsenic-rich soils, concentration of naturally occurring arsenic can be significantly higher (SGU, 2005).

When pyrite oxidises in the soil, arsenic is released but can be adsorbed again to iron oxyhydroxides when moving deeper in the soil profile. The structure of the soil profile is therefore of importance. The overburden in the area consists mainly of till that can be divided into three horizons, from the top O, B, and C-horizon (Figure 7).

The O-Horizon is primarily composed of organic matter with fresh litter on top with an increasing degree of decomposition with depth. The decomposition of organic matter releases nutrients and humic acids lowering the pH. Precipitated water percolates through the O-horizon and B-horizon. In the B-horizon an accumulation of fine particles such as organic material and precipitated Fe, Mn and Al -oxides and hydroxides take place creating a denser layer in the soil while the C horizon is a relatively unaffected soil which well reflects the soil parent material.

Within the study area, approx. 5 km north of Norsjö, Jacks et al.(2005) has prior to this study measured arsenic content in each of the three main soil horizons as shown in Figure 7. The studies showed an enrichment of arsenic in the B-Horizon indicating that arsenic is released from the top layer of soil and leached to the B-Horizon where it is assumed to be adsorbed onto Fe-oxyhydroxides.

Within a few hundred meters of these measurements, Jacks et al.(2005) also analysed arsenic concentration in a peat formation and in a small creek to which groundwater from the soil and peat were discharged into (Figure 7). The groundwater in the peat formation had a concentration between 7 and 32 $\mu\text{g/l}$ and the O-Horizon of the peat had an arsenic content of 340 mg/kg. In the stream sediment arsenic content was as high as 4 600 mg/kg due to the precipitation of iron-oxides with adsorbed arsenic. The stream water had an arsenic concentration of 70 $\mu\text{g/l}$. The dominating rock type of the small area in the study is black schist and in the vicinity of the stream a large outcrop of schist was observed with typical reddish-brown Fe-oxide coatings.

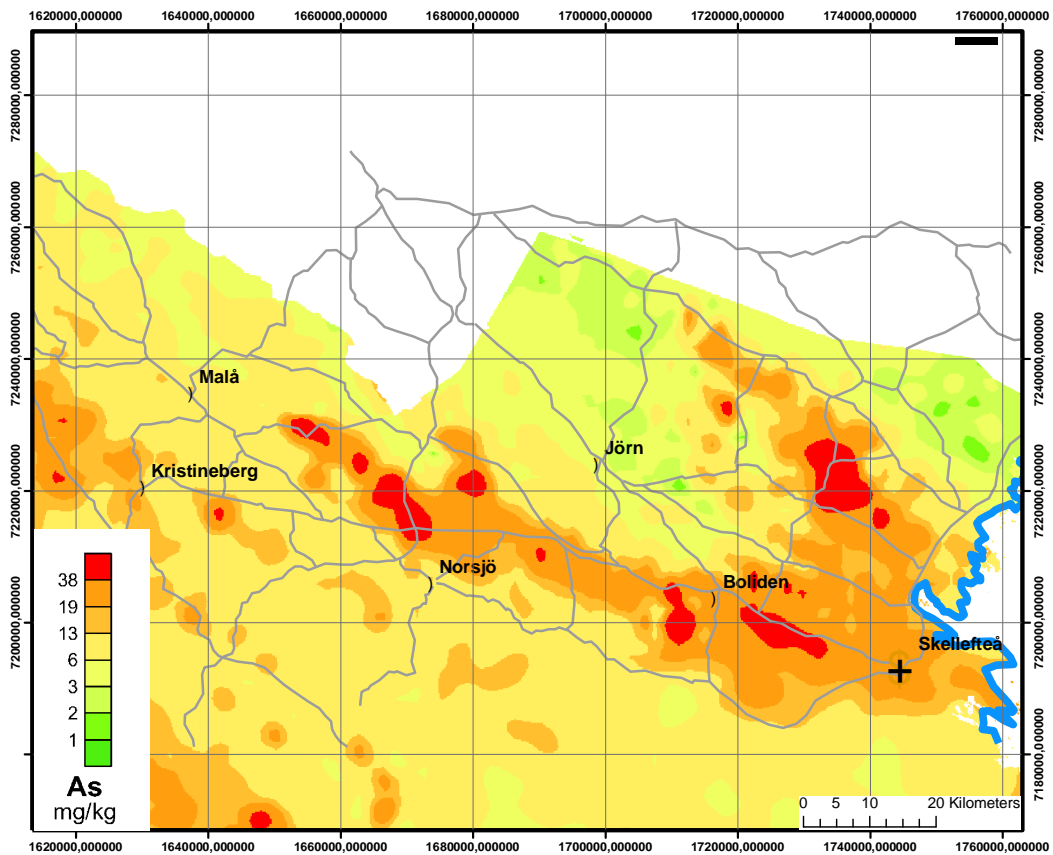


Figure 6. Arsenic content in till in the study area. (SGU, 2005 markgeokemiska kartan i Västerbotten, provided by the County Administration of Västerbotten).

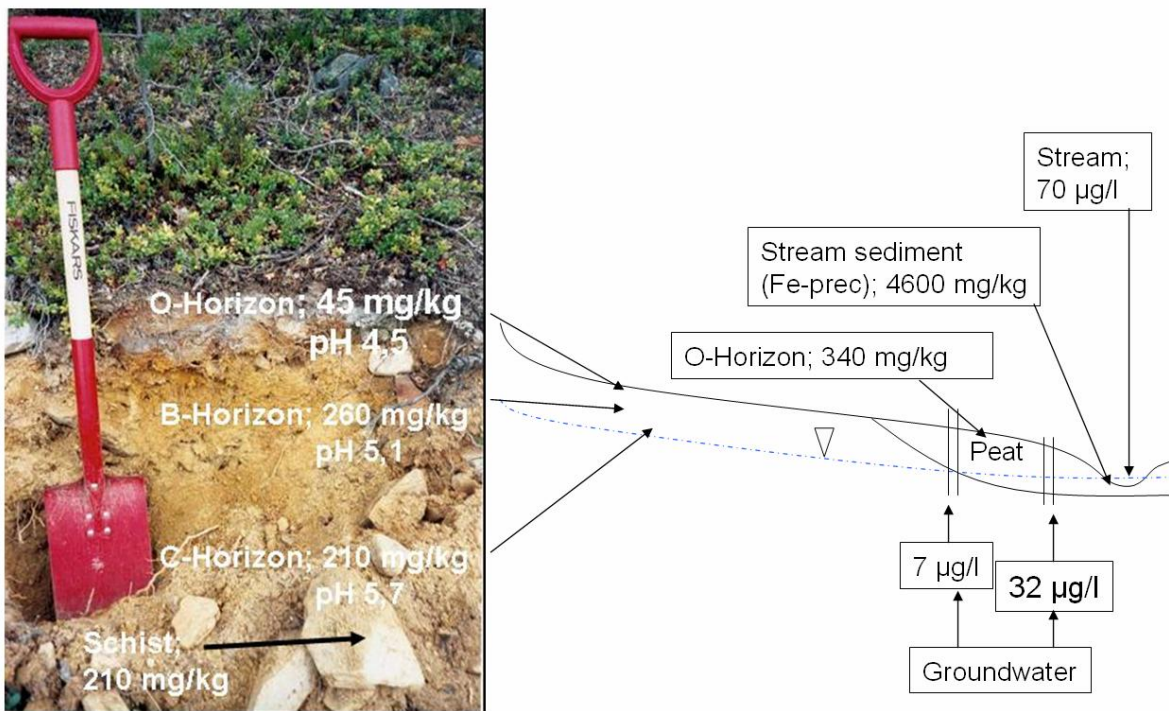


Figure 7. Arsenic content and pH in a well drained site and arsenic concentration around a peat formation (Jacks et al. 2005)

3.3 Geology

3.3.1 Bedrock geology

The geology of Västerbotten County can be divided into three main regions: i) the mountain region in the west on the border to Norway, ii) the eastern marginal zone of the Västerbotten mountains and iii) the Precambrian rocks which cover 70% of the total area of Västerbotten and is a part of the Fennoscandian shield.

The mountain region consists of sedimentary and volcanic rock deposited and formed 400 million years ago in the Iapetus Ocean. When the North American tectonic plate collided with the Scandinavian plate the sedimentary and volcanic rocks together with the underlying Precambrian rock was pushed over the Fennoscandian shield forming the mountains. The predominant rock types are gneisses, mica schist, quartzite and occasional phyllite (SGU map service, 2006). The eastern marginal zone is composed of dark schists, mainly of clay origin. The rocks east of the marginal zone were formed 1800-1900 million years ago and are in the western parts dominated by porphyric granite. To the north and north east acidic to intermediate volcanics and sedimentary rock are more common. The sulphide ores of the Skellefteå field are associated with marine acid volcanics which appear at or close to the boundary between the volcanic rocks and the overlying sedimentary formation (Lundberg, 1980). These rocks have a sulphur content of approximately 0.5%, present as pyrite and arsenopyrite (Jacks et al. 2005). The main bedrock groups over the study area are showed in Figure 8.

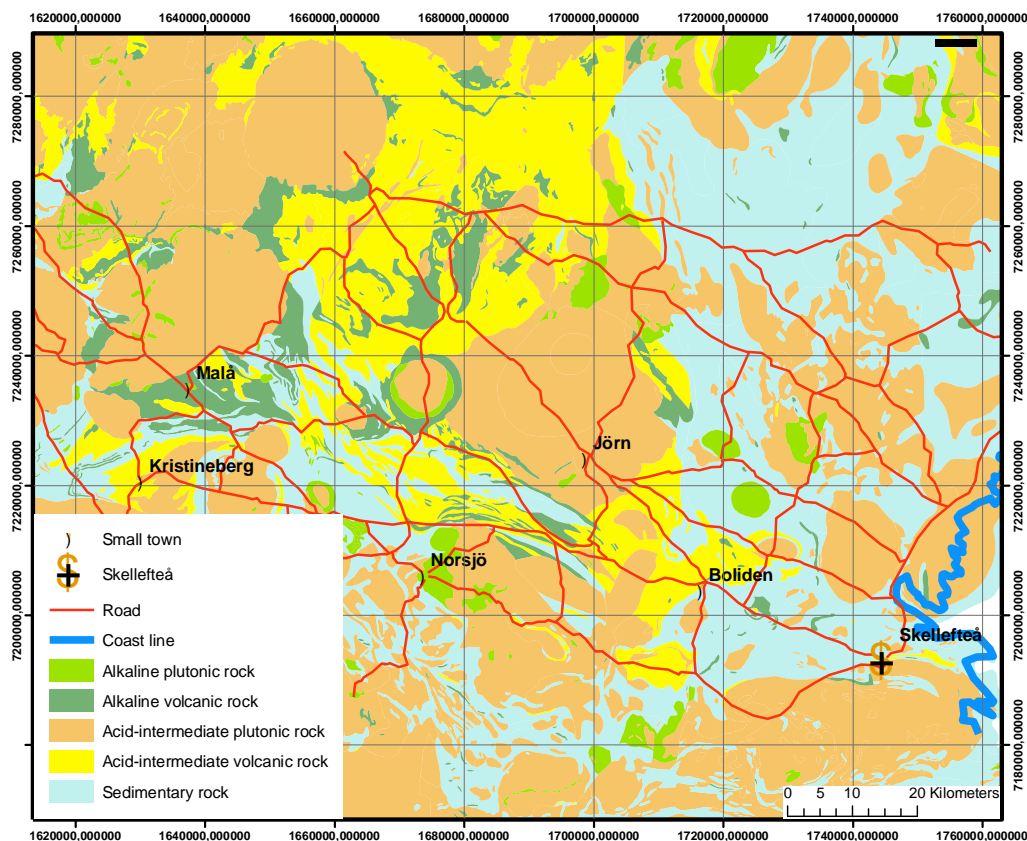


Figure 8. Map showing the main bedrock types in the study area (SGU, provided by the County Administration of Västerbotten).

3.3.2 Quaternary Geology

Within the last 2.4 million years numerous glacial periods have occurred and thick ice sheets have covered the northern parts of Europe. The major part of the overburden in the study area has been deposited during the withdrawal of the last glaciation, approximately 8-9000 years ago. It is composed of material that has been abraded from the local bedrock by the continental ice sheet. As the ice melted, clay, sand, silt and gravel was deposited which resulted in an unsorted soil, till. The transport distance and dilution of the arsenic concentration depends on the soil type. Basal till is deposited closer to the source rock than the top layer of till. The landscape is slightly uneven with glaciofluvial sediments and clay in the valleys and till covering the hills, roughly following the bedrock. The thickness of these alluvial sediments varies with a maximum thickness of about 50 m and a median value of about 15 m with occasional outcrops. Even though till is the dominating soil type, peat formations and more sandy soils can be found in the study area as showed in Figure 9.

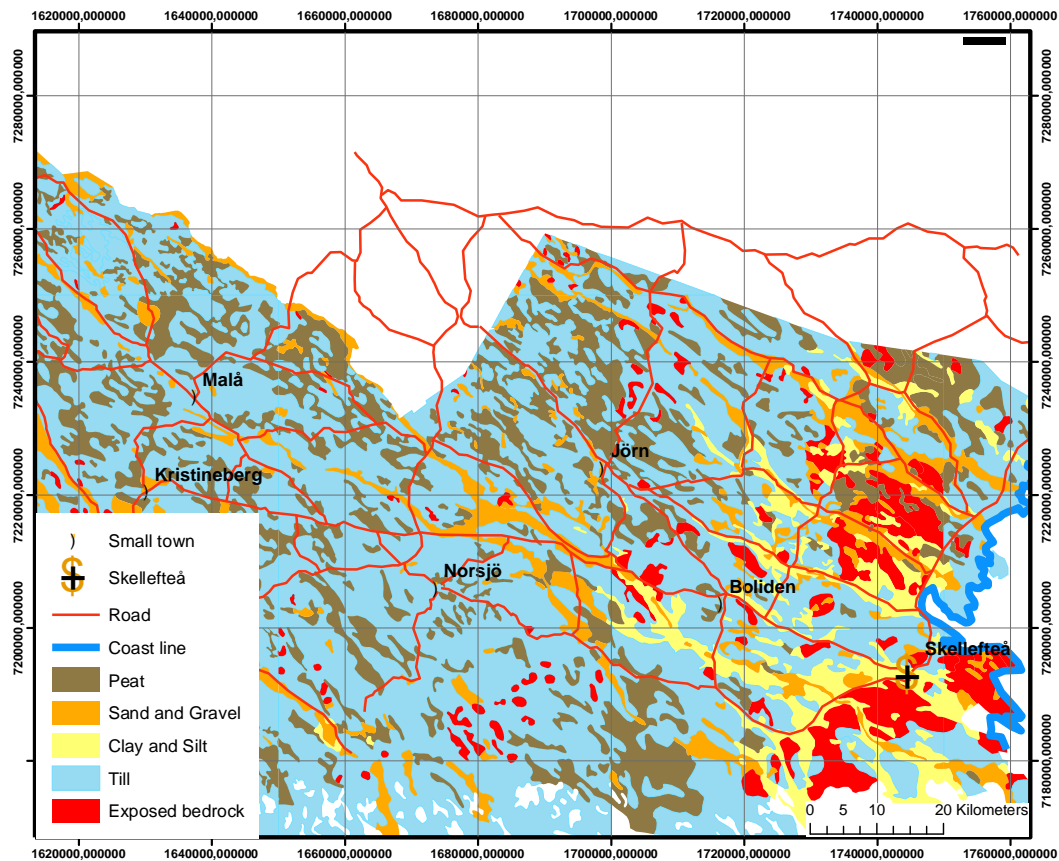


Figure 9. Map showing the different soil types in the study area (SGU, provided by the County Administration of Västerbotten).

3.4 Hydrology and climate

Climate and physical soil characteristics in the area, such as precipitation, overland flow, rate of infiltration, and the groundwater level and its fluctuations affect the mobility and redistribution of arsenic (Bhattacharya et al. 2002).

The climate is classified according to the Köppen's climate system as subarctic with an annual mean temperature of 1 °C (SMHI, 2006) and vegetation dominated by coniferous trees.

The study area has a yearly precipitation of 600 mm and an evapotranspiration rate of about 340 mm/year, thus the total runoff is about 260 mm/year (SMHI, 2006).

3.5 Hydrogeology

The saturated hydraulic conductivity for a sandy-silty glacial till has been determined to $10^{-6} - 10^{-9} \text{ ms}^{-1}$. In a sandy soil, average vertical particle velocity has been measured to 0.1-0.2 m/month (Grip and Rodhe, 1985). The groundwater level in till in Sweden generally lies shallow, within a few meters from the ground surface. The water level in these soils fluctuates during the year with amplitude of a few meters. In the northern part of Sweden low groundwater levels normally occur in late winter due to the long period of snow storage and during hot and dry summers. Peaks occur during autumn and in the spring as in response of snowmelt.

The recharge of groundwater in bedrock strongly depends on the thickness of the overburden. A thick overburden has saturated conditions during a large part of the year thus a continuous recharge takes place whereas a thin overburden might have unsaturated conditions during dry periods where no recharge take place. The groundwater flow from soil to the underlying rock at saturated conditions depends on the extent of fractures and fracture zones in the bedrock. The fractures in crystalline rock have an extreme high spatial variation and it is therefore difficult to predict how the fractures are situated in the rock. Due to the complexity of the water leading fractures, wells adjacent to each other might differ significantly in water chemistry.

4. Hypotheses

Based on the literature study two possible processes are here proposed responsible for the arsenic in drilled wells in Västerbotten. The hypotheses are illustrated as a conceptual model in Figure 10.

- I. This hypothesis views oxidation of sulphide minerals in till as the main source of arsenic. The oxidation results in arsenic anions that under the prevailing oxidizing environment form insoluble complexes with iron-oxyhydroxide. As a result of a rising water table the complex is later subjected to a reducing environment leading to mobilisation of both the arsenic anion and the iron cation. In a reducing environment, the anion stays in the water phase and may enter the network of fractures common in crystalline rocks. This hypothesis implies a low redox potential of water with high arsenic concentration.
- II. This hypothesis suggests the bedrock as an important source of arsenic in drilled wells. As dissolved oxygen or NO_3^- enters the fractures pyrite oxidation takes place releasing arsenic into the groundwater. No or little adsorption to iron occurs as the environment is too reducing to oxidize Fe^{2+} .

In hypothesis I a linear correlation between arsenic and iron concentration would be expected. Also the sulphate concentration and redox potential should be low.

In water typical for hypothesis II the arsenic concentration would correlate linear to sulphate. Nitrate concentration should be low and the redox potential should be moderate to low.

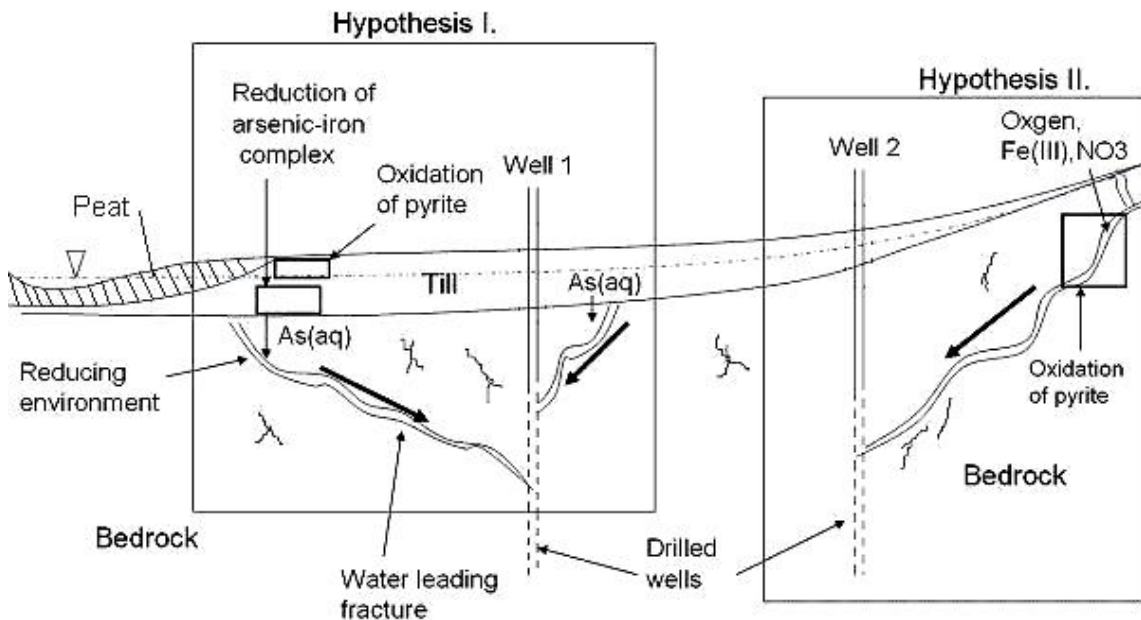


Figure 10. Conceptual model for hypothesis I and II.

5. Material and Method

5.1 Arsenic in groundwater from Västerbotten: Two hypotheses

In order to delineate geological conditions favourable for, and to understand the processes responsible for, high levels of geogenic arsenic in the groundwater of Västerbotten two main hypotheses describing the mobilisation pathway of As into the groundwater were developed. This was done in order to design the field work and laboratory analyses as well as providing groundwork for interpreting the results from the field, laboratory and GIS-work.

5.2 Field work

The field work was carried out in July and October of 2006. The work was initiated by finding suitable drilled wells for sampling through the well archive on the website of the SGU. In addition to these, other drilled and dug wells were sampled as the SGU database for existing drilled wells were incomplete with too few wells registered for this investigation. The additional wells were chosen randomly within the study area to give a good coverage of the Skellefte field.

5.2.1 Water samples

Most of the water samples were taken from water taps of a household.

Some households had filters for reducing the concentration of iron and in these cases the water was sampled before the filter. Some dug wells were sampled at the spring and not the water tap. A total of 44 wells were tested and from each well two samples were taken.

- One sample, for anion analysis, was filtered with a disposable 0.45 µm-Sartorius filter and stored in a 50 ml bottle.
- One sample, for cation and trace element determination, was filtered with a 0.45µm Sartorius filter and then acidified with suprapur HNO₃ (14 M).
- Arsenic speciation was performed with disposable cartridges (MetalSoft Center, PA) in the field. The cartridge allows only As(III) to pass through. A 25 ml bottle was filled with filtered and arsenic speciated water and then acidified.

5.2.2 Field parameters

Measurements of pH, conductivity, redox-potential (Eh), and water temperature were made on site. The pH electrode was calibrated once a day with standards of pH 4 and pH 7. The redoxpotential was measured with a voltmeter with an Ag/AgCl reference-electrode and a Pt-measuring electrode. This was connected to a "flow through cell" which was connected to the water tap with a rubber tube. The cell was used in order to decrease the contact between water and air before measurement. The same device also measured conductivity. On the second trip the redox potential and conductivity was not measured at all because of difficulties getting reliable values.

The position of each well was obtained by using a GPS receiver.

5.3 Laboratory Analysis

The water samples were analyzed at ACME ANALYTICAL LABORATORIES in Vancouver and at the Department of Land and Water Resources Engineering at the Royal Institute of Technology (KTH). The results from the analysis are presented in appendix 11.1.

5.3.1 Metals

A complete metal analysis was conducted at ACME ANALYTICAL LABORATORIES.

All the water samples were analyzed by ICP-MS and diluted to below 0.1% total dissolved solid before analysis.

4.3.2 Alkalinity, Nitrate, Sulphate and Chloride

The concentration of NO_3^- , SO_4^{2-} and Cl^- were analysed by ion chromatography and HCO_3^- was analysed by titration at the Department of Land and Water Resources Engineering at KTH.

5.4 Additional Data

A total of 183 additional sample data was received from three different municipalities located within the area of this study.

- Norsjö (appendix 10.2): Data for 77 samples was received. The samples contained coordinates for each sample and analysis of trace elements such as: As, B, Ba, Be, Pb, Cd, Co, Cu, Cr, Li, Mn, Mo, Ni, Se, Ag, Sr, Tl, U, V, Zn. Unfortunately important data such as Fe, SO_4 , HCO_3 had not been analysed.

- Malå (appendix 10.3): 9 complete analyses was received, however the coordinates for each sample was not given. Instead of the coordinates the property unit designation was given from where the samples were taken. This was converted to coordinates by using the PropertySearch function on Lantmäteriets webpage (www.lm.se). When converting into coordinates the coordinates given by PropertySearch might not coincide with the actual position of the well as the point given is the coordinates for the centre of the property. Two of the samples did not contain any information whether the well was a dug or a tube well. These wells were therefore excluded.

- Skellefteå (appendix 10.4): Data for 97 samples were received containing only As-analysis. Of these, 12 samples contained sufficient information regarding type of well and property unit designation. The rest of the samples were excluded. Due to the lack of data, these samples could only be used for evaluating the spatial distribution of arsenic in arcGIS. Coordinates was obtained by using the PropertySearch (www.lm.se).

5.5 Treatment of data

The treatment of data is described in the following chapter. The sequence of work is also illustrated in Figure 11.

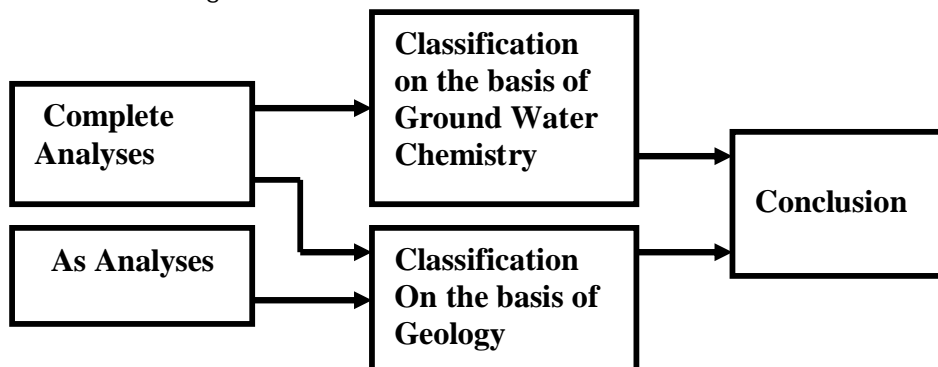


Figure 11. The sequence of work.

5.5.1 Redox classification

Two redox classifications were made. The samples obtained within this project and the samples received from Malå were classified into five different redox-classes by using Fe, Mn and SO₄ as indicator parameters, following the table in appendix 10.5 published by the Swedish Environmental Protection Agency (2000). The redox classes are ordered with decreasing Eh from redox-class 1-4. Redox-class 5 is mixed waters. The Norsjö samples did not contain data regarding SO₄ and Fe and was therefore put into only two redox-classes based on the concentration of Mn following the Mn column in appendix 10.5.

5.5.2 Arsenic versus analysed parameters

Aquachem 4.0 was used to determine water type through a piper diagram. The program was also used to find correlations between different parameters for samples with a complete analysis. Parameters below the detection limit were set to half the detection limit instead of zero. This was done in order for Aquachem to take these parameters in account also.

5.5.3 GIS

The samples was imported in ArcGIS and plotted on a bedrock map, a quaternary geologic map and a map showing the spatial distribution of arsenic in till. ArcGIS was then used to find correlation between elevated arsenic concentrations and the geologic environment for each of the samples.

6. Results

In this chapter results from the GIS work, laboratory results and field results are presented. Dug well (DW) data will be kept separate from tube well (TW) data in order to illuminate differences between the two types. The TW will also be divided into high arsenic wells ($>5\mu\text{g/l}$) and low arsenic wells ($< 5\mu\text{g/l}$). Detailed data of analyzed parameters and samples received from municipalities is presented in appendix 10.1 to 10.4.

6.1 Geology

The wells are situated in five different main bedrock categories:

- Acid to intermediate plutonic rock
- Acid to intermediate volcanic rock
- Alkaline volcanic rock
- Alkaline plutonic rock
- Sedimentary rock

Of 60 tube wells, 30 was situated in acid to intermediate plutonic rock, 12 in acid to intermediate volcanic rock, 8 in alkaline volcanic rock and 6 in sedimentary rock. No tube wells were installed in alkaline plutonic rock. The distributions of the wells are presented on a geological map in appendix 10.8.

6.1.1 Arsenic, major ions and geology

Of all the 140 arsenic analyses, 24 wells had an arsenic concentration above the national food administrations guideline value of $10\ \mu\text{g/l}$. The concentration varied from below detection limit of $0.5\ \mu\text{g/l}$ to $300\ \mu\text{g/l}$ with a median value of $0.87\ \mu\text{g/l}$ and a mean value of $14.2\ \mu\text{g/l}$. Of all the received arsenic analyses and the analyses from the wells tested within this study, 35 contained arsenic concentrations above $5\ \mu\text{g/l}$. Of these are 21 tube wells. The 21 high arsenic tube wells are distributed in four different bedrock groups:

- 4 in Alkaline volcanic
- 4 in Acid-intermediate volcanic
- 12 in Acid-intermediate plutonic
- 1 in Sedimentary rock

The high arsenic wells are scattered over the study area and no spatial trend can be seen toward a specific area. The distribution of arsenic between the four geologic groups are presented in Table 3.

This shows that the median arsenic concentration is higher in alkaline volcanic rock than the other three groups with a mean concentration of $46.8\ \mu\text{g/l}$ and a median value of $7.9\ \mu\text{g/l}$. Further the results indicate that wells situated in sedimentary rock contain the lowest arsenic concentrations.

Table 3. Statistical data for the distribution of arsenic between bedrock groups for TW.

Bedrock group	Number of wells	Min µg/l	Max µg/l	Mean µg/l	Median µg/l
Acid-intermediate volcanic rock	12	0.05	300	52	3
Alkaline volcanic rock	8	0.1	270	46.8	7.9
Acid-intermediate plutonic rock	30	0.05	201.2	13.4	2.1
Alkaline plutonic rock	0				
Sedimentary rock	10	0.25	22	3.2	0.55

6.2 Redox classification

Of the 28 tube wells that were classified into redox classes, 13 were a mixture of different ground waters (redox class 5, appendix 11.5). The second largest group was redox-class 1 with 8 tube wells followed by 3 wells of redox-class 4, 2 wells in class 3 and 2 wells in redox-class 2. Of the 9 tube wells exceeding an arsenic concentration of 5 µg/l, 3 were classified into redox-class 5, 4 in redox-class 1 and 2 in redox class 2. Redox class 3 and 4 had one well in each.

There were in total 23 dug wells that were classified according to the Swedish protection agency's method. The majority of these, 11 wells, were classified as mixed waters. Statistical data have been calculated and is presented in Table 4. The lowest median and mean value of arsenic in TW is found in mixed waters according to the table. There are however too few samples in the other classes to draw any certain conclusions from this. For the dug wells there is little difference in arsenic concentration between the classes, but again no reliable conclusions can be drawn due to the low number of samples. The results from the redox classification for each well are also presented in appendix 11.6.

Table 4. Statistical data for arsenic concentration in the five different redox-classes, divided into dug and tube wells.

TW:					
Redox class	Number of wells	Min As conc. µg/l ⁻¹	Max As conc. µg/l ⁻¹	Mean µg/l ⁻¹	Median µg/l ⁻¹
1 (High Eh)	8	0.1	201.1	31.3	1.3
2 (Moderately high Eh)	2	37.6	178.4	108	108
3 (Low Eh)	2	3	24.2	13.6	13.6
4 Very low Eh	3	2.2	293	99.5	3.3
5 Mixed water	10	0.25	12.9	2.95	0.375

DW:

Redox class	Number of wells	Min As conc. $\mu\text{g l}^{-1}$	Max As conc. $\mu\text{g l}^{-1}$	Mean $\mu\text{g l}^{-1}$	Median $\mu\text{g l}^{-1}$
1 (High Eh)	9	0.2	6.1	1.6	0.25
2 (Moderately high Eh)	0				
3 (Low Eh)	1	1.3	1.3	1.3	1.3
4 Very low Eh	4	1.2	3.3	2.3	2.4
5 Mixed water	11	0.25	4.2	1.9	2

Of the 26 TW from Norsjö which were classified into two classes, 15 were classified as redox-class 1, of these 5 had an arsenic conc. exceeding 5 $\mu\text{g/l}$.

The Norsjö samples contained 60 DW of which 55 were classified into redox class 1, 7 had arsenic conc. exceeding 5 $\mu\text{g/l}$. Basic statistical data have been calculated and is presented in Table 5. The median arsenic concentration for TW in redox-class 2 is 9.6 $\mu\text{g/l}$ whereas the median value for redox-class 1 is 2,8 $\mu\text{g/l}$. For DW redox-class 1 has got a median value of 0.5 $\mu\text{g/l}$ and for redox-class 2 the median lies at 1.8 $\mu\text{g/l}$. The redox classification for each well is presented in appendix 10.7.

Table 5. Statistical data for arsenic concentration for DW and TW in the two redox-classes based on Mn concentration. for tube and dug wells.

TW:

Redox-class	Number of wells	Min As conc. $\mu\text{g l}^{-1}$	Max As conc. $\mu\text{g l}^{-1}$	Mean $\mu\text{g l}^{-1}$	Median $\mu\text{g l}^{-1}$
1	11	0.3	46	14.17	2.8
2	9	3.1	270	59.7	9.6

DW:

Redox-class	Number of wells	Min As conc. $\mu\text{g l}^{-1}$	Max As conc. $\mu\text{g l}^{-1}$	Mean $\mu\text{g l}^{-1}$	Median $\mu\text{g l}^{-1}$
1	55	0.025	60	3.41	0.5
2	5	0.6	15	5.64	1.8

6.3 Arsenic speciation

Arsenic(III) speciation was performed on samples VB-5, VB-22, VB-56, VB-57, VB-59, VB-60 and VB-61. These wells all had a total arsenic concentration exceeding 5 $\mu\text{g/l}$. The result and the As(III)/As(tot) ratio is presented in Table 6. The speciation showed that arsenic in well VB-5 and VB-57 is present only as As(III) which indicate reducing conditions. The proportion As(III) in well VB-22 was 2 % making As(V) the dominating species indicating an oxic environment. The As(III)/As(tot) ratio in the rest of the wells ranged from 10 to 24 % indicating a slightly reducing environment.

Table 6. Distribution of arsenic species in wells with As(tot) > 5µg/l.

Sample ID	As(tot) µg/l	As(III) µg/l	As(III)/As(tot)
VB-61	201.2	49.2	0.24
VB-5	293	318.9	1.09
VB-22	12.9	0.25	0.02
VB-56	6.5	.8	0.12
VB-57	24.2	24.3	1.00
VB-59	6.1	1.2	0.20
VB-60	178.4	18.1	0.1

6.4 Water type

The most common water type is Ca-HCO₃ (9 of 35, 25.7%), this group is followed by NO₃-Ca-HCO₃ type waters (17.1%) and Ca-Mg-HCO₃ type waters (14.3%). As the Piper plot in Figure 12 shows there is no or little difference in water type between dug and tube wells. The major ions of the sampled ground waters are dominated by Ca²⁺ and HCO₃⁻, although NO₃⁻ is also present in quantity.

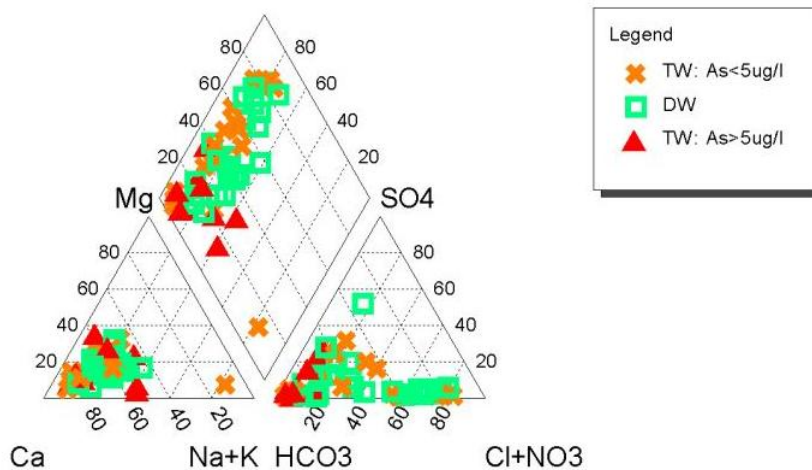


Figure 12. Piper plot with major ions represented.

6.5 Redox potential(Eh)

The Eh ranged between a maximum of 295 mV and a minimum of -120 mV with a median value of 123 mV. The data of the Eh is incomplete and only comprise the VB-5 to VB-60 samples due to difficulties getting reliable values. The Eh seldom stabilized, which is why the measurements were aborted after approximately 5 minutes. The reason for the unreliable values might be that the potential in a natural water is a mixed potential of the individual redox couples which seldom is at equilibrium. It is also difficult to eliminate contact between atmospheric oxygen and the water, even though a “flow through cell” is being used. It is particularly important to eliminate the water and air contact when measuring Eh of very reducing waters (Back, 2001).

6.6 pH and Alkalinity

The pH from 41 wells are grouped according to type of well and are presented in Figure , showing max, min, the 25 and 75 percentile and the median value. The pH of dug wells are generally lower than the pH of tube wells. This is due to the larger weathering rate in the top soil layer. Also a higher proportion of groundwater influenced by rain (pH 5) may reduce the pH in dug wells. The dry summer with extremely low groundwater levels,

followed by a recharge may have affected the result in the wells tested in October. The low ground water level may have lead to aeration of the deeper soil layers resulting in a larger weathering rate than usual.

The alkalinity for DW, median value of 21.7 mg/l, is lower than the alkalinity for TW. Between the two TW groups the alkalinity in high arsenic wells were higher than in wells with a low arsenic concentration with a median value of 70.8 mg/l compared to 51.4 mg/l (figure 14).

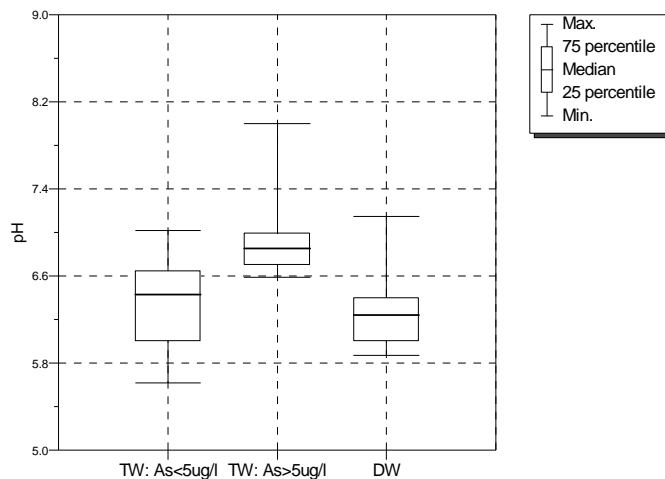


Figure 13. Box plot of pH for dug and tube wells.

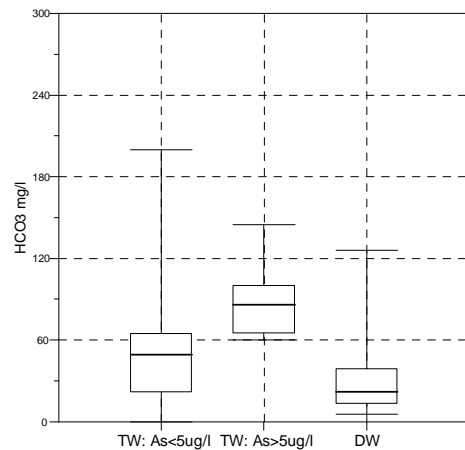


Figure 14. Box plot of alkalinity for dug and tube wells.

6.7 Anions and Cations

Nitrate and sulphate are presented as box plots in figure 15a-b. Nitrate concentrations are generally low ranging from below the detection limit of 1 mg/l to 128 mg/l, with a median value of 0.86 mg/l (because nitrate b.d.l. were set to 0.5 mg/l). The guide line value for nitrate in potable water is set by the Swedish Food Administration (SLV) at 4.4 mg/l and the acceptable limit for public water supply is set at 22 mg/l.

The box plot in figure 15a shows that the high arsenic wells have lower nitrate concentration than the other two groups with a median value of 0.5 mg/l. No significant difference in nitrate concentration can be seen between the non arsenic contaminated TW and the DW.

The sulphate concentration ranges from 0.35 mg/l to 32 mg/l with a median value of 2.85 mg/l. Figure 15b shows no clear difference between the two TW groups but shows a lower sulphate concentration in DW than in TW.

The two most abundant cations were calcium and sodium.

Calcium ranged from 2.3 mg/l to 56 mg/l and sodium ranged from 1.5 mg/l to 31 mg/l. The calcium concentration was within the expected range based on the Swedish Environmental Protection Agency's (SEPA) Environmental Quality Criteria.

The sampled wells showed a wide span in iron concentration. Of 35 wells, 12 had a concentration below the detection limit of 10 µg/l. The median value of iron was 32 µg/l and the mean value was 223 µg/l. These values can be compared to SLV's guide line value for iron which is set at 50 µg/l.

Three wells had a manganese concentration below the detection limit of 2 µg/l. The maximum value was 832 µg/l in well VB-54 and the median value was 10.3 mg/l. The guide line value is set at 20 µg/l with an acceptable limit for public water supply at 50 µg/l.

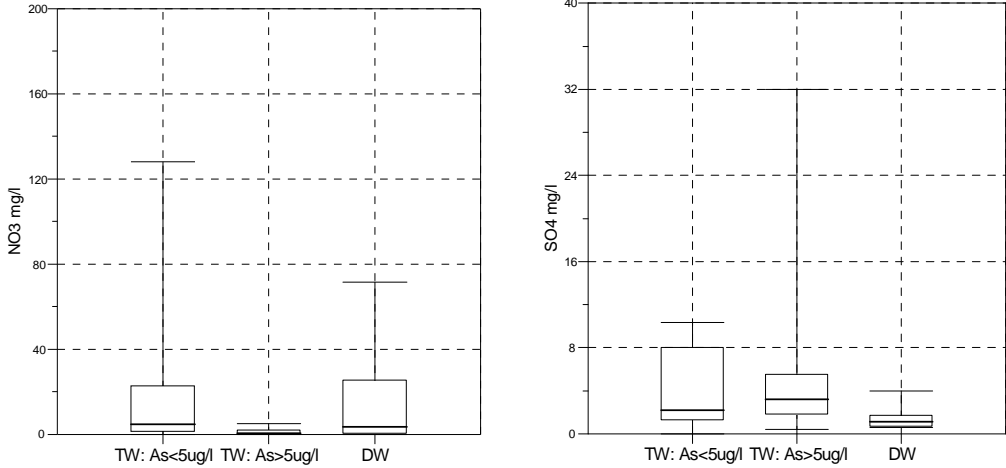


Figure 15. a) Box plot of nitrate for three different groups. b) Box blot of sulphate for three different groups.

7. Discussion

7.1 Limitations of study

The number of well in this study is limited. Some of the wells are situated close to the border between two rock types and as the complex system of fractures may connect adjacent bedrock types it is difficult to interpret the geological results. Also in some bedrock groups only a few well were installed which makes the statistical analyse uncertain. It would be desirable with more wells in each bedrock group in order to draw more certain conclusions.

7.2 Correlation between arsenic and laboratory parameters

Arsenic has been plotted versus several parameters in order to find correlations. The plots, which will be presented in this section, are a mean to interpret the groundwater analysis.

Arsenic was plotted versus nitrate in Figure 16 and as the box plot in Figure 15a indicated that high arsenic concentrations are connected to low nitrate concentrations. The correlation is very clear and shows that none of the high arsenic groundwaters have a nitrate concentration exceeding 5.5 mg/l.

Arsenic was also plotted against sulphate in Figure 17a-b. Figure 17a shows that the two wells, VB-5 and VB-60, with 293 µg/l and 178.4 µg/l arsenic, respectively, have low concentration of sulphate while well M2 deviates from the others with a sulphate concentration of 32 mg/l. This value can be compared to the median and 95th percentile from SEPA's data base over sulphate concentration in groundwater in igneous rock in the coastal region of northern Sweden which is approximately 22 mg/l and 52 mg/l respectively. In Figure 17b, the extreme values of arsenic and sulphate have been removed but still no clear correlation can be determined from these two plots.

There is a slight linear correlation between arsenic and alkalinity in tube wells as Figure 20b shows which was also indicated in Figure 16. Well M7 have slightly higher alkalinity than the rest of the samples but are still within the expected range, based on the environmental quality criteria written by SEPA. Figures 19 and 20 show there are no correlation between arsenic and the two most abundant cations, calcium and sodium.

Arsenic was also plotted versus the potassium concentration in Figure 23 in order to look for influences from fertilisers. Well VB-57 has got higher concentration of potassium than the other high arsenic wells with a concentration of 7.8 mg/l.

In both hypotheses, iron would theoretically be correlated linearly to the arsenic concentration. The real relation between arsenic and iron is plotted in Figure 21 and it shows that only well VB-5 has got both high As and Fe concentration. Except for VB-5, the different wells are scattered randomly in the plot.

Iron was also plotted versus nitrate in Figure 22. The plot shows that high iron concentration correlates with low nitrate concentration.

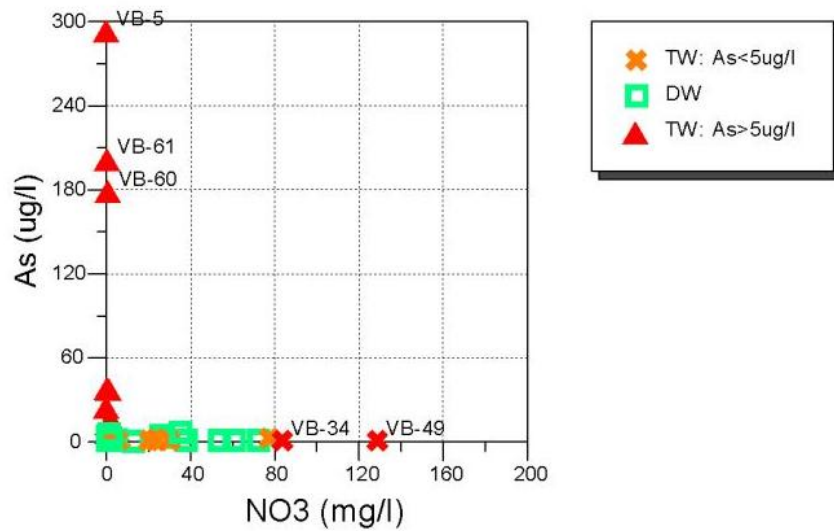


Figure 16. Scatter plot of arsenic versus nitrate from dug wells (DW) and tube wells (TW).

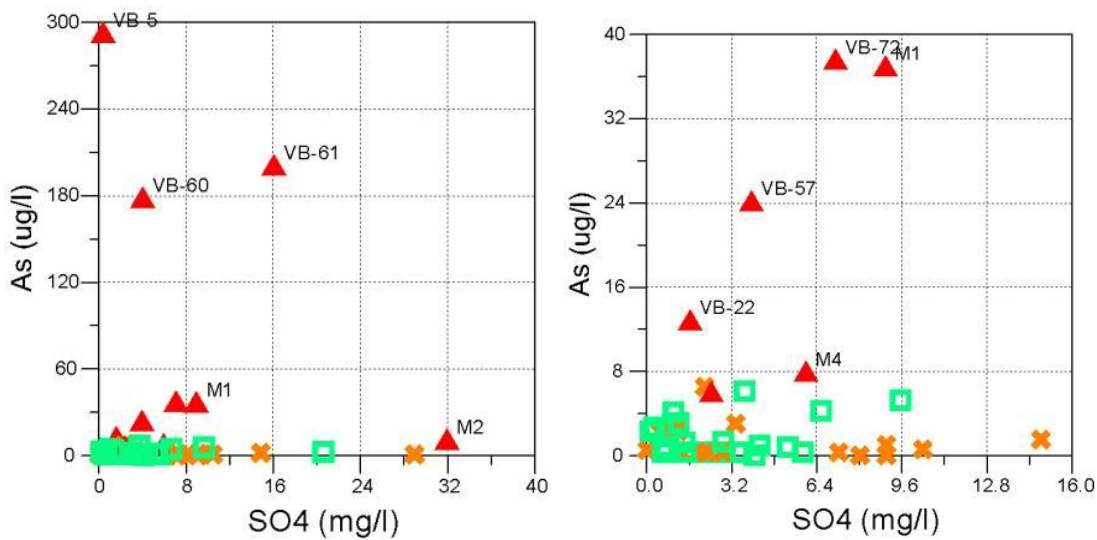


Figure 17. a) Arsenic versus sulphate from dug wells (DW) and tube wells (TW). b) Arsenic versus sulphate with the extreme values on both axes removed.

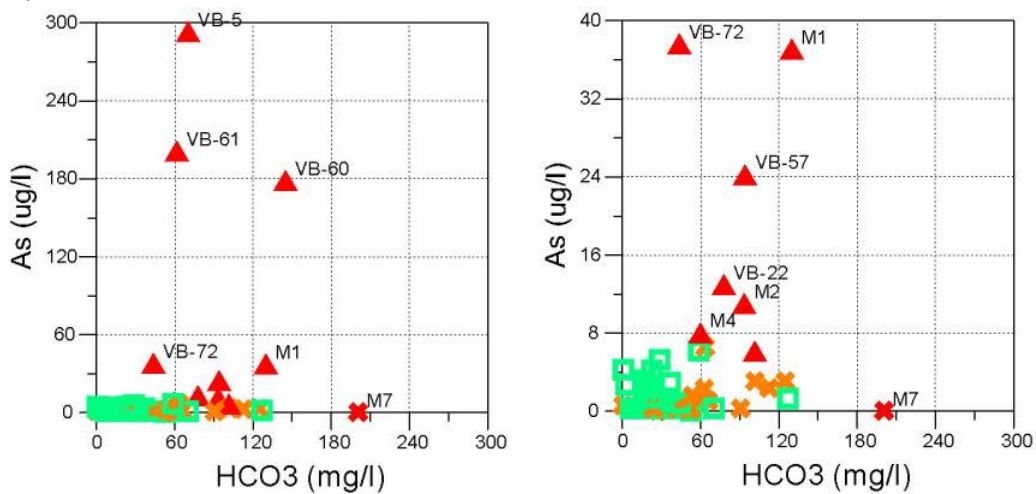


Figure 18. a) Arsenic versus alkalinity from dug wells (DW) and tube wells (TW). b) Arsenic versus alkalinity with the extreme values on both axes removed.

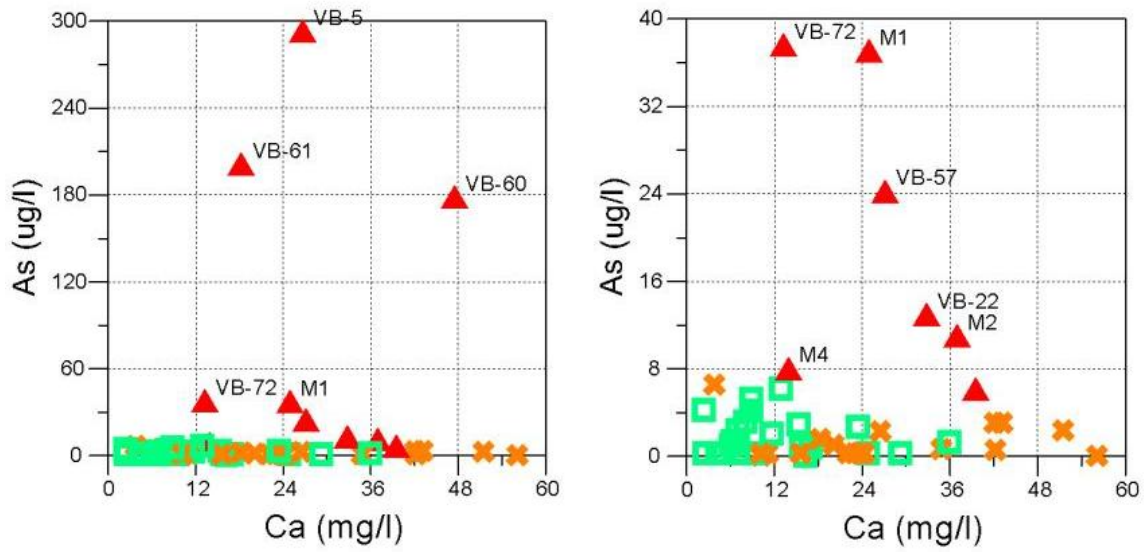


Figure 19. a) Arsenic versus calcium from dug wells (DW) and tube wells (TW). b) Arsenic versus calcium with the two extreme values of arsenic removed.

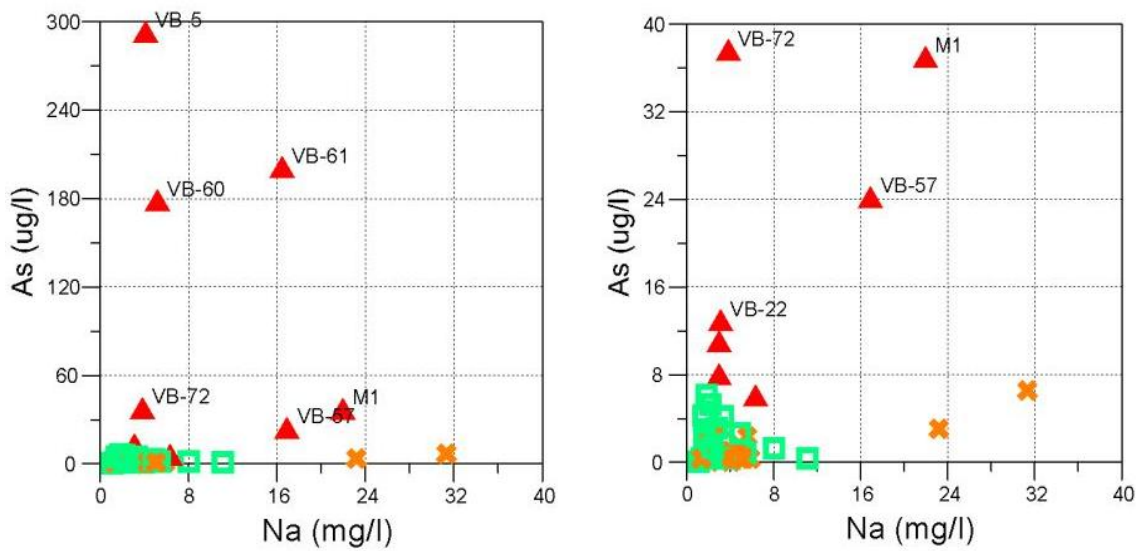


Figure 20. a) Arsenic versus sodium. b) Arsenic versus sodium with the two extreme values of arsenic removed.

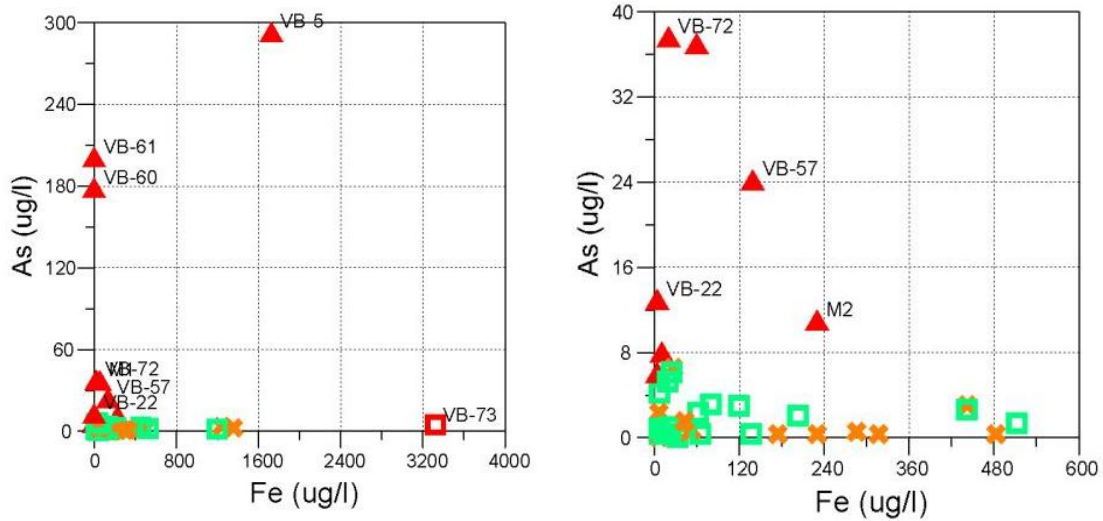


Figure 21. a) Arsenic versus iron. b) Arsenic versus iron with the two extreme values of arsenic and iron removed.

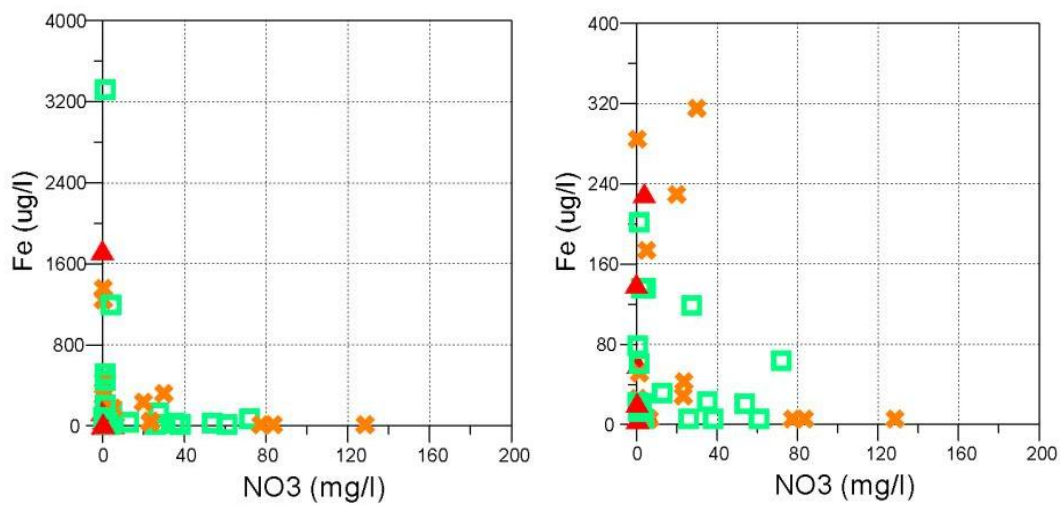


Figure 22. a) Iron versus nitrate. b) Iron versus nitrate with the three highest values of iron removed.

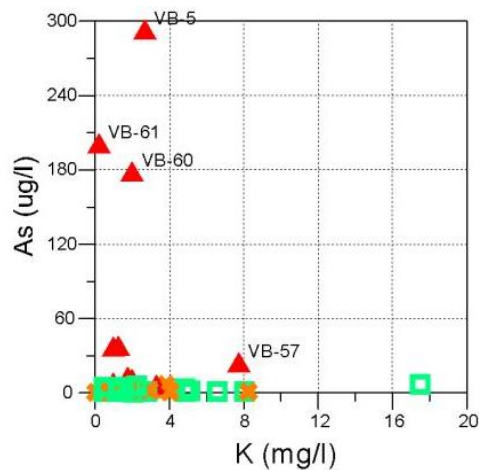


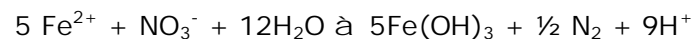
Figure 23. Arsenic versus potassium.

7.3 Apparent relation between groundwater parameters and arsenic concentration

The difference in pH between the two TW, high and low arsenic, types is in accordance with the decreased adsorption of arsenic to metal hydroxides at higher pH. The difference in alkalinity can be explained by a longer contact time between water and bedrock for high arsenic wells, which also results in a higher pH.

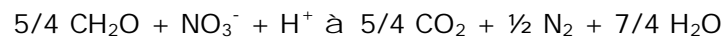
When nitrate is present it may lead to the oxidation of Fe^{2+} resulting in the formation of iron-oxyhydroxides to which arsenic is adsorbed thus arsenic and iron concentrations are low in high nitrate waters. Tracer tests, with the injection of nitrate and arsenic, performed by Höhn et al. (2006) showed that the oxidation of Fe^{2+} and the removal of As(V) occurred rapidly suggesting that the ferric iron was oxidised by nitrate reduction.

The overall chemical reaction for Fe^{2+} oxidation by nitrate reduction can be represented by the following reaction (Pedersen et al. 1990):



As the reaction shows 1 mole nitrate oxidises 5 moles iron. The database of SEPA shows a median nitrate value of 0.84 mg/l in drilled wells in the area which agrees with the median nitrate concentration from this study of 0.86 mg/l. Therefore most of the nitrate concentration in these wells is assumed to originate from natural processes.

Nitrate can also be reduced by microbial oxidation of organic matter through the following reaction:



This shows that 1 mg/l CH_2O (organic matter) can reduce 1.65 mg/l nitrate (Pedersen et al. 1990). In the groundwater of the study area, a concentration of a couple of mg/l of dissolved organic matter can be expected (though not analysed). Thus there would not be enough dissolved organic matter to reduce the total amount of nitrate in the arsenic free waters of this study. Therefore it is assumed that ferrous iron is the primary reductant for nitrate in the area which also explains the nitrogen-arsenic relationship in Figure 16 and the nitrogen-iron relationship in Figure 22. Also, as the sulphide minerals are oxidized an increase in the sulphate concentration would be expected. For the tube wells, for which the recharge area has low redox potential, arsenic concentration can be high even though the sulphate concentration is low, as sulphate was flushed out when the primary mineral was first oxidized. In these areas the reduction of secondary arsenic rich Fe-oxyhydroxide should result only in an increase in Fe^{2+} concentration and not SO_4^{2-} .

7.4 Classification of high arsenic tube wells into hypotheses I and II.

As written in section 5.7 the high arsenic wells all have low nitrate concentrations, therefore this will not be commented for the wells described below.

7.4.1 Hypothesis I wells

VB-5:

The highest arsenic concentration, 293 µg/l, is found in VB-5. The well is situated in acid-intermediate volcanic rock. This well has got the highest iron concentration (1.75 mg/l) of all samples. The sulphate concentration is low and redox classification according to the redox-table in appendix 11.5 put the well in redox-class 4, that is to say water with very low redox potential. Arsenic speciation also indicated a reducing water as arsenic was present only as As(III). The redox potential was measured on site in this well and even though the results from the measurement was considered unreliable it showed a very low redox potential (-120 mV). The potential is by far the lowest value from these measurements. From these results the conclusion can be drawn that the processes leading to high arsenic concentration in this well works accordingly with hypothesis I.

VB-57:

The well has an arsenic concentration of 24.2 µg/l and is situated in acid to intermediate plutonic rock close to Drängsmark next to a cultivated field. Sulphate concentration is low (4 mg/l). The alkalinity and iron concentration was 95 mg/l and 140 µg/l respectively. The iron, sulphate and manganese concentration for this well puts it into redox class 3 (low redox potential), this was also confirmed by the arsenic speciation as arsenic was present solely as As(III). The potassium concentration for the samples was plotted in Figure 22 and this showed that well VB-57 had a relatively high potassium concentration (7.8 mg/l). An explanation could be the use of an excess amount of Nitrogen-Phosphorus–Potassium type fertilizers on the near by field resulting in elevated K-concentrations in the ground water. If the high K-concentration is a result from an anthropogenic source as fertilizers the phosphate ions will also contribute to the elevated arsenic concentration through ions exchange. From the low redox potential the conclusion that this is a hypothesis I well is drawn.

7.4.2 Hypothesis II wells

VB-61:

The arsenic concentration in this well was 201 µg/l. The iron, manganese and sulphate concentration put the well into redox class 1. The low iron concentration, the relatively high sulphate concentration and the result from redox classification makes this well a probable hypothesis II well

M1:

Well M1 had an arsenic concentration of 32 µg/l. It is drilled in acid-intermediate plutonic rock. The sulphate concentration, 9.5 mg/l, is among the highest of the samples but according to the Swedish Environmental Protection Agency (SEPA) this is about average for the area. Iron concentration was 60 µg/l but even though this is twice the median value of iron it is still a relatively low concentration. The alkalinity is high, 130 µg/l, and the redox classification put the well in redox class 1. These data indicate that the water is aerobic with a long contact time between water and bedrock which means that the well is a hypothesis II well.

VB-22:

VB-22 with an arsenic concentration of 12.9 µg/l is situated in acid-intermediate volcanic rock. The iron, manganese and sulphate concentration is 5 µg/l, 0.66 µg/l and 1.65 mg/l respectively indicating that the sample is a mixture of waters. Arsenic speciation showed that 98% of the arsenic is present as As(V) which means that the water is oxic. The alkalinity was measured to 78.1 mg/l which is close to the median value for tube wells in the area. Based on the arsenic speciation the well can be connected to hypothesis II. However as the redox classification indicates, a mixture of waters from different aquifers might have taken place which makes the conclusion uncertain.

M2:

Well M2 has got 11 µg/l arsenic. The well is drilled in alkaline volcanic rock and is situated 8 km south east of Malå. The iron, manganese and sulphate concentration was 230 µg/l, 10 µg/l and 32 mg/l respectively which placed the well into redox class 5, i.e. mixed water. The sulphate concentration was highest of all samples which makes hypothesis II the most probable even if a mixture of water has taken place.

M4:

The arsenic concentration in wells M4 is 8.1 µg/l and the well was installed in acid-intermediate plutonic rock. The iron, manganese and sulphate concentration is 11 µg/l, < 1 µg/l and 6.9 mg/l respectively which places the well into redox-class 1. Both alkalinity and potassium concentrations are low. Based solely on the redox-classification the conclusion is drawn that the arsenic in well M4 originates from processes connected to hypothesis II.

VB-72:

This well, with an arsenic concentration of 37.6 µg/l, supplies several house holds with water. Redox classification indicated a moderately high redox potential. Even though the well has got a low alkalinity, the relatively high sulphate concentration and the low iron concentration indicate a hypothesis II well.

7.4.3 Mixed waters

VB-59:

The water contained 6.1 µg/l arsenic. Iron, manganese and sulphate concentrations were < 5 µg/l, 22.6 µg/l and 2.45 mg/l respectively which in the redox-classification indicated a mixed water. The arsenic speciation showed that 80% of the arsenic in this well is present as As(V) meaning a relatively oxic water. This means that the probable hypothesis for this well is hypothesis II however as the redox classification indicated a mixed water the conclusion is uncertain.

VB-56:

The concentration of arsenic was 6.5 µg/l. The well was installed in acid-intermediate plutonic rock close to well VB-57. The iron, manganese and sulphate concentrations were low, 26 µg/l, 10.8 µg/l and 2.15 mg/l respectively and redox classification indicated a mixed water. However the arsenic speciation showed that 88% of the all the arsenic was present as As(V) indicating an oxidising environment. Even though well VB-57 and VB-56 is situated less than 300 meters apart there are some clear differences. The Mn concentration is almost 20 times larger in VB-57, the Fe concentration is 5 times larger

and the potassium concentration is twice that in this well. This supports the result from the redox-classification for VB-56. The well is probably connected to more than one aquifer which makes the results hard to interpret, but based on the arsenic speciation the most probable hypothesis for this well is hypothesis II.

VB-60:

The well has an arsenic concentration of 173 µg/l. The well is situated in alkaline volcanic rock. Nitrate, sulphate and iron concentrations are low but the alkalinity is relatively high, 145 mg/l which is the second highest values of the samples within this study. Redox classification put the well in redox class 5 which mean that the sample is probably a mixture of different waters. Therefore it is difficult to tie the arsenic in this well to a specific hypothesis and process. The relatively low sulphate concentration point toward a hypothesis I well but the high alkalinity indicate a hypothesis II well but as the redox classification showed the sample is probably a mixture of water which makes it hard to connect the arsenic concentration to a specific hypothesis.

VB-76:

This well, with an arsenic concentration of 6.1 µg/l, is the only dug well from the samples taken within this study. This well also has got the highest concentration of potassium of all the sampled wells which could depend on the use of fertilisers on a near by field. The well is situated in an out flow area in the vicinity of a lake. The arsenic in this well may therefore originate from processes according hypothesis II as well as hypothesis I.

7.5 Arsenic summary

Arsenic in this study ranged from below detection limit up to 300 µg/l. Of the drilled and dug wells, 29% respectively 11% had an arsenic concentration exceeding 10µg/l. The results can be compared to results from an extensive study performed by the Geological Survey of Finland, where the geology is similar to the geology in Sweden. This study showed that 22.5% of the drilled wells(n=530) in the Pirkanmaa region had an arsenic concentrations exceeding 10 µg/l. Also in the tampere belt similar concentrations have been found (Backman 2006). Thus the geology in the nordic region can give rise to elevated arsenic concentrations affecting a large number of domestic wells where arsenic analysis in many cases is missing.

8. Conclusions

Hydrogeological and hydrochemical conditions favourable for elevated concentrations of arsenic were delineated through the correlation between groundwater chemistry, geology and arsenic concentrations. A number of conclusions were drawn from the results of this study:

- Arsenic concentrations in sampled wells (n=140) of the Skellefte field varied between below detection limit (0.5 µg/l) and 300 µg/l. Of these 24 had an arsenic concentration exceeding the national food administrations guideline value of 10 µg/l. The high arsenic groundwater correlates with high arsenic till in the region.
- Two hypotheses are proposed as responsible for the high arsenic groundwater in the Skellefte field:
 - hypothesis I suggests reduction of iron-oxyhydroxides in till as the main source,
 - hypothesis II suggests oxidation of primary minerals in the crystalline rock as the main source of arsenic in the groundwater.
- High arsenic groundwater and alkaline volcanic rock correlate while low arsenic concentration correlates with sedimentary rocks.
- It was revealed that a large number of wells were redox-classified as “mixed waters”, thus it was hard to conclude whether they were reduced or oxidised.
- Dug wells have lower arsenic concentrations than tube wells. This would be due to the aeration of the water that takes place in the open well with following precipitation of iron-oxyhydroxides. The arsenic is adsorbed iron-oxyhydroxides.
- High nitrate concentration correlated with low dissolved iron and arsenic concentration. It is suggested that nitrate may lead to the oxidation of Fe²⁺ resulting in the formation of iron-oxyhydroxides that arsenic may be adsorbed to, thus arsenic and iron concentrations are low in high nitrate waters. Wells with high (>10 mg/l) nitrate concentrations are low in arsenic (<5 µg/l).
- Each of the high arsenic wells were linked to their respective hypothesis based on the groundwater composition. The process described in hypothesis II are responsible for the main part of arsenic in the ground water even though hypothesis I cannot be disregarded completely as two samples were identified as potential hypothesis I wells.

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Appendices

- Appendix 10.1. Results of the analyses from the sampled wells.
- Appendix 10.2. Analysis received from Norsjö municipality.
- Appendix 10.3. Analyses received from Malå municipality.
- Appendix 10.4. Arsenic analysis received from Skellefteå municipality.
- Appendix 10.5. Redox-classification table.
- Appendix 10.6. Redox classes for VB and Malå samples.
- Appendix 10.7. Redox-classes of the Norsjö samples.
- Appendix 10.8. Geologic map with the sampled wells plotted.

Appendix 10.1. Results of the analyses from the sampled wells.

StationID	Sample_type	pH	temp oC	Eh	EC
				mV	uS/cm
VB-4	dugwell	6.32	8.1	125	237
VB-5	tubewell	6.74	11.7	-120	160
VB-5b	dugwell	7.15	14	67	63.2
VB-5c	dugwell	6.56	8.9	126	89.8
VB-6	tubewell	6.54	9.7	130	153.2
VB-13	tubewell	6.43	11.2	130	232
VB-13b	tubewell	6.29	11.2	160	267
VB-22	tubewell	6.79	12.3	107	188.8
VB-22b	dugwell	6.24	9.4	114	176.3
VB-23	dugwell	6.77	7.1	103	93.3
VB-23b	tubewell	7.02	9.4	127	229
VB-32	dugwell	5.89	10.5	115	42
VB-34	tubewell	5.62	10.2	295	177.2
VB-36	tubewell	6.58	15	165	133.5
VB-36b	dugwell	6.12	10.1	279	73.3
VB-38	tubewell	6.65	8.4	79	75.6
VB-45	dugwell	6.4	6.1	95	75.1
VB-46	tubewell	5.7	12.5	130	104.6
VB-47	tubewell	6.39		90	190.7
VB-48	tubewell	6.59	11.3	121	349
VB-49	tubewell	7.02	9.1	120	230
VB-51	dugwell	6	11.4	127	25.3
VB-54	dugwell	6.29	12.1	130	245
VB-56	tubewell	6.92	6.8	101	181
VB-57	tubewell	6.99	6.8	111	296
VB-58	dugwell	5.87	16	157	56.9
VB-59	tubewell	6.69	9.8	172	243
VB-60	tubewell	6.59		91	356
VB-61	tubewell	7.85	13		
VB-62	dugwell	6.17	6.3		
VB-63	tubewell	5.25	6.6		
VB-64	dugwell	5.55	11		
VB-65	dugwell	5.1	6		
VB-66	dugwell	5.35	5		
VB-67	dugwell	5.62	10.5		
VB-68	tubewell	5.79	15		
VB-69	tubewell	5.73	12		
VB-70	dugwell	5.6	8.2		
VB-71	dugwell	6.83	7.4		
VB-72	tubewell	7.37	5.1		
VB-73	dugwell	4.1	6.5		
VB-74	dugwell	4.8	6		
VB-75	tubewell	5.87	9.2		
VB-76	dugwell	6.1	7.1		

StationID	HCO3	Cl	Fe	Mn	SO42-	As	NO3-	Al
	mg/l	mg/l	ug/l	ug/l	mg/l	ug/l	mg/l	ug/l
VB-4	69.4	7.38	5	29.07	2.52	0.25	60.05	22
VB-5	70.8	3.51	1728	270.38	0.39	293	0.50	9
VB-5b	16.8	1.05	78	3.78	1.15	3.1	0.50	10
VB-5c	37.8	2	5	1.64	0.63	0.8	2.92	3
VB-6	41.5	5.11	173	26.03	2.85	0.25	4.69	27

VB-13	21.6	3.1	51	4.09	10.35	0.6	1.11	30
VB-13b	0	0	284	20.06	0	0.5	0.50	38
VB-22	78.1	2.95	5	0.66	1.65	12.9	0.62	8
VB-22b	38.9	4.71	63	24.96	2.76	0.25	71.43	33
VB-23	35.5	2.07	118	2.25	0.92	2.9	26.62	103
VB-23b	101	3.85	441	219.44	1.41	3.3	0.50	7
VB-32	14.2	1.38	135	10.89	0.57	0.25	3.99	37
VB-34	51.9	6.12	5	298.06	1.23	0.25	83.08	12
VB-36	62	1.31	1349	166.45	0.46	2.2	0.50	56
VB-36b	22	1.23	5	1	0.95	4.1	25.38	4
VB-38	112	4.49	5	0.96	1.14	2.3	76.92	9
VB-45	12.9	1.94	135	10.33	1.71	0.25	1.68	19
VB-46	12.1	10.22	28	3.76	1.27	0.25	22.81	94
VB-47	50.8	13.93	229	30.56	1.89	0.25	19.75	8
VB-48	124	18.11	1234	233.82	3.33	3	0.50	7
VB-49	49.4	12.25	5	1.79	2.22	0.25	128.07	35
VB-51	5.49	1.23	13	2.44	0.68	0.25	0.35	42
VB-54	126	6.87	1186	832.45	1.37	1.2	3.54	32
VB-56	63	2.64	26	10.76	2.15	6.5	0.35	22
VB-57	94.6	11.84	140	192.54	3.96	24.2	0.50	9
VB-58	10.1	1.61	21	29.47	1.42	0.25	0.50	32
VB-59	102	1.59	5	22.58	2.45	6.1	0.75	4
VB-60	145	5.82	5	65.37	4.05	178.4	0.93	25
VB-61	61.61	4	5	3.54	16.1	201.2	0.50	4
VB-62	18.91	2	60	6.16	0.088	2.3	0.50	21
VB-63	16.71	4	201	27.63	20.5	2	0.50	69
VB-64	15.49	2	20	2.51	4.2	0.9	1.82	37
VB-65	14.40	2	12	3.29	3.48	0.25	0.50	82
VB-66	21.35	2	20	4.38	5.82	0.25	53.14	48
VB-67	43.19	12	5	1.76	5.26	0.8	37.24	13
VB-68	22.94	3	481	28.52	7.21	0.25	0.50	62
VB-69	53.07	2	42	5.96	14.8	1.5	22.98	5
VB-70	22.20	2	510	95.12	2.85	1.3	0.71	2
VB-71	27.45	2	17	1.19	9.49	5.2	0.50	8
VB-72	43.92	2	21	58.36	7.12	37.6	0.50	19
VB-73	0	2	3313	44.65	6.5	4.2	0.50	948
VB-74	1.83	2	440	87.31	0.25	2.6	0.80	226
VB-75	89.79	3	315	32.9	28.9	0.25	29.49	7
VB-76	57.46	2	22	8.8	3.64	6.1	34.45	36

StationID	B	Ba	Ca	Cd	Cr	Cu	K	Li
	ug/l	ug/l	mg/l	ug/l	ug/l	ug/l	mg/l	ug/l
VB-4	15	28.16	29035	0.07	0.25	50.6	7977	3.7
VB-5	7	16.41	26760	0.07	0.25	2.4	2710	4
VB-5b	2.5	2.44	7866	0.025	0.25	1.9	1420	0.6
VB-5c	2.5	10.47	15429	0.025	0.25	90.5	1484	0.1
VB-6	9	5.14	22912	0.16	0.25	6.6	2682	5.2
VB-13	2.5	27.27	34665	0.1	0.25	289.2	2025	1.2
VB-13b	2.5	34.81	42017	0.09	0.25	217.4	2294	1.2
VB-22	9	1.97	32913	0.025	0.25	6.7	1756	1.2
VB-22b	10	78.62	24654	0.37	0.25	172.4	4751	0.4
VB-23	2.5	15.88	15142	0.3	0.25	41.4	4752	0.2
VB-23b	2.5	14.98	42635	0.005	0.25	0.6	2431	0.6
VB-32	2.5	9.75	4012	0.23	0.25	534.8	686	1.1
VB-34	6	40.02	24130	0.025	0.25	660.7	3953	1
VB-36	2.5	21.09	26291	0.025	0.25	7.7	2440	1.4
VB-36b	8	4.92	8642	0.025	0.25	92.1	1336	1.8
VB-38	13	21.94	51369	0.025	0.25	22.8	2283	5.2

VB-45	2.5	12.04	6931	0.025	0.6	1.5	1297	0.2
VB-46	10	29.16	10888	0.25	0.25	332.3	2759	1.3
VB-47	10	9.02	23633	0.09	0.25	4.3	2971	16.6
VB-48	23	56.35	43039	0.025	0.25	2.2	3756	8.3
VB-49	15	52.25	21975	0.09	0.25	14.9	8204	1.5
VB-51	2.5	12.24	2291	0.025	0.25	146.3	396	0.2
VB-54	9	25.15	35694	0.1	0.25	3.8	5051	9.4
VB-56	25	3.83	3666	0.025	0.25	3.4	3757	4
VB-57	18	17.7	27245	0.025	0.25	2.1	7766	6.7
VB-58	2.5	11.01	4830	0.025	0.25	3.2	1390	0.4
VB-59	13	37.95	39565	0.45	0.25	7.2	3340	1.5
VB-60	9	117.91	47594	0.1	0.25	2.8	2032	0.6
VB-61	74	6.74	18.34	0.025	2.6	6.3	0.234	0.9
VB-62	2.5	4.91	6.827	0.025	1.1	1.7	1.042	0.6
VB-63	2.5	85.11	11.368	0.025	1.2	9.3	1.27	1.4
VB-64	2.5	9.51	5.803	0.025	1.1	87.7	0.83	0.1
VB-65	6	13.72	4.181	0.025	1.1	4.3	0.767	0.3
VB-66	12	40.07	9.322	0.025	0.9	112.2	6.494	0.2
VB-67	6	27.5	16.706	0.025	1.7	183.8	2.729	4.2
VB-68	2.5	11.37	10.064	0.23	1.5	2753.9	0.733	0.3
VB-69	9	13.17	18.092	0.025	2.3	135.5	2.857	2.5
VB-70	2.5	5.81	6.118	0.025	1	21.5	1.179	0.6
VB-71	5	7.38	8.675	0.025	1.1	13.2	2.192	9.4
VB-72	17	4.42	13.312	0.025	1.7	1.9	1.272	4.2
VB-73	2.5	17.97	2.134	0.025	1.4	17.7	0.415	0.8
VB-74	34	49.31	23.188	0.025	0.6	5.5	1.248	1
VB-75	28	7.79	15.443	0.025	1.1	35.1	3.996	3.8
VB-76	16	45.3	12.657	0.025	2.7	6.1	17.395	0.2

StationID	Mg	Na	Ni	P	Pb	S	SiO2	Zn
	ug/l	ug/l	ug/l	ug/l	ug/l	mg/l	mg/l	ug/l
VB-4	5058	10998	0.5	10	0.3	7	15.74	33.3
VB-5	2079	4161	0.9	465	0.2	0.5	21.94	34.9
VB-5b	886	2629	0.4	10	0.2	4	15.19	7.7
VB-5c	914	1842	0.1	10	1.6	1	7.45	34.3
VB-6	2423	4208	1.5	10	0.2	8	16.25	183.3
VB-13	3867	5293	0.8	10	1.2	26	12.68	179.3
VB-13b	4537	4058	0.3	10	0.7	31	13.54	369.7
VB-22	3452	3174	0.02	10	0.2	4	8.63	8.9
VB-22b	2800	3840	0.2	10	0.6	8	10.09	563.9
VB-23	686	1580	1.3	136	2.8	1	5.29	27.2
VB-23b	3191	3270	0.1	36	0.2	4	11.26	39
VB-32	806	2399	3.2	10	2.7	0.5	11.43	442.9
VB-34	5045	2725	2.6	10	2.8	4	8.44	155.1
VB-36	849	1525	0.3	10	0.6	1	5.60	9.7
VB-36b	1385	3170	1.8	10	0.7	2	11.53	39.3
VB-38	3692	5492	0.3	20	0.5	3	12.51	18.7
VB-45	1317	2489	4.4	10	0.7	4	14.47	18
VB-46	1974	3533	4.5	10	2	3	10.50	345.5
VB-47	5486	5233	0.3	10	0.2	5	18.43	35
VB-48	6637	23079	0.1	10	0.1	9	19.06	10.2
VB-49	8960	5746	3.4	10	0.5	6	11.56	36.3
VB-51	463	1297	2.2	10	1.3	2	7.54	37.7
VB-54	5311	7869	5.8	20	0.2	4	19.55	11.3
VB-56	1522	31247	0.4	10	0.1	6	15.18	4.4
VB-57	8663	16964	0.2	31	0.2	10	20.10	16.8
VB-58	761	2155	3.5	10	0.2	4	12.12	22.7
VB-59	4287	6362	0.1	27	2.2	7	11.69	322.2

VB-60	17414	5258	0.2	10	0.2	11	9.45	3.3
VB-61	760	16486	0.3	10	0.7	6	19.48	88.8
VB-62	878	1519	1.6	10	0.2	3	10.38	5.6
VB-63	2154	1586	2	10	0.4	8	13.28	108.5
VB-64	669	1746	0.8	10	1.9	1	10.12	80
VB-65	864	1917	2.9	10	0.3	2	7.78	111.4
VB-66	1513	1479	0.5	10	12.5	2	7.71	23.6
VB-67	5653	5263	3.7	10	0.6	2	17.75	72.9
VB-68	836	1289	13.5	10	5.5	3	8.11	468.7
VB-69	4251	4752	5.3	10	0.7	6	17.82	196.3
VB-70	643	2291	0.2	10	0.3	1	12.47	27.4
VB-71	3218	2052	0.1	10	0.2	3	12.58	12.4
VB-72	3929	3852	1.4	10	0.1	3	11.82	2.5
VB-73	668	1393	3.5	10	1.1	3	12.07	17.3
VB-74	3482	4807	1.6	10	0.6	24	13.46	135.8
VB-75	2463	4997	5.9	10	0.3	10	13.44	283.1
VB-76	2683	1736	0.4	10	1.5	1	7.78	217.1

Appendix 10.2. Analysis received from Norsjö municipality.

SampleID	Type	Al	As	Mn	Be	Pb	B	Cd	Co
EB 073	tube	2	22	370	0.5	0.05	5	0.005	0.006
EB 014	tube	2	300	220	0.5	0.05	3	0.005	0.0025
EB 078	tube	7	0.4	100	0.5	0.05	13	0.005	0.0025
EB 020	tube	8	0.5	1.3	0.5	2.2	3	0.04	0.06
EB 079	tube	7	1.9	2.7	0.5	0.8	0.5	0.01	0.07
EB 038	tube	2	0.5	0.7	0.5	0.2	2.83	0.005	0.05
GB 060	tube	3	270	57	0.5	0.05	11	0.02	0.12
EB 085	tube	3	45	1.3	0.5	0.3	25	0.08	0.2
EB 061	tube	3	46	0.3	0.5	0.3	20	0.005	0.15
EB 042	tube	5	0.6	3.6	0.5	0.4	4	0.27	0.05
EB 070	tube	0.5	12	0.4	0.5	0.05	0.5	0.005	0.01
EB 012	tube	13	21	390	0.5	0.1	6	0.02	0.08
EB 057	tube	3	0.7	4.1	0.5	1	17	0.23	0.05
EB 048	tube	5	7	0.7	0.5	0.3	15	0.1	0.006
GB 015	tube	3	6	52	0.5	0.05	23	0.005	0.0025
EB 045	tube	3	2.8	2.2	0.5	0.3	7	0.06	0.02
EB 069	tube	6	3.1	100	0.5	0.2	19	0.01	0.05
EB 068	tube	2	0.5	4.1	0.5	0.5	20	0.04	0.04
EB 003	tube	4	0.3	0.4	0.5	0.7	5	0.02	0.01
EB 063	tube	4	0.3	3.1	0.5	3.3	12	0.05	0.11
EB 056	tube	10	8	100	0.5	0.4	7	0.005	0.25
EB 011	tube	2	9.6	350	0.5	0.05	13	0.005	0.25
EB 029	tube	1	1.2	3.8	0.5	0.9	20	0.06	0.38
EB 085	tube	15	40	32	0.5	0.5	58	0.02	2
GB 060	tube	5	190	55	0.5	0.05	20	0.02	0.04
EB 085	tube	3	8.1	2800	0.5	0.9	35	0.17	1.3
EG 066	dug	2	3.1	0.8	0.5	0.05	3	0.005	0.0025
EG 084	dug	30	0.2	2.3	0.5	25	6	0.22	0.24
GG 087	dug	0.5	16	0.4	0.5	0.4	50	0.02	0.02
GG 088	dug	34	0.6	0.8	0.5	0.2	1	0.02	0.04
EG 084	dug	32	0.6	62	0.5	0.2	23	0.03	0.28
EG 084	dug	39	0.1	2	1.3	20	13	0.04	0.12
EG 089	dug	1	0.4	11	0.5	0.2	3	0.005	0.03
EG 040	dug	42	0.1	5.6	0.5	1.6	19	0.03	0.09
GG 076	dug	16	0.1	23	0.5	0.6	2	0.005	0.24
EG 017	dug	14	0.8	26	0.5	1	5	0.01	0.36
EG 019	dug	0.5	0.5	1.4	0.5	1	4	0.01	0.02
GG 021	dug	38	0.1	1.4	0.5	0.9	0.5	0.005	0.02
EG 030	dug	5	1.8	23	0.5	0.6	5	0.08	0.19
EG 041	dug	8	0.8	5.7	0.5	0.8	3	0.005	0.1
GG 071	dug	58	0.2	8	0.5	1.3	3	0.02	0.3
GG 047	dug	53	0.1	1.5	0.5	0.8	11	0.02	0.06
GG 046	dug	9	0.1	0.4	0.5	0.6	10	0.08	0.04
GG 043	dug	15	0.3	2.7	0.5	1.2	6	0.18	0.05
EG 081	dug	24	0.3	10	0.5	5.1	3	0.03	0.14
EG 083	dug	11	0.1	4.6	0.5	1.5	7	0.08	0.28
EG 082	dug	100	0.5	3.2	0.5	1.6	6	0.02	0.09
EG 080	dug	29	0.2	4.6	0.5	2	6	0.09	0.13
EG 084	dug	28	0.1	1.7	0.5	12	5	0.13	0.18
EG 009	dug	8	13	1.2	0.5	0.7	4	0.02	0.73
EG 052	dug	38	11	5.9	0.5	1.4	2	0.005	0.09
GG 049	dug	7	15	230	0.5	0.4	2	0.005	0.98
EG 050	dug	21	4.3	5.5	0.5	1.9	1	0.07	0.06

GG 025	dug	5	0.5	4.5	0.5	1.3	33	0.005	0.06			
GG 026	dug	4	2.6	0.3	0.5	0.3	0.5	0.005	0.04			
GG 055	dug	2	25	0.3	0.5	0.2	0.5	0.1	0.03			
EG 034	dug	15	0.6	26	0.5	1.5	0.5	0.03	0.09			
EG 054	dug	10	0.2	3.4	0.5	1	14	0.03	0.06			
GG 051	dug	1	16	0.1	0.5	0.2	24	0.005	0.11			
EG 018	dug	130	0.2	4.2	0.5	1.9	15	0.13	0.07			
EG 023	dug	100	1.1	190	0.5	0.3	0.5	0.02	1.8			
EG 037	dug	24	0.2	7.3	0.5	2	6	0.05	0.26			
EG 001	dug	49	0.2	3.1	0.5	3.4	4	0.06	0.05			
EG 022	dug	4	0.2	1.2	0.5	0.4	0.5	0.07	0.02			
EG 072	dug	160	9.7	50	0.5	0.4	4	0.03	0.38			
EG 077	dug	5	0.6	33	0.5	1	7	0.05	0.12			
EG 016	dug	2	0.1	9.7	0.5	0.4	3	0.16	0.04			
EG 005	dug	40	0.9	10	0.5	3.3	4	0.25	0.24			
EG 006	dug	68	0.6	35	0.5	1.1	12	0.07	0.7			
EG 007	dug	250	1.4	48	0.5	1.1	12	0.06	1.1			
EG 035	dug	160	1.8	66	0.5	0.3	3	0.01	0.06			
GG 013	dug	14	1.6	33	0.5	0.4	2	0.02	0.24			
GG 004	dug	9	0.1	0.4	0.5	1.1	15	0.01	0.08			
EG 002	dug	23	0.4	0.4	0.5	1.9	8	0.04	0.14			
GG 062	dug	5	0	0.7	0.5	0.3	12	0.11	0.03			
GG 027	dug	1	0.6	0.6	0.5	0.5	6	0.005	0.14			
EG 031	dug	2	1.7	26	0.5	1.1	36	0.04	0.5			
EG 010	dug	1	1.6	12	0.5	1.4	51	0.07	0.53			
GG 032	dug	20	0.3	4.3	0.5	0.05	22	0.05	0.29			
GG 033	dug	3	0.4	4.8	0.5	0.5	4	0.01	0.06			
EG 028	dug	10	0.3	0.7	0.5	0.8	12	0.02	0.02			
GG 053	dug	3	0.3	0.05	0.5	0.2	17	0.02	0.08			
EG 060	dug	4	60	5.5	0.5	0.2	41	0.01	0.18			
EG 075	dug	1	11	0.3	0.5	0.05	9	0.02	0.01			
EG 066	dug	2	3.3	0.4	0.5	0.05	5	0.02	0.02			
EG 044	dug	2	2	0.3	0.5	0.3	32	0.03	0.02			

SampleID	Cu	Cr	Li	Mo	Ni	Se	Ag	Sr	Tl	U	V	Zn
EB 073	0.09	5	4.9	0.16	0.2	0.05	0.01	110	0.5	0.07	3.2	68
EB 014	0.5	2.5	3.6	1.3	0.3	0.05	0.01	35	0.5	0.007	1.8	50
EB 078	0.1	3	8.9	0.66	0.05	0.05	0.01	74	0.5	0.12	1.6	21
EB 020	180	2.2	0.5	0.05	1.7	0.2	0.01	21	0.5	0.03	0.2	90
EB 079	190	1.6	1.8	0.78	0.3	0.2	0.01	25	0.5	0.16	0.28	290
EB 038	38	2	2.2	0.26	0.1	0.1	0.01	28	0.5	0.18	0.25	24
GB 060	0.2	9.7	0.5	12	0.1	0.3	0.01	150	0.5	2.4	1.6	2
EB 085	59	13	1.3	0.5	0.05	1	0.01	290	0.5	1.5	0.0025	51
EB 061	14	9.2	0.5	0.61	0.05	1	0.07	150	0.5	0.77	1.7	17
EB 042	61	1.4	7.1	0.69	2.4	0.05	0.01	66	0.5	20	0.0025	89
EB 070	1.6	0.2	5.3	0.17	0.4	0.2	0.01	20	0.5	0.11	0.0025	2
EB 012	0.2	1.1	11	4.1	0.05	0.1	0.01	51	0.5	1.4	0.0025	58
EB 057	100	2.9	0.5	0.42	0.1	0.3	0.1	73	0.5	0.25	1.2	330
EB 048	11	4.4	2.4	3.8	0.3	0.6	0.1	84	0.5	2.8	2.6	76
GB 015	0.5	0.3	7.1	2.5	0.1	0.2	0.01	130	0.5	0.17	0.0025	6
EB 045	6.1	0.4	5.6	2.6	0.05	0.3	0.01	29	0.5	0.95	0.15	66
EB 069	12	1.8	1.7	14	0.4	0.05	0.01	110	0.5	0.75	0.0025	31
EB 068	86	0.7	1.6	1	1.3	0.2	0.01	33	0.5	0.59	3	88
EB 003	44	0.4	0.5	0.4	0.4	0.3	0.02	11	0.5	0.03	2.2	12
EB 063	430	1	0.5	0.86	1.7	0.1	0.04	36	0.5	1.3	1.5	84
EB 056	0.4	0.3	5.1	1.8	3.3	0.2	0.08	110	0.5	0.16	0.52	2
EB 011	0.8	0.03	230	0.39	4.4	0.05	0.01	29	0.5	0.03	0.97	2
EB 029	53	0.03	2.1	0.51	6.1	0.4	0.01	47	0.5	3.6	0.0025	53

EB 085	33	1.1	2.9	1.8	11	0.4	0.01	510	0.5	4	6.5	9
GB 060	0.2	1.6	0.5	14	0.4	1	0.3	150	0.5	2.4	1.4	3
EB 085	170	1.9	2.7	0.41	8.2	1	0.01	150	0.5	0.97	0.0025	140
EG 066	0.09	1.1	0.5	1.7	0.2	0.5	0.02	40	0.5	0.3	0.43	0.5
EG 084	270	0.7	1.7	0.42	2.3	0.2	0.01	61	0.5	0.14	0.21	160
GG 087	75	0.4	0.5	1.3	0.4	0.4	0.01	18	0.5	0.07	0.14	23
GG 088	25	0.7	0.5	3	0.5	1	0.01	15	0.5	0.06	2.2	2
EG 084	1.7	0.4	2.1	1.6	2.9	0.9	0.01	130	0.5	0.75	0.0025	4
EG 084	1.9	1	1.3	0.7	1.9	0.9	0.1	57	0.5	0.07	0.0025	26
EG 089	3.5	0.03	0.5	0.91	0.8	0.3	0.01	20	0.5	0.003	0.0025	450
EG 040	59	2.3	0.5	0.14	0.2	0.2	0.01	36	0.5	0.18	0.64	13
GG 076	57	1.2	0.5	0.11	1.3	0.05	0.01	13	0.5	0.03	0.6	15
EG 017	110	1.8	0.5	0.07	0.3	0.05	0.01	36	0.5	0.04	0.71	110
EG 019	79	2.2	0.5	0.01	0.05	0.2	0.01	21	0.5	0.03	0.51	47
GG 021	8.2	1.5	0.5	0.31	0.4	0.1	0.01	19	0.5	0.06	2.1	12
EG 030	25	3.7	1.3	0.92	0.1	0.05	0.01	96	0.5	0.24	1.8	41
EG 041	250	3.9	0.5	0.22	0.3	0.05	0.01	47	0.5	0.07	0.46	56
GG 071	660	3.4	0.5	0.09	1.6	0.2	0.02	31	0.5	0.08	0.35	31
GG 047	320	1.9	0.5	0.07	1.7	0.3	0.01	19	0.5	0.05	0.34	43
GG 046	130	2.3	1.2	0.43	0.7	0.2	0.01	22	0.5	0.07	0.35	28
GG 043	380	1.2	0.5	0.01	2.2	0.1	0.01	9	0.5	0.003	0.31	33
EG 081	550	6.8	3.6	0.25	1.9	0.5	0.06	70	0.5	0.14	0.37	170
EG 083	86	2.5	0.5	0.2	1.7	0.3	0.03	56	0.5	0.03	0.27	120
EG 082	150	3	0.5	0.08	2.6	0.2	0.01	24	0.5	0.07	0.32	33
EG 080	33	2.1	0.5	0.05	2.1	0.05	0.01	17	0.5	0.04	0.25	570
EG 084	1400	3.7	1.4	0.06	1.8	0.2	0.02	54	0.5	0.06	0.23	88
EG 009	65	5.6	0.5	2.1	0.3	0.2	0.04	18	0.5	0.38	0.45	190
EG 052	41	3.3	0.5	0.31	0.1	0.2	0.04	24	0.5	0.09	0.71	18
GG 049	2	6.7	0.5	0.15	0.05	0.3	0.03	46	0.5	0.06	0.9	8
EG 050	270	3.7	0.5	1.8	0.7	0.4	0.04	28	0.5	0.08	0.44	97
GG 025	76	5.8	0.5	0.11	0.3	0.2	0.03	38	0.5	0.07	0.43	18
GG 026	29	3.8	0.5	0.33	0.05	0.2	0.03	31	0.5	0.05	0.51	20
GG 055	3.7	2.9	0.5	0.34	0.05	0.2	0.02	29	0.5	0.1	0.49	51
EG 034	970	4.2	1.5	0.47	0.4	0.4	0.03	69	0.5	0.04	1.3	250
EG 054	250	2.8	0.5	0.03	0.4	0.2	0.03	63	0.5	0.03	0.34	45
GG 051	79	2.2	0.5	1.2	0.05	0.2	0.03	18	0.5	0.05	0.52	12
EG 018	2200	1.5	0.5	0.09	0.9	0.2	0.03	51	0.5	0.42	0.0025	130
EG 023	29	1.4	2.9	0.08	1.1	0.3	0.01	27	0.5	0.16	0.0025	5
EG 037	91	1.2	9.4	0.5	1.6	0.2	0.01	50	0.5	0.14	0.0025	84
EG 001	140	1.9	0.5	0.08	2.5	0.1	0.04	42	0.5	0.06	0.0025	42
EG 022	100	0.4	0.5	0.27	0.5	0.2	0.01	21	0.5	0.05	0.0025	240
EG 072	20	1.4	4.9	0.37	0.5	0.2	0.01	43	0.5	0.1	0.0025	43
EG 077	290	3.9	3.6	0.3	0.9	0.5	0.3	76	0.5	0.36	1.4	42
EG 016	27	0.4	0.5	0.34	0.1	0.2	0.01	12	0.5	0.01	0.0025	300
EG 005	180	0.03	0.5	0.1	0.6	0.05	0.01	40	0.5	0.27	0.0025	730
EG 006	240	3	0.5	0.26	1.1	0.1	0.05	33	0.5	0.6	0.0025	73
EG 007	45	1.1	0.5	4.4	3.8	1	0.05	55	0.5	2.9	0.16	640
EG 035	88	3.6	0.5	0.45	0.2	0.05	0.01	32	0.5	0.65	1.5	19
GG 013	70	1	1.9	1.9	0.1	0.05	0.01	27	0.5	0.42	0.0025	14
GG 004	180	2.7	5.5	0.25	0.3	0.3	0.04	81	0.5	0.07	1.2	37
EG 002	310	0.8	0.5	1.4	0.2	0.5	0.02	33	0.5	0.18	0.0025	37
GG 062	9	2.2	3.4	0.34	2.9	0.4	0.01	79	0.5	0.04	0.0025	15
GG 027	20	0.03	110	0.42	5.4	0.05	0.01	19	0.5	0.02	1	14
EG 031	34	0.03	1.1	0.51	4.3	0.4	0.01	67	0.5	0.25	0.0025	13
EG 010	440	0.03	1.5	1.3	5.9	1	0.01	70	0.5	0.08	0.0025	85
GG 032	3.5	1.3	2.8	0.1	2.6	1	0.01	42	0.5	0.07	0.98	15
GG 033	200	0.03	0.5	0.08	0.8	0.3	0.01	33	0.5	0.01	0.0025	38
EG 028	79	0.03	1.2	0.24	5.5	0.1	0.01	66	0.5	0.15	0.18	29
GG 053	6.7	0.2	0.5	1	1.9	0.4	0.01	37	0.5	0.09	0.0025	39
EG 060	2.2	0.2	0.5	0.05	3.2	0.2	0.01	130	0.5	0.14	0.0025	22

EG 075	0.3	0.5	0.5	2.5	0.05	0.4	0.01	24	0.5	0.19	0.18	3
EG 066	0.09	0.6	0.5	2.3	0.05	2	0.07	39	0.5	0.31	0.23	0.5
EG 044	91	0.3	0.5	0.97	1	0.5	0.01	11	0.5	0.06	0.0025	17

Appendix 10.3. Analyses received from Malå municipality. Arsenic concentrations above 5 $\mu\text{g/l}$ are marked in grey.

SampleID	Type	pH	EC	Temp	HCO3	Cl	Fe	Mn	SO4	As
			mS/m		mg/l	mg/l	ug/l	ug/l	mg/l	ug/l
M1	tubewell	8.2	21	20.8	130	5	60	40	9.5	37
M2	tubewell	7.4	22	11	94	3.6	230	10	32	11
M3	tubewell	6.8	8.2	20.8	27	2.1	5	10	9.1	0.5
M4	tubewell	7.9	13		60	1.8	11	1	6.9	8.1
M5	tubewell	7.2	16	19.4	65	2.1	5	1	9.6	1.1
M6	dugwell	6.7	13	19.6	52	3.2	31	3	4.5	0.2
M7	tubewell	7.4	34	25	200	4.3	5	1	8	0.1

SampleID	Ca	Cd	Cr	Cu	K	Mg	Na	Ni	Sb	Se	NH4	NO2	PO4	F
	ug/l	ug/l	ug/l	ug/l	ug/l	ug/l	ug/l	ug/l	ug/l	ug/l	mg/l	mg/l	mg/l	mg/l
M1	25000	0.07	0	5	1000	2300	22000	0.1	0.5	0.05	0.01	0	0	2.8
M2	37000	1.6	0.3	20	2000	2100	3700	7.4	0.5	0.9	0.02	0.01	0.01	0.24
M3	10000	0.02	0.1	150	200	1500	3300	0.5	0.5	1	0	0	0	0.25
M4	14000	0.12	0.1	32	1600	4100	3600	0.5	0.5	0.9	0	0	0.01	0.6
M5	20000	0.03	0.3	5	1000	5000	4200	0.1	0.5	0.2	0.03	0	0.01	1.7
M6	16000	0.03	0.4	19	2200	1200	1900	0.3	0.5	0.05	0.01	0	0.01	0.05
M7	56000	0.03	1.2	15	1500	6300	4100	0.2	0.5	0.3	0	0	0.01	0.25

Appendix 10.4. Arsenic analysis received form Skellefteå municipality.

SampleID	Type	As
		ug/l
S1	Tubewell	4.8
S2	Tubewell	2
S4	Tubewell	0.1
S3	Tubewell	2.9
S5	Tubewell	0.2
S6	Tubewell	0.005
S7	Tubewell	0.2
S8	Tubewell	4.7
S9	Tubewell	17
S10	Tubewell	0.9
S11	Tubewell	0.005

Appendix 10.5. Redox-classification table (Swedish environmental protection agency, 2000).

Class	Description	Fe [mg/l]	Mn [mg/l]	SO ₄ [mg/l]	
				Southern Sweden	Northern Sweden
1	High redox-potential/aerobic water	< 0.1	< 0.05	> 5	> 2
2	Moderately to high redox-potential/aerobic water	< 0.1	> 0.05	> 5	> 2
3	Low redox-potential anaerobic water	> 0.1	> 0.05	> 5	> 2
4	Very low redox-potential/anaerobic water	> 0.1	> 0.05	< 5	< 2
5	Mixed water				
	Type 1	< 0.1	All values	< 5	< 2
	Type 2	> 0.1	< 0.05	All values	All values

Appendix 10.6. Redox classes for VB and Malå samples. Arsenic concentrations >5µg/l are marked grey.

StationID	Sample_type	Fe ug/l	Mn ug/l	SO4 mg/l	As ug/l	Redox-class
M6	dugwell	31	3	4,5	0,2	1
VB-4	dugwell	5	29,07	2,52	0,25	1
VB-5b	dugwell	78	3,78	1,15	3,1	5: type 1
VB-5c	dugwell	5	1,64	0,63	0,8	5: type 1
VB-22b	dugwell	63	24,96	2,76	0,25	1
VB-23	dugwell	118	2,25	0,92	2,9	5: type 2
VB-32	dugwell	135	10,89	0,57	0,25	5: type 2
VB-36b	dugwell	5	1	0,95	4,1	5: type 1
VB-45	dugwell	135	10,33	1,71	0,25	5: type 2
VB-51	dugwell	13	2,44	0,68	0,25	5: type 1
VB-54	dugwell	1186	832,45	1,37	1,2	4
VB-58	dugwell	21	29,47	1,42	0,25	5: type 1
VB-76	dugwell	22	8,8	3,64	6,1	1
VB-62	dugwell	60	6,16	0,088	2,3	5: type 1
VB-64	dugwell	20	2,51	4,2	0,9	1
VB-65	dugwell	12	3,29	3,48	0,25	1
VB-66	dugwell	20	4,38	5,82	0,25	1
VB-67	dugwell	5	1,76	5,26	0,8	1
VB-70	dugwell	510	95,12	2,85	1,3	3
VB-71	dugwell	17	1,19	9,49	5,2	1
VB-73	dugwell	3313	44,65	6,5	4,2	5: type 2
VB-74	dugwell	440	87,31	0,25	2,6	4
M1	tubewell	60	40	9,5	37	1
M2	Tubewell	230	10	32	11	5: type 2
M3	Tubewell	5	10	9,1	0,5	1
M4	Tubewell	11	1	6,9	8,1	1
M5	Tubewell	5	1	9,6	1,1	1
M7	Tubewell	5	1	8	0,1	1
VB-5	Tubewell	1728	270,38	0,39	293	4
VB-6	Tubewell	173	26,03	2,85	0,25	5: type 2
VB-13	Tubewell	51	4,09	10,35	0,6	1
VB-13b	Tubewell	284	20,06	0	0,5	5: type 2
VB-22	Tubewell	5	0,66	1,65	12,9	5: type 1
VB-34	Tubewell	5	298,06	1,23	0,25	5: type 1
VB-38	Tubewell	5	0,96	1,14	2,3	5: type 1
VB-46	Tubewell	28	3,76	1,27	0,25	5: type 1
VB-47	Tubewell	229	30,56	1,89	0,25	5: type 2
VB-48	Tubewell	1234	233,82	3,33	3	3
VB-49	Tubewell	5	1,79	2,22	0,25	5: type 1
VB-56	Tubewell	26	10,76	2,15	6,5	5: type 1
VB-57	Tubewell	140	192,54	3,96	24,2	3
VB-59	Tubewell	5	22,58	2,45	6,1	5: type 1
VB-60	Tubewell	5	65,37	4,05	178,4	2
VB-23b	Tubewell	441	219,44	1,41	3,3	4
VB-36	tubewell	1349	166,45	0,46	2,2	4
VB-61	Tubewell	5	3,54	16,1	201,2	1
VB-63	Tubewell	201	27,63	20,5	2	5: type 2
VB-68	Tubewell	481	28,52	7,21	0,25	5: type 2
VB-69	Tubewell	42	5,96	14,8	1,5	1
VB-72	Tubewell	21	58,36	7,12	37,6	2
VB-75	Tubewell	315	32,9	28,9	0,25	5: type 2

Appendix 10.7. Redox-classes of the Norsjö samples. Arsenic concentrations >5µg/l are marked grey.

StationID	Sample_type	Mn ug/l	As ug/l	Redox-class
EB 073	tubewell	370	22	2
EB 014	Tubewell	220	300	2
EB 078	Tubewell	100	0,4	2
EB 020	Tubewell	1,3	0,5	1
EB 079	Tubewell	2,7	1,9	1
EB 038	Tubewell	0,7	0,5	1
GB 060	Tubewell	57	270	2
EB 085	Tubewell	1,3	45	1
EB 061	Tubewell	0,3	46	1
EB 042	Tubewell	3,6	0,6	1
EB 070	Tubewell	0,4	12	1
EB 012	Tubewell	390	21	2
EB 057	Tubewell	4,1	0,7	1
EB 048	Tubewell	0,7	7	1
GB 015	Tubewell	52	6	2
EB 045	Tubewell	2,2	2,8	1
EB 069	Tubewell	100	3,1	2
EB 068	Tubewell	4,1	0,5	1
EB 003	Tubewell	0,4	0,3	1
EB 063	Tubewell	3,1	0,3	1
EB 056	Tubewell	100	8	2
EB 011	Tubewell	350	9,6	2
EB 029	Tubewell	3,8	1,2	1
EB 085	Tubewell	32	40	1
GB 060	Tubewell	55	190	2
EB 085	Tubewell	2800	8,1	2
EG 066	dugwell	0,8	3,1	1
EG 084	dugwell	2,3	0,2	1
GG 087	dugwell	0,4	16	1
GG 088	dugwell	0,8	0,6	1
EG 084	dugwell	62	0,6	2
EG 084	dugwell	2	0,07	1
EG 089	dugwell	11	0,4	1
EG 040	dugwell	5,6	0,09	1
GG 076	dugwell	23	0,1	1
EG 017	dugwell	26	0,8	1
EG 019	dugwell	1,4	0,5	1
GG 021	dugwell	1,4	0,1	1
EG 030	dugwell	23	1,8	1
EG 041	dugwell	5,7	0,8	1
GG 071	dugwell	8	0,2	1
GG 047	dugwell	1,5	0,1	1
GG 046	dugwell	0,4	0,1	1
GG 043	dugwell	2,7	0,3	1
EG 081	dugwell	10	0,3	1
EG 083	dugwell	4,6	0,1	1
EG 082	dugwell	3,2	0,5	1
EG 080	dugwell	4,6	0,2	1
EG 084	dugwell	1,7	0,1	1
EG 009	dugwell	1,2	13	1
EG 052	dugwell	5,9	11	1
GG 049	dugwell	230	15	2

EG 050	dugwell	5,5	4,3	1
GG 025	dugwell	4,5	0,5	1
GG 026	dugwell	0,3	2,6	1
GG 055	dugwell	0,3	25	1
EG 034	dugwell	26	0,6	1
EG 054	dugwell	3,4	0,2	1
GG 051	dugwell	0,1	16	1
EG 018	dugwell	4,2	0,2	1
EG 023	dugwell	190	1,1	2
EG 037	dugwell	7,3	0,2	1
EG 001	dugwell	3,1	0,2	1
EG 022	dugwell	1,2	0,2	1
EG 072	dugwell	50	9,7	2
EG 077	dugwell	33	0,6	1
EG 016	dugwell	9,7	0,09	1
EG 005	dugwell	10	0,87	1
EG 006	dugwell	35	0,6	1
EG 007	dugwell	48	1,4	1
EG 035	dugwell	66	1,8	2
GG 013	dugwell	33	1,6	1
GG 004	dugwell	0,4	0,1	1
EG 002	dugwell	0,4	0,4	1
GG 062	dugwell	0,7	0,025	1
GG 027	dugwell	0,6	0,6	1
EG 031	dugwell	26	1,7	1
EG 010	dugwell	12	1,6	1
GG 032	dugwell	4,3	0,3	1
GG 033	dugwell	4,8	0,4	1
EG 028	dugwell	0,7	0,3	1
GG 053	dugwell	0,05	0,3	1
EG 060	dugwell	5,5	60	1
EG 075	dugwell	0,3	11	1
EG 066	dugwell	0,4	3,3	1
EG 044	dugwell	0,3	2	1

Appendix 10.8. Geologic map with the sampled wells plotted.

