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Combining shallow geothermal energy and groundwater remediation

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ABSTRACT

Application of aquifer thermal energy storage (ATES) is growing rapidly to provide sustainable heating and cooling to buildings. However, presence of soil and groundwater contaminants in many urban environments limits applicability of ATES and hampers redevelopment of these sites. Combination of ATES and remediation is therefore considered a promising new concept to achieve both energy savings and improvement of the groundwater quality.

Several methods to combine ATES with remediation are discussed. The most promising combination concepts are considered to be 1) ATES and natural attenuation, 2) ATES and containment, 3) ATES and aboveground water treatment and 4) ATES and in-situ remediation.

Physical, chemical and biological measurements are presented of the first ATES system in the Netherlands that is, on purpose, constructed in an aquifer contaminated with chlorinated hydrocarbons. The applied combination concept focusses on natural attenuation and containment of the contaminants. Measurements show that conditions for reductive dechlorination improve at one of the monitoring wells.

Additional lab experiments with aquifer material from the ATES site demonstrate that after inoculum with dechlorinating bacteria full degradation of CIS and VC to ethene is achieved within four weeks. Further studies are required to show the effectiveness of this method under field conditions.

1. INTRODUCTION

Among the different types of underground thermal energy storage concepts, Aquifer Thermal Energy Storage (ATES) is an effective method to provide heating and cooling to buildings and industrial processes (for a review on system types see e.g. Sanner et al., 2003 or Lee, 2010). Application of Aquifer Thermal Energy Storage (ATES) in the Netherlands is growing exponentially (Drijver et al., 2012). The government wishes to stimulate this growth to diminish energy use and reduce emissions (Taskforce WKO, 2009). However, because many urban city centres deal with contaminated soil and groundwater (NVOE, 2010), more and more ATES ambitions are confronted with the presence of contaminants. Traditionally this is seen as a problem as it is not allowed to move contaminated groundwater (permits for ATES are not granted in that case). Since excavation (especially for deeper contaminations) is too costly, pump-and-treat is inefficient and natural attenuation is generally very slow, redevelopment of these sites is hampered.

The actual effects of ATES on contaminants are, however, still poorly understood. Potential negative effects include increased spreading and mobilization of contaminants and increased dissolution from pure product when a well is placed in a DNAPL source area (Zuurbier et al., 2013). However, it is also hypothesised that dilution, mixing of water qualities and temperature changes may have a positive influence on in situ biodegradation rates, and thereby stimulate natural attenuation (NVOE, 2010; Hartog, 2011).

From the perspective of energy storage, one of the primary concerns is the risk of biological well clogging. However, other types of clogging may also occur, for example chemical clogging when water of different qualities is mixed (Beek, 2010). Well clogging leads to increased maintenance costs and reduces the lifetime of the ATES system, which potentially makes ATES financially infeasible.

Although ATES and groundwater remediation are both considered proven technology in the Netherlands when practiced independently, the combination is not straightforward. The design approach of conventional ATES systems and conventional remediation techniques differ substantially. ATES systems have a typical flow rate of up to several hundred cubic meters of groundwater per hour and are designed to operate for more than 20 years without clogging problems. On the other hand, in-situ remediation techniques like pump-and-treat, chemical oxidation or biostimulation are designed to function no more than a few years and are known to cause clogging problems.

The goal of this research is to improve knowledge on the combination of ATES and remediation and is based on results of a Dutch research program (www.meermetbodemenergie.nl). After a theoretical background on the contaminants that are most frequently encountered and their biodegradation characteristics in section 2, several combination concepts are discussed in section 3. Section 4 presents the results of physical, chemical and biological monitoring of the first ATES system in the Netherlands that was, on purpose, installed in a contaminated aquifer. Implications of the combination concepts and monitoring results are discussed in section 5, leading to the conclusions in section 6.

2. THEORETICAL BACKGROUND

2.1 Contaminants

The contaminants that are found most often in the depth domain of ATES systems (20-200 m bgl) are chlorinated hydrocarbons (CHC). Commonly applied as degreasers at dry cleaners and chemical and metal processing factories, these CHC have entered the groundwater after leakage or improper disposal (Grindstaff, 1998). Known to be carcinogenic (Fawell and Gowers, 2000), chlorinated solvents are considered a threat to groundwater systems and human health (Bradley, 2000). Due to their recalcitrant nature, CHC's are among the most difficult contaminants to be cleaned up, especially when they exist as dense non aqueous-phase liquid (DNAPL). As DNAPL is denser than water, it can travel over large distances, not only in the horizontal but also in the vertical direction. Hot spots with pure product can be a source for dissolved pollutant for very long periods of time, resulting in the typical contamination plumes. Among chlorinated solvents, perchloroethylene (PCE), trichloroethylene (TCE) and cis-l,2-dichloroethylene (cis-l,2-DCE) and vinyl chloride (VC) are the main representatives. Currently, there are more than 10.000 sites in the Netherlands contaminated with chlorinated solvents (PCE and TCE), which are mainly located in urban area (SKB, 2008; BOEG, 2010).

2.2 Degradation of chlorinated solvents

Under specific biochemical conditions, chlorinated solvents are degraded by naturally occurring bacteria to substances that are harmless or less harmful to the environment (Kuchovsky and Sracek, 2007; Mulligan and Yong, 2004). Several biodegradation pathways are identified that occur under different environmental conditions: cometabolic oxidation, aerobic oxidation, anaerobic oxidation and reductive dechlorination (Bradley, 2000). Biodegradation rates are generally very low under aerobic conditions, especially for the highly chlorinated compounds. However, under anaerobic conditions, higher degradation rates are observed (Luijten, 2004; Pas, 2000). Due to a reducing reductive potential for the compounds with less chlorine substituents, studies report incomplete degradation, with accumulation of DCE or VC (Bradley, 2000; Vogel et al., 1987; Vogel and McCarty, 1985). Although reductive dechlorination is commonly observed in anaerobic aquifers contaminated with CHC, degradation rates highly depend on the biochemical conditions, including the presence of electron donor, contaminant, biomass and pH, redox conditions and temperature (Rectanus, 2007; Suarez, 2010).

2.3 Anticipated effects of ATES in contaminated aquifers

Operating an ATES system in a contaminated area increases the dynamics of the groundwater system. Seasonally, groundwater, including all dissolved and suspended substances, is transported between wells. Furthermore, water outside the extraction area is influenced due to increased dispersive mixing. Water is abstracted over a depth range of typically 20 to 50 meters in length. This leads to mixing of water from different depths and possibly different chemical composition. This mixing can induce precipitation and clogging of the wells, but also dilute dissolved substances (e.g. contaminants). The presence of pure product (DNAPL) may lead to a long period of delivery of contaminants to the aqueous phase and thereby increase the amount of dissolved contaminants (Zuurbier, 2013). Therefore, the presence of pure phase contaminants can be very important in determining the effect that ATES has on the presence and spreading of contaminants. However, in most cases it is very hard to determine where and how much DNAPL is present.

Besides effects of mixing, also temperature changes may influence microbial activity and biodegradation rates (e.g. van 't Hoff, 1884). As a rule of thumb, the degradation rate doubles for every increase of 10 °C (this is known as the Q10 rule; van 't Hoff, 1884). However, Hartog (2011) shows that the net effect of the temperature changes induced by low temperature ATES (which is currently the common case) with an energy balance is generally too low to significantly increase biochemical reaction rates. Storage of high temperature water (>25°C) and/or a net heating of the subsurface could however increase degradation rates.

3. COMBINATION CONCEPTS

To explore possible combinations of ATES and CHC remediation, different types of ATES systems (closed loop, open doublet, open recirculation, open monowell) have in theory been combined with twelve conventional and non-conventional remediation techniques (MMB, 2012). From the matrix of combinations a selection of the most promising combination concepts was made: 1) ATES and natural attenuation, 2) ATES and containment, 3) ATES and aboveground water treatment and 4) ATES and in-situ remediation.

If remediation of CHC by a combination of ATES and natural attenuation and/or containment is insufficient, a combination of ATES with aboveground water treatment or in-situ remediation can be an option. The most effective aboveground water treatment technique to remove CHC is by air stripping. This is a proven technology but the disadvantages are the duration of the treatment operation and the costs. Because of environmental restrictions, discharge of extracted groundwater on surface water may require iron removal which further increases the costs. Anaerobic water treatment techniques for CHC like nitrogen or vacuum stripping, which could make infiltration possible and iron removal unnecessary, could lead to lime precipitates due to degassing of CO_2 and are considered too costly to be viable alternatives.

An alternative to aboveground water treatment is the combination of ATES with in-situ remediation. In-situ remediation techniques use oxidizing, reducing or bacteriological active additives to degrade CHC. Oxidizing additives are not advisable because most ATES systems operate in anaerobic aquifers. By introducing oxygen, oxides like iron oxide are formed which can easily clog the ATES wells. Reductive additives that stimulate the biological degradation of CHC are more advisable. Slow release carbon sources like vegetable oil or chitin can be used to create a biological active screen. This screen will have to be installed perpendicular to the groundwater flow induced by the ATES system. Fast degrading carbon sources like alcohol (ethanol or methanol) can be added to the ATES infiltration wells or in special additive infiltration wells located close to the ATES infiltration wells. Adding reductive additives will stimulate the biological degradation of CHC but it could also lead to biofouling of the ATES wells. To prevent biofouling, vegetable oil and chitin will have to be applied at a safe distance of the ATES wells. Alcohol can be periodically dosed at high, sterilizing concentrations which may reduce clogging risks at the point of injection. Although precautions can be taken, it is yet unclear if afore mentioned actions will completely prevent well clogging.

By making use of bacteriological active additives the clogging problems of reductive, carbon rich additives could be overcome. The goal of adding carbon source is to stimulate the growth of dechlorinating bacteria. These bacteria will consecutively degrade CHC. Undesired side effect of the addition of carbon source is the growth of specific bacteria that may induce biofouling. The challenge is to increase the number of dechlorinating bacteria without the growth of biofouling inducing bacteria. This challenge can be met by growing a high density bacteria culture of dechlorinating bacteria and by adding this culture to the aquifer without the addition of a carbon source.

To test the lifetime and dechlorinating capacity of dechlorinating bacteria, Dehalococcoides (DHC), without the presence of a carbon source, lab tests have been conducted. CHC contaminated soil and groundwater samples (site description in section 4) were under anaerobic condition inoculated with 10 ml $1*10^7$ cells DHC/ml. Redox conditions during the test were iron to sulphate reducing. The concentration TOC increased after the inoculation from 6,3 mg/l to 18 mg/l and declined to < 10 mg/l after 20 weeks. The initial CHC concentration was 1000 µg/l cis-DCE. Because CHC degradation was faster than expected, at week four, eight and thirteen additional cis-DCE (1000 µg/l) was spiked. The results of the tests are shown in figure 1.

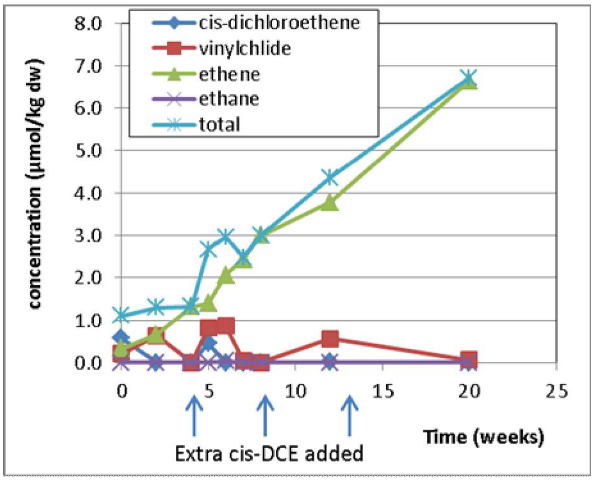

Figure 1: Degradation test: by adding only dechlorinating bacteria (no carbon source) CHC is completely degraded.

The test shows that under suboptimal iron to sulphate reducing redox conditions and with no carbon source added, CHC is fully degraded to harmless ethane. This was confirmed by additional q-PCR DNA and RNA analyses on DHC showing that the density of the DHC population (derived from DNA copies) and activity of the DHC population (derived from RNA copies) remains on the same level over 20 weeks.

Because the translucent high density dechlorinating bacteria culture can be grown at relatively low cost and the addition of these bacteria is not expected to increase clogging risks, the combination of in-situ CHC remediation with bacteriological active additives seems a promising remediation technique to combine with ATES.

4. CASE DESCRIPTION

4.1 ATES system description

The first combination concept for ATES and remediation in the Netherlands was realized in 2010 and named Sanergy (Slenders et al., 2010). The site is located in the city of Eindhoven in the south-east of the Netherlands and is situated at a former industrial site that is being redeveloped towards an area for working and living. Due to historical industrial activity, the site has been contaminated with chlorinated hydrocarbons and benzene. As part of the redevelopment, in 2010 an ATES system was installed in the contaminated aquifer with the goals to 1) reduce CO_2 emission from heating and cooling for connected buildings over 50%, 2) reduce spreading of the contaminants and 3) stimulate natural biodegradation (Slenders et al., 2010).

In its final form, the recirculation ATES system will consist of 23 wells that pump 2.8 million m^3 groundwater each year. Currently, 14 wells are installed and operational (Figure 2). The system is operated as a recirculation system (also known as mono-directional/ once-through-cooling). This means that extracted groundwater with ambient aquifer temperature is used for cooling in summer and heating in winter, without re-using stored energy as is done in bi-directional systems (Dickinson et al., 2009). The extraction wells are positioned in a ring around the contamination, surrounding the injection wells. This lay-out was chosen to reduce spreading of the contaminants by regional groundwater flow by 60-80%.

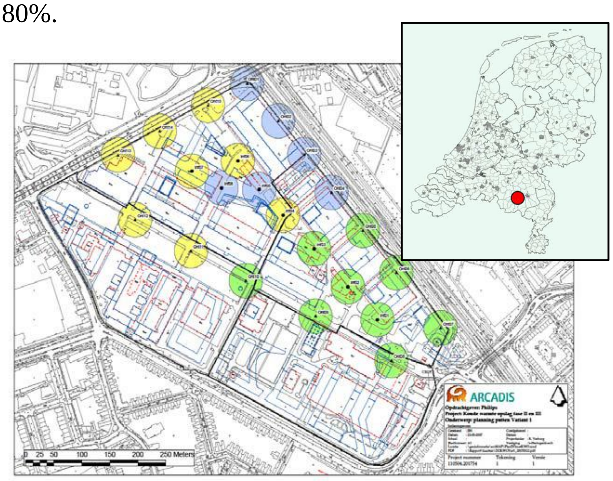

Figure 2: Schematic lay-out of ATES wells at the study site. Yellow and blue wells are currently operational; the green wells are expected to be commissioned in 2016.

4.2 Local hydrogeology and contamination

The wells of the ATES system are placed in an aquifer 27-90 m below ground level (bgl). The groundwater flow velocity in this aquifer is estimated to be 37 m/year in north-northeast direction. The aquifer is bounded at the bottom side by layers of silty sand and clay. The top of the aquifer is bounded by a 3 m thick clay layer which is however not present at all locations.

Table 1: Hydrogeological description

Depth [m bgl]	**Hydrogeological layer**	**Lithology**
0-3	Surficial layer	Loam, clay and very fine sand
3-24		Fine to very fine sand
24-27	1st aquitard	Clay
27-90	1st aquifer	Fine to course sand, occasional thin clay layers
90-150	2nd aquitard	Fine sand and clay
150-250	2nd and 3rd aquifer	

Benzene contamination is mainly limited to the phreatic groundwater and the upper meters of the 1st aquifer. Chlorinated hydrocarbons are found throughout the upper part of the 1st aquifer (Figure 3). In the lower part of the aquifer (60-90 m bgl) no contaminants were found within the system boundaries of Sanergy.

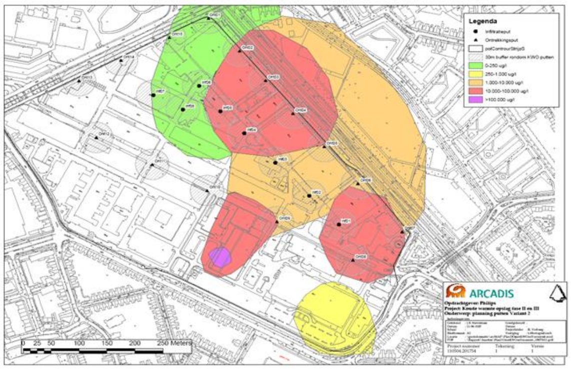

Figure 3: Distribution of CHC contamination at 30-60 m bgl. Colors indicate contaminant concentration levels with green (0-250 µg/L), yellow (250-1000 µg/L), orange (1000-10000 µg/L) and red (10000-100000 µg/L).

4.3 Measurements and analyses

To determine the influence of the ATES system, physical, chemical and biological conditions were monitored in 14 observation wells (Figure 4). Groundwater samples were taken at three moments in time: 15 September 2010, 14 April 2011 and 28 September 2011.

Physical monitoring

In spring 2010, 10 DIVERs were installed to monitor temperature and piezometric head several times per day. DIVER's were installed in observation wells 5a-30, 5a-44, 6-47, 29SH-30, 29SH-40, 29SH-50, 30SBP-40, 30SBP-50, 31SAN-40, 31SAN-50 (observation well locations are indicated with their number-bottom of filter screen). These physical measurements were used to determine where and when the local groundwater is influenced by the ATES system.

Groundwater sampling

To determine the chemical and biological composition, groundwater samples were collected using low flow-purging (0.2 L/minute) from a hose lowered to the well screen. Using a flow-through cell, pH, dissolved oxygen, redox potential, temperature and electrical conductivity were measured while purging. Care was taken to start sampling only after constant values for the above parameters were reached. Both filtered and unfiltered samples were collected under anaerobe conditions and conserved in the field when necessary.

Chemical analysis

To characterize redox state, concentrations were determined of nitrate, sulphate, sulphide, Fe^{2+} (as total iron after 0.45 µm filtration) and methane. Nutrient availability was determined from ammonium, ortho-phosphate and dissolved organic carbon. To investigate the effect of temperature on ion distribution, the following (additional) cat-ions were measured: Ca, K, Mg, Mn and Na; and an-ions: Cl and bicarbonate. The availability of metals were measured by ICP-OES (Al, Ca, Cu, Fe, Mg, Mn, Na, P, Pb, S, Sr, Ti, Zr) and ICP-MS (As, B, Ba, Cd, Co, Cr, K, Li, Mo, Ni, Sb, Tl, U, V, Y, Zn). To determine CHC and BTEX contaminant concentrations, the following species were analysed: PER (tetrachloroethene), TRI (trichloroethene), 1,1-DCE (only with TCA contamination), *trans*-DCE (*trans*-dichloroethene), CIS (*cis*-dichloroethene), VC (vinylchloride), ethene/ethane, chloroethane (only with TCA contamination), 1,1-DCA (dichloroethane), 1,2-DCA (dichloroethane), 1,1,1-TCA (trichloroethane), benzene, toluene, ethylbenzene, xylene and naphthalene.

Hydrogen concentrations are indicative for the available electron donor for Dehalococcoides (DHC), the bacterium that is known to be able to degrade CHCs under anaerobic conditions to ethane (e.g. Friis et al., 2007). Therefore hydrogen concentrations were measured while sampling the groundwater.

Additional to the contaminant concentrations, isotope ratios ($^{12}C/^{13}C$) were determined for the chlorinated compounds. While dilution does not affect the isotope ratio, biodegradation is more efficient for the lighter (^{12}C) isotopes (Abe et al., 2009). Using isotope ratios, in principle it is possible to discriminate between dilution and biodegradation, however interpretation of field data is complicated because the isotope ratio for VC changes as a result of both degradation of the parent-product CIS as degradation of VC itself. Also mixing with contaminated groundwater with another isotope ratio (for example where some degradation has taken place) can make analysis more difficult.

Biological analyses

Using Q-PCR the microbiology was characterized by total bacteria/ archea, functional genes and specific micro-organisms. Considering different microbiological degradations pathways the following micro-organisms and genes were analysed: 1) reductive dechlorination: Dehalococcoides, tceA, vcrA and bvcA, 2) micro-aerophilic oxidation of VC: EaCoMT, etnE and etnC and 3) micro-aerophilic oxidation of CIS: Polaromonas.

4.4 Physical monitoring

Operation of the ATES system started in March 2010. While exact flow rates of each ATES well are not available, results of the continuous physical monitoring were used to determine where and when water injected by the ATES system reaches an observation well. Ambient aquifer temperature is around 13 °C and injection temperatures range between 8 and 18 °C depending on the need for cooling or heating. In August 2011 temperature fluctuation up to 0.5 °C are observed in 29SH-50 and simultaneously in 29SH-30 with lower amplitude (0.1 °C). If these temperature fluctuations are induced by the ATES system and considering a thermal retardation around two in these type of sandy aquifers (Thorne, 2005), this suggests that groundwater at this location has been fully replaced by water injected by the ATES system. In the observation points to the south and west: 31SAN and 30SBP (Figure 4) no temperature signals were observed and therefore it is uncertain whether infiltration water has already reached these observation points. In May 2011 a 0.1 °C temperature decrease was observed in 5A-44, which could indicate the arrival of cold water infiltrated during winter 2010/2011.

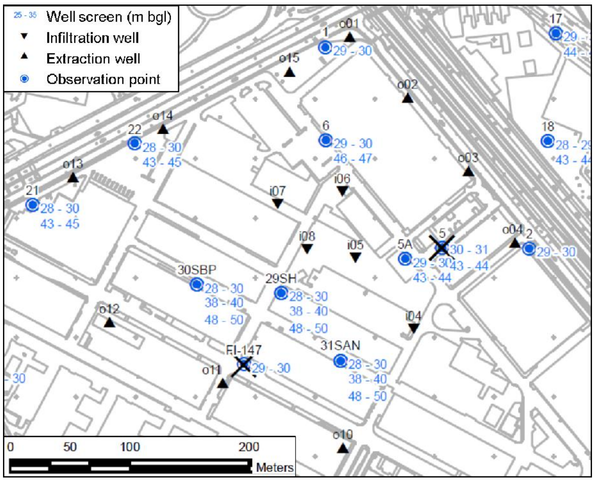

Figure 4: Location of ATES wells and observation wells. Blue numbers indicate the depth range (m bgl) of the filter screens at each location.

4.5 Geochemical effects

The water injected by the ATES system is a mixture of groundwater from different depths and different extraction wells and could therefore have different composition than the resident groundwater at the observation wells. Therefore, chemical analysis of groundwater samples can give additional information on the spreading of injected water. Geochemical conditions are stable during the first two observation rounds, suggesting that injected water has not yet reached the observation wells. In the third observation round, however, significant changes are observed in pH, redox potential, electrical conductivity and concentrations of Cl^-, Na^+, TOC, H^+ and metals, mainly chromium, copper and nickel. Due to the historic activity on the location it is likely that different pockets of groundwater, each with different chemical and biological characteristics, have been mixed by the ATES system. The changes are most pronounced in 29SH, but based on changes in metal concentrations it is likely that injected water has also reached 5A, 30SBP and 31SAN. Conditions in observation well 6 are stable indicating that injected water has not yet reached this location at the time of the third observation round. A complete overview of monitoring results is available in MMB, (2012b). The most noticeable changes in 29SH-30 and 29SH-50 include: a pH change from 7 to 8, and for the middle filter (29SH-40) even to 10, a doubling in electrical conductivity, fluctuating redox potential, increase in Na^+ and a strong increase in Cl^- concentrations. The changes in Chloride concentrations (100 – 300 mg/L) are too large to be attributed to dechlorination of CHC's, but rather should be explained by mixing and spreading of groundwater with higher Cl^- concentrations. 29SH also shows an increase in dissolved organic carbon (TOC) and hydrogen concentrations.

4.6 Microbiology

Biological measurements where only taken in the first (15 September 2010) and third (28 September 2011) observation rounds. The first groundwater monitoring round was conducted only 5 months after start-up of Sanergy, and the results are regarded as representative for the undisturbed situation. In the third measurement round, one and a half year after implementation of Sanergy, biological measurements show an increased biological activity. In monitoring well 29SH (which is expected to be the first monitoring well to be influenced by Sanergy) the concentrations of dechlorinating bacteria *Dehalococcoides Ethenogenes* (DHC) increased with a factor 1000. Also enzymes that encode for VC-reductase (vcrA and bvcA) increased 100-1000 fold (Figure 5). This indicates an increase in biological activity of bacteria that perform degradation of the contaminants. The simultaneously measured increase of hydrogen is an indication that the conditions for degradation have improved in the period September 2010 – September 2011 (Figure 6). Since hydrogen is extremely rapidly consumed after its production, and therefore not transported in the subsurface it is likely that in the direct surroundings of monitoring well 29SH biodegradation of chlorinated solvents occurs.

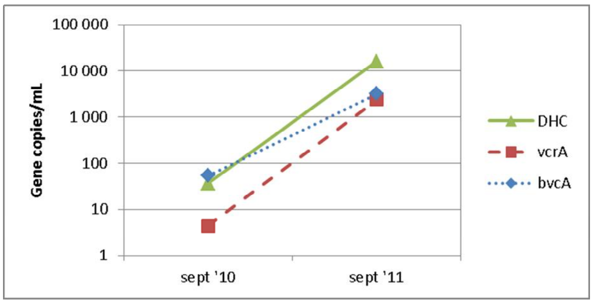

Figure 5: Increase in dechlorinating bacteria Dehalococcoides Ethenogenes (DHC) and enzymes that encode for VC-reductase (vcrA and bvcA) at 29SH-50 (50 m bgl).

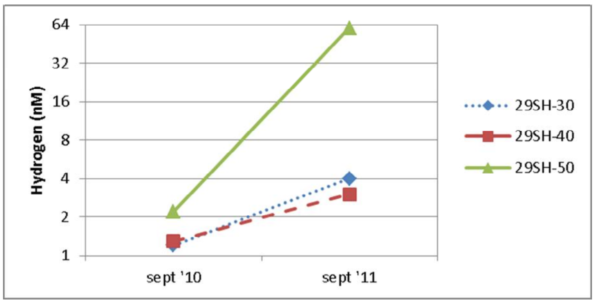

Figure 6: Increase in H_2 in observation well 29H.

4.7 Contaminants

Concentrations of contaminants (TRI, CIS and VC) fluctuate and do not allow discriminating between effects of biodegradation or mixing and transport of water with different contaminant levels. Results of the $^{12}C/^{13}C$ isotope analysis are shown in figures 7 and 8. Because the lighter isotopes are preferentially degraded, this should result in increasing ^{13}C fractions in the mother compounds and decreasing ^{13}C fractions in the daughter products. Interpretation is however complicated by the fact that isotope ratios of CIS and VC (the predominant compounds found here) can both increase by degradation and decrease by formation due to degradation of their parent products (respectively TRI and CIS). The only location where TRI is found is at 5A-30, at this location ^{13}C fraction of CIS is decreasing, which indicates that more CIS has formed than degraded. In the deeper observation well 5A-44, ^{13}C is increasing, indicating that at this lower level more CIS has degraded than formed. At other locations, no TRI is present, and therefore an observed decreasing ^{13}C fraction will be caused by induced groundwater flow. It is not possible to distinguish an unambiguous trend for VC. Although isotope ratios at individual locations do not show a general trend, comparison of the range in isotope ratios of TRI (-25 to -22 ‰) and CIS (-20 to +20 ‰) suggest that more CIS has been degraded than formed in this area in the observed period. At least, when all TRI would have been degraded, the isotope ratios in the daughter product CIS should have been returned to the original values in the mother compound. Now that the isotope values of CIS prove to be higher, significant and predominant degradation of CIS to VC is confirmed.

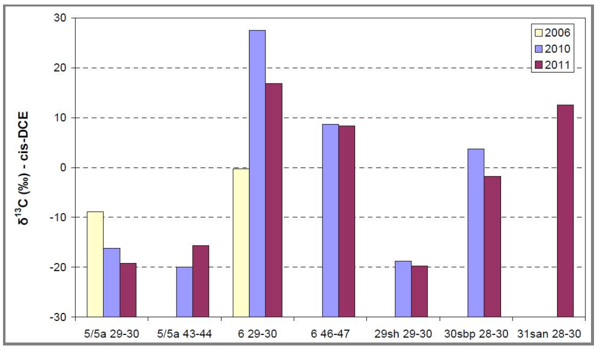

Figure 7: Comparison ^{13}C fraction of CIS between 2006, 2010 and 2011.

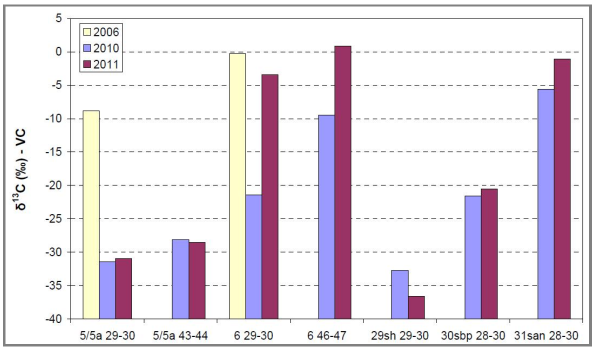

Figure 8: Comparison ^{13}C fraction of VC between 2006, 2010 and 2011.

5. DISCUSSION

Although both ATES and remediation are proven technology when practiced independently, the combination of both is not straightforward. The flow rate of an ATES system can be more than 100 times as large as the flow rate of a pump and treat system. Treating all the extracted groundwater aboveground would therefore be far too expensive. It would be more logical to treat the groundwater in situ. However standard in situ techniques often use additives to create the optimal conditions for biodegradation of the contaminants. The induced (bio)chemical reactions can cause clogging of the ATES wells which would threaten the long term operation of the ATES system.

In Eindhoven the first combination project was started in 2010, consisting of a recirculation system in CHC contaminated groundwater. The remediation process can be considered natural attenuation. The first monitoring results show that in the area of influence the conditions for biodegradation have significantly improved, presumably due to groundwater mixing and the increased groundwater dynamics caused by the ATES system. It is yet uncertain if these positive results will last on the long term. The positive results can also be explained by the effects of well installation. An increase in microbiological activity after installing groundwater wells has been found before and can be attributed to the associated disturbance of the subsurface (Heidemij, 1986; Van der Weiden and Willemsen, 1991). These effects proved to be temporary. It is therefore essential to continue monitoring to find out how the chemical characteristics and biological activity will develop in the future.

It has to be noted that it is unclear whether the principle of natural attenuation can be expected to be successful at other sites. Because of the specific industrial history of the location, the presence of unspecified contaminations in the aquifers and the dependency on the hydrogeochemical conditions in the aquifer, it is expected that the observed changes in Eindhoven are site specific and cannot be generalized. In measurements in the city centre of Utrecht, where several ATES systems have operated in CHC contaminated groundwater for more than 10 years (not combined with intentional remediation efforts), no evidence was found for any positive effects (MMB, 2012b). The lack of (indications for) positive effects might either be a consequence of less favourable hydrogeochemical conditions in the aquifer, or shows that the positive effects fade away in time.

For the cases where active stimulation of (bio)degradation of CHC is needed, the most promising concepts were analyzed. Natural attenuation might work at some locations, but the effectiveness on the long term is yet uncertain. Containment in combination with aboveground treatment will work, but will be less beneficial then a combination with in-situ remediation.

Options have been studied to prevent problematic (bio)fouling of the ATES wells when using in-situ remediation techniques. Slow release carbon sources like vegetable oil or chitin can be used to create a biological active screen at a safe distance from the ATES wells. Periodical injection of highly concentrated alcohol might also be an interesting option. However, it is yet unclear if these options will be sufficient to prevent well clogging. The addition of (only) dechlorinating bacteria is considered the most promising option. Laboratory experiments have shown that these bacteria remain active for at least 20 weeks, even if the hydrogeochemical conditions are not optimal. Furthermore the addition of these bacteria is not expected to increase clogging risks. As a next step field experiments should be undertaken to further investigate this option.

Beside the opportunities, one has to be aware that the application of ATES in CHC contaminated groundwater may also adversely affect groundwater quality. When ATES is practiced near DNAPL it can lead to enhanced dissolution, potentially resulting in an increase in both the volume of contaminated groundwater and the total amount of dissolved contaminant (Zuurbier et al., 2013). It is therefore important to assess both the opportunities and the risks for each location considered.

6. CONCLUSIONS

Combining ATES and groundwater remediation can be beneficial for both. From the ATES point of view it opens opportunities for application in contaminated areas. From a remediation point of view it could help to accelerate groundwater quality improvement.

Several methods to combine ATES with remediation are discussed. The most promising combination concepts are considered to be 1) ATES and natural attenuation, 2) ATES and containment, 3) ATES and aboveground water treatment and 4) ATES and in-situ remediation.

Monitoring of the first ATES system in the Netherlands that is, on purpose, constructed in an aquifer contaminated with chlorinated hydrocarbons, shows that conditions for reductive dechlorination improve at one of the monitoring wells. Continued measurements should show whether degradation continues and ATES really can stimulate biodegradation of groundwater pollution on the long term. So far, no negative effects of the CHC contamination on the functioning of the ATES system are observed.

Additional lab experiments with aquifer material from the ATES site demonstrate that after inoculum with dechlorinating bacteria full degradation of CIS and VC to ethene is achieved within four weeks. Further studies are required to show the effectiveness of this method under field conditions.

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