

Project:

ASR application for Domestic Fresh Water Supply in the Vietnamese Mekong Delta

Report No 1:

Concept for initial water quality assessment and water monitoring plan during implementation for ASR pilot in Hau Giang province

**WP1: ASR Site development
Water Quality assessment**

**WP2: Monitoring
Monitoring Plan**

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**PARTNERS
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Disclaimer

The opinions expressed in this report are those of the authors and do not necessarily reflect the views of the institutions supporting the project.

Summary

This report describes general water quality considerations during Managed Aquifer Recharge (MAR) operation. These are based on three main aspects:

- Operational risk due to clogging
- Risk of contamination of the ambient groundwater and aquifer
- Risk of water quality deterioration during storage e.g. due to redox reactions

Surface water quality in Hau Giang province is affected by domestic and industrial (un-)treated wastewater discharges as well as agricultural runoff and discharges from aquaculture. The water quality of the treated water meets the Vietnamese technical guidelines for drinking water. Ambient groundwater quality is variable ranging from fresh to brackish and may contain elevated concentrations of iron, ammonium and coliforms.

The main recommendations are:

1. Injectant should be of high quality to prevent chemical, mechanical and biological clogging.
2. Injectant should meet the Vietnamese technical guidelines for groundwater and should not be lower than the ambient groundwater quality.
3. Injectant should be sourced from the fully treated surface water before disinfection to prevent the development of disinfection by-products.
4. Before pilot development, water samples should be taken from the treated surface water and ambient groundwater of the selected site during rainy and dry seasons and analysed for a broad range of parameters.
5. Sediment sampling during the drilling of the monitoring well and further data acquisition is required to estimate geochemical reactions in the underground.
6. During operation, water quantity monitoring should be done to assess the recovery efficiency and well performance , and provide data for the cost-benefit analysis.
7. During operation, water quality monitoring should be done for parameters critical for clogging, for potential contamination of the groundwater and for geochemical reactions in the subsoil relevant for recovered water quality.

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Abbreviations

AOC	assimilable organic carbon
ASR	aquifer storage and recovery
ATP	adenosine triphosphate
BGR	Federal Institute of Geosciences and Natural Resources (Germany)
BOD	biological oxygen demand
BPA	bisphenol A
CEC	cation exchange capacity
COD	chemical oxygen demand
CRMGG	"Climate resilient management of groundwater and geohazards" project
DBPs	disinfection by-products
DO	dissolved oxygen
DOC	dissolved organic carbon
DONRE	Department of Natural Resources and Environment
DWRPIS	Division of Water Resources Planning and Investigation in the South
<i>E. coli</i>	<i>Escherichia coli</i>
EC	electric conductivity
EDCs	endocrine disrupting components
Eh	redox potential
GrwV	Grundwasserverordnung (German Groundwater Ordinance)
HAAs	haloacetic acids
HAWASUCO	Hau Giang Water Supply and Sewerage and Urban Construction Joint Stock Company
IoT	internet of things
LoRaWAN	low range wireless area network
MAR	managed aquifer recharge
MFI	membrane filtration/fouling index
NAWAPI	National Center for Water Resources Planning and Investigation
NOM	natural organic matter
PAC	poly aluminium chloride
pe	electronic activity
PFAS	per- and polyfluoroalkyl substances
PFW	Partners for Water
PhACs	pharmaceutically active compounds
SAR	sodium adsorption ratio
SI	saturation index
TDS	total dissolved solids
THMs	trihalomethanes
TOC	total organic carbon
TSS	total suspended solids
UV	ultraviolet
VEI	Vitens-Evides International
VIWAT	"Integrated Solutions for Sustainable Development in the Mekong Delta - Land, Water, Energy and Climate" project
WP	work package
WQI	water quality index
WWF	World Wildlife Fund
XRD	X-ray diffraction
XRF	X-ray fluorescence

1. Introduction

1.1. Objective of the project

In the Partner for Water (PFW) project “ASR application for Domestic Fresh Water Supply in the Vietnamese Mekong Delta”, the ambition is to build an Aquifer Storage and Recovery (ASR) pilot location in Hau Giang province with the objectives of:

- successfully developing and demonstrating a functioning infiltration system in the local context based on the ‘proof-of concept’ (from the Netherlands, Chile, and other countries),
- showcasing the capacity of ASR to provide sufficient freshwater in times of shortages,
- addressing local and federal authorities in bringing about a legal framework within which ASR is applicable, and
- creating a coalition within the interested stakeholders towards a clear upscaling path.

The funding of this project was approved in June 2023 by the PFW and the project is intended to run until mid-2025. The consortium under the PFW scheme consists of Deltares, VEI, WWF Vietnam and HAWASUCO with BGR and DWRPIS as cooperating and co-financing partners.

1.2. Goals and outline of document

The project encompasses three main work packages (ASR site development, Monitoring and Upscaling) with numerous tasks (Figure 1), of which two will be addressed in this document, namely

1. Water quality assessment before implementation, and
2. Pilot monitoring and assessment during implementation.

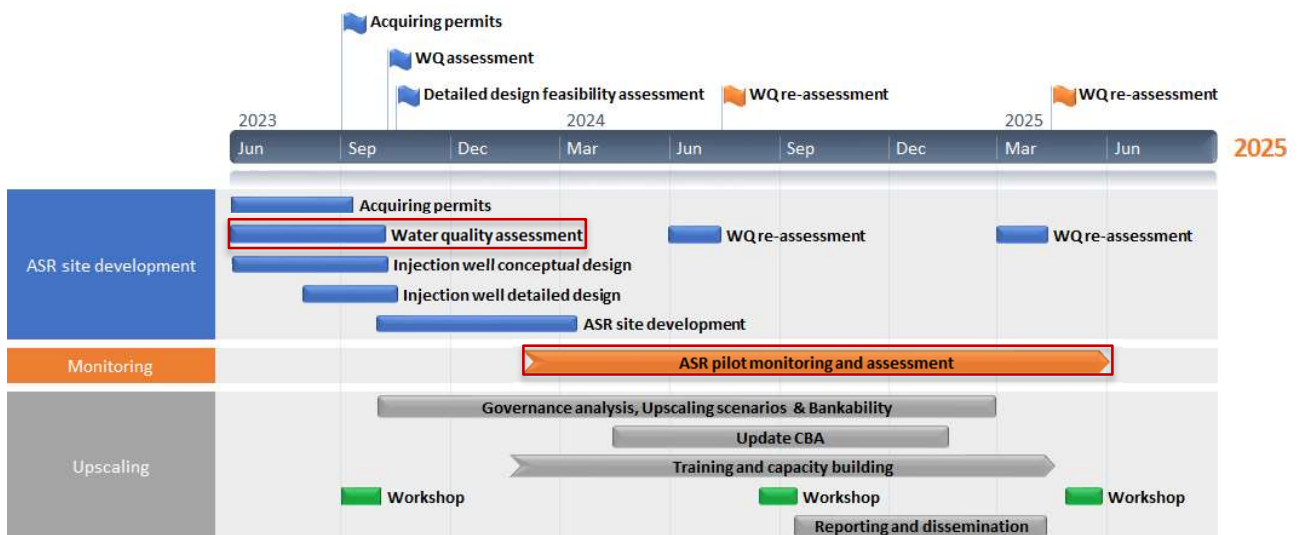


Figure 1: Project schedule

Report No 1: Concept for initial water quality assessment and water monitoring plan during implementation for ASR pilot in Hau Giang province

This report provides the basis for the development of a site-specific monitoring plan once one of the potential sites has been selected.

It first describes general water quality considerations applicable to MAR systems, then it provides an overview of the water quality in Hau Giang province. After a brief overview of international MAR regulations and critical water quality considerations, recommendations for an initial water quality assessment and further data acquisition are provided. Lastly, recommendations are done for continuous monitoring of water quantity and quality during the implementation of the pilot.

After each section, a brief summary is provided for a quick overview of the most relevant points.

2. General water quality considerations

Clogging can be a major issue in ASR operations reducing recharge rates (Martin 2013), eventually leading to failure of the system, and is hence the biggest risk to a successful operation (Jeong et al. 2018, Martin 2013, Pyne 1995). There are four forms of clogging (Dillon et al. 2022):

- **physical** clogging due to the recharge of suspended solids or clay swelling when fresh water is recharged to brackish aquifers containing sodium montmorillonite,
- **mechanical** clogging e.g. due to entrapped air and gas binding
- **chemical** clogging from geochemical reactions leading to precipitation e.g. of iron and manganese hydroxides due to the infiltration with oxygenated water,
- **biological** clogging due to the growth of microorganisms stimulated by nutrients and organic matter.

The presence of total suspended solids (TSS), organic matter, nutrients and organic pollutants poses a potential problem for aquifer storage and recovery, if source water treatment is not sufficient to reduce the concentrations of these parameters.

High loads of **suspended solids** have to be removed before aquifer recharge to prevent mechanical clogging of the well. Apart from their clogging potential during infiltration or injection, their importance arises from the binding of other pollutants (e.g. nutrients, hydrophilic organics, trace metals) mainly with the finer fraction of particulates (Deletic and Orr 2005, Dempsey et al. 1993, Mikkelsen et al. 1997). While coarser particles are likely to be settling out before use in ASR schemes, the finer, more contaminated, fraction is likely to be injected into the aquifer and not to be recovered, as observed at Andrews Farm (Pavelic et al. 2006a). Particulate-facilitated transport is therefore a potential pathway for contaminants to be distributed in the aquifer and should not be neglected (Dempsey et al. 1993, Liebens 2001).

Nitrate and **dissolved oxygen** cause oxidation in the previously anoxic aquifer and change redox conditions significantly around the recharge well. Nitrate is highly soluble during ASR and does not attach readily to the aquifer matrix. Dissolved oxygen and nitrate are the major electron donors injected into the aquifer, stimulating microbial activity (posing the risk of biological clogging (Jeong et al. 2018, Kalwa et al. 2021)) and redox reactions. Even with very highly treated water, dissolved oxygen would normally not be removed and would therefore lead to changes in redox conditions in the aquifer.

Natural organic matter (NOM) (as indicated by total organic carbon –TOC–, and dissolved organic carbon –DOC) has to be removed before aquifer recharge to prevent biological clogging and groundwater contamination (Hägg et al. 2021). Clogging might not become an issue during short-term experiments, but should be considered for potential upscaling of the set up and long-term infiltration. The degradation rate of NOM varies widely. In general, smaller molecules with carboxylic and phenolic groups are favourable for biodegradation while large aromatic molecules are more stable (Howard 2000). Fulvic acids are soluble over a wide range of pH, while humic acids are insoluble at lower pH. Humic molecules are most elongated and flexible at high pH, low salinity and low NOM concentration, while they form spheres at higher salinity (0.05 M), in the presence of trivalent cations and at high NOM concentrations (Ghosh and Schnitzer 1980). This behaviour plays an important role for cation and metal binding and pH buffering. Depending on the properties of the organic molecule, pH and surface properties of the clay or oxides, NOM can adsorb to clay minerals and oxides via a range of mechanisms (Jardine et al. 1989, Stevenson

1994). The usually positive surface charge of the underlying clay or oxides thus changes to negative surface charge and thereby influences the cation exchange capacity (CEC), sorption of other organic molecules and colloidal stability of the particles (Kretzschmar and Sticher 1997, Stevenson 1994).

NOM in the injectant is the major driving force for **microbial growth**. Its oxidation acts as an energy source for microorganisms and controls the **redox status** especially near the injection well. In the theoretical sequence of subsurface reduction reactions from a highly-oxidizing to highly-reducing conditions, the dominant redox sensitive species are oxygen, nitrate, manganese (IV), iron (III), sulphate and methane (Stumm and Morgan 1995). In the natural subsurface environment, these processes will be partly taking place simultaneously due to heterogeneities of the matrix, distribution of reactants and microorganisms, meaning that there is usually no equilibrium between all redox couples in the groundwater (Lindberg and Runnells 1984). Meaningful redox potential (Eh or pe) values are therefore very difficult to obtain (Chapelle et al. 1995). Groundwater is typically low in both particulate and dissolved organic carbon (<0.5 mg-C/L) (Malard and Hervant 1999, Pabich et al. 2001). The recharge with organic-, oxygen- and nutrient-enriched injectant therefore is a major disturbance to the system (Bahr et al. 2002). Around the well (<10 m) where particulates are entrained (Greskowiak et al. 2005, Skjemstad et al. 2002), a so-called "treatment" or "proximal" zone with highly elevated microbial activity develops. Here, dissolved oxygen is eliminated within a few days. It is then followed by nitrate reduction, which lowers the redox potential from +300 to +400 mV to -200 to -400 mV (Vanderzalm et al. 2006), triggering other geochemical reactions such as the degradation of organic contaminants, mineral dissolution/precipitation, sorption and pH changes that influence the quality of the recovered water (Dillon et al. 2005).

Dissolution of minerals, especially carbonates, is induced by undersaturation with low ion activity in the recharge waters, especially by highly treated injectants (Johnson et al. 1999). The decrease of pH due to release of CO₂ during organic matter degradation also accelerates dissolution of carbonates (Herczeg et al. 2004). The interplay between oxidative/reductive changes induced by ASR and dissolution/precipitation reactions with minerals can be complex (Rafiq et al. 2022). The injection of oxygen into an anoxic aquifer with dissolved Fe and Mn will lead to the **precipitation of amorphous (oxy)hydroxides**, while reductive dissolution of Mn- and Fe-oxides starts after oxygen is completely depleted and nitrate is being reduced (Stumm and Morgan 1995). Both reactions occur in ASR operations close to the injection well, where the injected organic matter induces redox and pH changes (Greskowiak et al. 2005). While the precipitation of minerals will lead to co-precipitation and adsorption of trace metals, their dissolution will in turn mobilise previously incorporated or adsorbed trace metals (Antoniou et al. 2012, Bahr et al. 2002).

Pyrite oxidation is of special importance as it releases mobile oxyanions such as arsenate and selenate and mobile cations such as nickel and zinc. It also significantly reduces the pH due to the formation of sulphuric acid in solution, which can mobilise other adsorbed metals. Multiple ASR sites have experienced a decrease in recovered water quality due to geogenic metal mobilization related to pyrite oxidation (Brown et al. 2006, Rafiq et al. 2022). Over a number of ASR cycles, these effects seem to be decreasing and shifting towards the buffer zone of the ASR scheme

In general, **organic contaminants** with long half-lives and low to moderate soil sorption capabilities would be of highest concern as their properties enables their transportation to groundwater and they have limited degradation in the aquifer. The focus should be on previously detected pesticides in drinking water and surface water samples.

Trace organic contaminants like endocrine disrupting components (EDCs), pharmaceutically active compounds (PhACs), phthalates and bisphenol A (BPA) have been detected in treated effluents at concentrations usually in the ng/L range (Toze 2006). They are spreading into a wide range of aquatic environments (Heberer 2002) and might also be present in Hau Giang's canal water due to the discharge of untreated wastewater. They comprise a series of complex organic contaminants with varying chemical properties leading to a wide range of behaviours (Mackay and Boethling 2000). Experiments with five different EDCs showed that some degraded at different rates under aerobic conditions, while all of them were persistent under anaerobic conditions (Ying et al. 2003). Field experiments with six different PhACs showed that while four of them were retarded, two of them were very mobile (Oppel et al. 2004). It is likely that the more persistent components would be attenuated by sorption to the matrix, which would increase their residence time and hence increase the chances of degradation (Mackay and Boethling 2000).

For **Pesticides** it was shown that changing redox conditions affect transformation routes and therefore mobility of an array of compounds (Vink and van der Zee 1997).

Other **emerging pollutants** like per- and polyfluoroalkyl substances (PFAS) (Cáñez et al. 2021, Page et al. 2019) as well as "**novel entities**" (Steffen et al. 2015) like microplastics are coming into focus as well. PFAS are a large, complex group of synthetic chemicals with multiple fluorine atoms attached to an alkyl chain that have been used in consumer products around the world like water resistant fabrics, grease-resistant paper, non-stick cookware, stain-resistant fabrics etc. (Prevedouros et al. 2006). PFAS are of concern due to their persistence in the environment and potential health effects on humans. They bioaccumulate in humans and act as hormone-disrupting chemicals. They are also carcinogenic and can cause liver and kidney damage (developmental toxicity, immunotoxicity, hepatotoxicity) (Lau et al. 2007). Their presence has been shown in rainwater at levels exceeding drinking water standards (Kallenborn et al. 2004, Loewen et al. 2005), and they are also reaching surface waters (Xiao 2017) through industrial discharges, landfills and wastewater treatment sludge (Prevedouros et al. 2006). They are not removed by common water treatment processes (Yu et al. 2009), but require the use of activated carbon, nanofiltration, reverse osmosis or advanced oxidation processes (Arias Espana et al. 2015, Kucharzyk et al. 2017).

Microorganisms are problematic regarding drinking water supply, but the attenuation in the aquifer and the water treatment from HAWASUCO should be able to cope with potential recharge and recovery of microorganisms. Once injected, the inactivation of pathogens in the subsurface can be achieved by adsorption, filtration, oxidation, degradation by native groundwater microorganisms and die-off (Foppen and Schijven 2006, Pitt et al. 1999). The main factors favouring long-term survival of pathogens in the ground are small size (reduced straining especially for viruses), soluble organics (reduced adsorption), low-oxygen levels (reduced oxidation), low temperature, high pH and low salinity (prolonged life span) and the native microbiota (John and Rose 2005). Monitored ASR sites achieved complete removal of pathogens at storage times longer than 90 days (Dillon et al. 2005).

If treated water with **residual chlorine** is used for injection, the formation of carcinogenic **disinfection by-products** (DBPs) like trihalomethanes (THMs) and haloacetic acids (HAAs) can occur during chlorination of waters containing natural organics acids and bromide or can form in-situ when residual chlorine comes in contact with dissolved organic matter in the aquifer (McQuarrie and Carlson 2003, Pyne 1995). It has been shown that HAAs are highly degradable under aerobic conditions (Thomas et al. 2000) found near the injection well and are usually eliminated in a few days, and were not found in the recovered water of any monitored site (Dillon et al. 2005, Pavelic et al. 2005). THMs, on the other hand, are degraded under anaerobic conditions and elimination rates are highly variable depending on the redox state of the aquifer system (Dillon et al. 2005, McQuarrie and Carlson 2003). They are typically attenuated within a few weeks of storage due to anaerobic microbial activity, but might be very persistent in aerobic unconfined ASR systems (Bloetscher et al. 2014, Pavelic et al. 2005). As DBPs degradation is enhanced by microbial activity, very low organic carbon concentration (<1 mg/L) in the recharge water can increase their half-life significantly (Pavelic et al. 2006b). The formation of DBPs could be avoided when disinfection of injectant is undertaken with ultraviolet (UV) or is used in post-treatment rather than pre-treatment, as ASR provides the opportunity for natural pathogen removal (Pyne 1995).

SUMMARY:

As outlined above, the water quality of the injected water needs to be of high quality for two main reasons:

- (1) **Prevent clogging** and hence failure of the ASR system. Main concerns for clogging are suspended solids, oxygenated nutrients, organic matter and oxygen. It also depends on the grain size distribution of the aquifer and the gravel pack.
- (2) **Prevent contamination of the ambient groundwater**, which is used for domestic water supply. Main concerns for contamination are the mobilization of geogenic trace metals through redox reactions in the aquifer, the development of disinfection by-products as well as trace organic contaminants that are not removed in the surface water treatment plant and would end up in the aquifer.

The water quality of the recovered water is influenced by

- (3) **Redox reactions** triggering dissolution and/or precipitation of minerals. The injection of oxygen is a major driver of these reactions. The mobilization of geogenic trace metals depends on their concentrations and speciation in the aquifer matrix.
- (4) **Oxidation of organic matter from the aquifer**. The triggered redox reactions depend on the concentration and composition of the organic matter in the aquifer matrix.
- (5) **The formation and attenuation of disinfection by-products.**

It is therefore also important to know more about the aquifer matrix to predict potential geochemical reactions and potential for clogging.

3. Overview of water quality in Hau Giang province

Water quality assessment: Studying suitability of the available treated water and potential additional requirements prior to infiltration. This is a milestone (go/no-go decision). If the requirements with or without some incremental changes are not met, the continuation of the pilot is jeopardized.

3.1. Site set-up

The pilot site will be on one of the premises of the local water supply company HAWASUCO who is the main partner of the project. Four sites were assessed during the site selection process. Two of the potential sites (1-Long My and 2-Nga Bay) have a surface water treatment plant on the premises, while the other two potential sites (3-Cai Tac and 4-Long Thanh) receive treated surface water from Nga Bay via a pipeline and mix it with groundwater extracted at the site (Figure 2).

Source water for the ASR pilot should be sourced from treated surface water and should not contain extracted groundwater, as this would decrease the efficiency of the pilot in enhancing groundwater reserves. Accordingly, if the sites Long Thanh or Cai Tac would be selected, a separate water storage tank containing only treated surface water should be installed at the site to prevent the use of groundwater in the pilot.

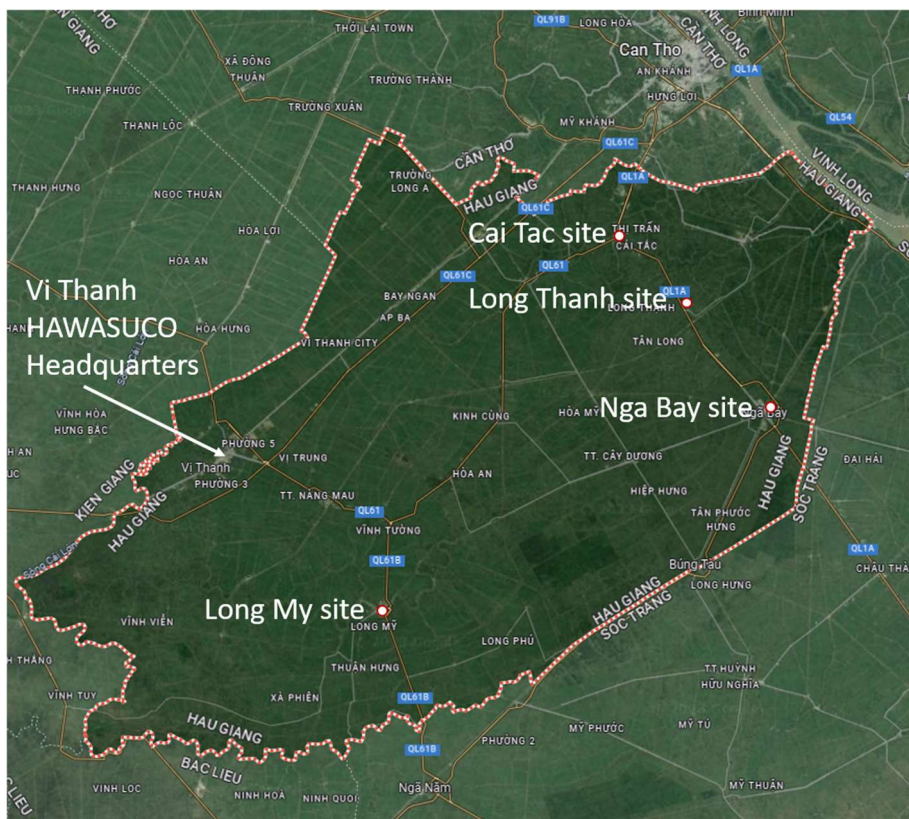


Figure 2: Four potential sites in Hau Giang province (1-Long My, 2-Nga Bay, 3-Cai Tac, 4-Long Thanh) and HAWASUCO headquarters (HQ) in Vi Thanh

3.2. Raw surface water quality in Hau Giang province

The surface water treatment system of HAWASUCO consists of a surface water intake from a canal. The exact raw surface water quality is currently unknown, but there are a number of water quality issues, especially when the water in the canal is kept stagnant to prevent saline intrusion. Common organic and microbiological pollution is induced by the discharge of domestic and industrial wastewater, runoff from agriculture and aquaculture as well as from solid waste disposal and fuel residues from ships. Excess nutrients are introduced by fertilizer use.

Surface water monitoring of 44 stations sampled five times in 2021 by Hau Giang DONRE (Thuan 2022) found that out of 15 parameters, **TSS** (mean 55 mg/L), biological oxygen demand **BOD** (mean 9.0–11.2 mg/L), chemical oxygen demand **COD** (mean 19.8 mg/L), **PO₄** (mean 0.13–0.18 mg/L), **coliforms** (mean 9500 MP/100 mL) and **iron** (mean around 1 mg/L) far exceeded the allowable limits of the national technical regulation for surface water quality (QCVN 08-MT:2015/BTNM, column A1). NH₄ (mean 0.2–0.4 mg/L) and NO₂ (mean 0.05–0.08 mg/L) exceeded the limits only slightly and not at all locations. Dissolved oxygen (DO) (mean about 3.3 mg/L) and NO₃ (mean about 0.5 mg/L) were commonly low, indicating the impacts of eutrophication. Many stations were hence classified as of “bad” water quality (WQI 26–50). No inorganic or organic trace elements were analysed. While salinity was higher in the dry season, it was found in Kien Giang province that problems related to TSS, organic matter and bacteria in water bodies tended to be more severe in the rainy season due to an increase in surface runoff containing pollutants from upstream and waste from regional human activities (Nguyen et al. 2022).

Chau et al. (2015) measured 15 frequently used **pesticides** in 260 samples from different water sources in An Giang and Can Tho province and found pesticide concentrations were highest in canal waters, but were also found in groundwater (5 out of 22 samples), harvested rainwater (all 6 samples) and bottled water (9 out of 18 samples). In surface water samples, isoprothiolane was most frequent (97%, mean 0.55 µg/L, max 8.49 µg/L) followed by fenobucarb (91.2%, mean 0.17 µg/L, max 2.32 µg/L), and fipronil (83.4%, mean 0.15 µg/L). Propiconazole (39.2%, mean 0.5 µg/L), pretilachlor (71.8%, mean 0.21 µg/L), and quinalphos (78.5%, mean 0.17 µg/L) also contributed to total pesticide loads. Up to 12 different pesticides were detected together creating combination toxicity. The Vietnamese National Technical Regulation for Domestic Water Quality (QCVN 01:2009/BYT (MoH 2018)) does not consider any of these studied pesticides. The European Commission guideline value for concentration of total pesticides in drinking water (0.5 µg/L) was exceeded in 95.6% of surface water samples. Concentrations were highest during the summer-autumn rice crop season (main source of the fungicide isoprothiolane). Pesticides have already entered groundwater samples at some locations with isoprothiolane found in three samples.

In Can Tho province, a study (Toan et al. 2013) found ten different pesticides in 223 surface water samples. The fungicide isoprothiolane was detected most frequently (95% of the samples, median 0.16 µg/L). The insecticide fenobucarb occurred in more than 80% of the samples, but concentrations were low (median 0.04 µg/L). In 119 surface water samples from Dong Thap, isoprothiolane was the most frequently detected fungicide (100% of samples, max. concentration 11.24 µg/L, and median 2.72 µg/L). Fenobucarb was the most frequently detected insecticide (90.8% of the samples, median 0.11 µg/L), followed by buprofezin (58.7% of samples, median 0.19 µg/L). After heavy rains, insecticide cypermethrin concentrations occasionally reached up to

3.55 µg/L due to particle transport. Isoprothiolane was even detected in all 54 drinking water samples (median 0.17 µg/L), followed by fenobucarb (67% of samples, median 0.04 µg/L) and pretilachlor (48% of samples, median 0.01 µg/L).

VIWAT Engineering (Weihnacht 2021) analysed 117 surface water samples in December 2018 and December 2019 in Ca Mau province for carbamazepine, isohexol, isopamidol, isopromide, sulfamethoxazole, trimethoprim, azoxystrobin, propiconazole, trifloxystrobin, cyclamate, sucralose, but found no measurable concentrations. This might be due to the time of sampling with local farmers having not applied pesticides shortly before the sampling, the application of different pesticides, or to adsorption onto plastic bottles during transport from Vietnam to Germany (no glass bottles possible).

In the same study (Weihnacht 2021), the **DOC** was mostly below 10 mg/L, but could increase to 10–20 mg/L behind closed sluice gates. Half of the samples had cell counts for the total **coliforms** of about 10⁴ to 10⁵ CFU/100mL, for **E. coli** values between 10² and 10³ and for **enterococci** values of 10¹, meaning over 75% of all samples exceeded the requirements for surface water at all levels. TSS were also high with a mean of 120 mg/L. Except for Fe and Mn, dissolved trace metal concentrations were below threshold values for all samples.

HAWASUCO itself is taking samples of raw surface water from their water intakes for the surface water treatment plants on a quarterly to semi-annual basis and analysing it for 14 parameters (Table 1).

Table 1: Minimum, median and maximum values of analysed parameters in raw surface water intakes from six different water treatment plants (Vi Thanh, Nga Bay, Tan Phuoc Huong, Cay Duong, Tan Binh, Nga Sau) and at three different times each (end of April, end of June, early August of 2022), and Mot Ngan WTP end of June 2022. These 19 samples were compared to national technical regulations on surface water quality (QCVN 08-MT-2015/BTNMT, column A1). With light red background: values above the national guidelines.

Parameter	Units	Methods	QCVN, A1	Min.	Median	Max.
pH		TCVN 6492:2011	6.0-8.5	6.25	6.89	7.90
TSS	mg/L	TCVN 6625:2000	20	42.0	60.0	80.0
BOD ₅	mg/L	TCVN 6001- I :2008 or SMEWW5210D:2017	4	6.0	10.0	16.0
COD	mg/L	SMEWW 5220.C:2012	10	12.0	18.0	26.0
NO ₃ -N	mg/L	TCVN 6180:1996	2	0.34	0.44	0.55
NO ₂ -N	mg/L	TCVN 6178:1996	0.05	0.02	0.04	0.08
NH ₄ -N	mg/L	TCVN 6179-1:1996	0.3	0.08	0.22	0.34
PO ₄ -P	mg/L	TCVN 6202:2008	0.1	0.12	0.16	0.24
Fe	mg/L	TCVN 6177:1996	0.5	0.40	0.80	1.20
As	mg/L	US.EPA Method 200.7 or SMEWW 3113.B:2017	0.01	<0.001	<0.001	<0.002
Cl	mg/L	TCVN 6194:1996	250	20	30	40
F	mg/L	SMEWW 4500.F-.B&D:2017	1	<0.1	<0.1	0.40
Coliforms	MPN/100mL	TCVN 6187-2:1996	2500	480	7500	24000
<i>E. coli</i>	MPN/100mL	TCVN 6187-2:1996	20	nd	15	4800

Analytical results from HAWASUCO agree well with the results from the DONRE (Thuan 2022). Concentrations are exceeding the surface water guidelines (QCVN 08-MT-2015/BTNMT, column A1 (MONRE 2015b)) for TSS, BOD₅, COD, and PO₄ in all cases and for NO₂, Fe, Coliform, and *E.coli* in 42%, 68%, 79% and 42% of samples, respectively. NO₃, As, Cl, and F were not critical and below their thresholds in all cases. However, no pesticides or other organic contaminants have been analysed.

3.3. Treated water quality from HAWASUCO

HAWASUCO is treating the raw surface water by adding PAC (poly aluminium chloride) as flocculant. Then particles are flocculating and settling in a settling tank (Figure 3). Afterwards water is flowing through a sand filtration tank containing activated carbon. Eventually the water is disinfected with chlorine.



Figure 3: Settling tank (left) and filtration tank (right) at Vi Thanh WTP.

Treated water is tested by HAWASUCO

- every day for residual chlorine, pH, turbidity (and Cl in dry season)
- every month for smell, colour, As, *E. coli*, coliforms
- every 6 months for NH₄, NO₂, NO₃, Al, Cu, Fe, Mn, Ni, Zn, SO₄, TDS, salinity, hardness, and permanganate index.

HAWASUCO has provided test results for Jan 2023 from 11 different treatment plants with three samples, each for the parameters: turbidity, pH, residual chlorine, smell, colour, *E.coli*, coliforms and some for arsenic (Table 2). Results show that all measured parameters are below the thresholds for domestic water except in four cases, where residual chlorine was slightly above the threshold.

Table 2: Minimum, median and maximum values of analysed parameters in treated surface water from eleven different water treatment plants with three samples each for January 2023. These 33 samples were compared to national technical regulations on domestic water quality (QCVN 01-1-2018/BYT). With light red background: values above the national guidelines.

	Units	Methods	QCVN DW	Min.	Median	Max.
turbidity	NTU	TCVN 6184:2008	<2	0.14	0.41	1.44
pH		TCVN 6492:2011	6.0 - 8.5	6.86	7.11	7.64
residual chlorine	mg/L		0,91	0.20	0.35	1.00
smell, taste		HAWASUCO-HD-09	nd	nd	nd	nd
colour	TCU	SMEWW 2120C:2017	<15	3.94	9.17	9.91
<i>E. coli</i>	CFU/100mL	TCVN 6187-1:2019	<1	0	0	0
Coliforms	CFU/100mL	TCVN 6187-1:2019	<3	0	0	0
As	mg/l	TCVN 6626:2000	<0.01	<0.001	<0.001	0.005

nd: not detected

The technical guideline for domestic water QCVN 01-1-2018/BYT (MoH 2018) also prescribes threshold values for a range of other parameters: TDS, Cl, SO₄, F, Na, Al, B, Ba, Cd, Cr, Cu, Fe, Hg, Mn, Ni, Pb, Sb, Se, Zn, NH₄, NO₂, NO₃, permanganate index, cyanide, *Staphylococcus aureus*, *Ps. Aeruginosa*, 61 organic contaminants (8 chlorinated alkanes, 6 aromatic hydrocarbons, 3 chlorinated benzenes, 3 complex organic compounds, 27 pesticides, 14 disinfectant by-products), alpha and beta radioactivity.

Further water quality data for the selected site will be gathered and presented in report no 3.

SUMMARY: From the available data, the main concern with the water quality of the treated surface water from HAWASUCO would be residual chlorine (creating potential disinfectant by-products in the aquifer). Hence, it is recommended to divert treated water for injection before the chlorination step.

Persistent and mobile trace organic contaminants (for which no data are available) could be injected and might not be attenuated in the aquifer. This does not pose a risk for deterioration of water quality of the recovered water compared to the initial treated surface water, as it is unlikely that the concentration of organic contaminants would increase through the storage in the aquifer. The only source of organic contaminants is water being infiltrated and thus even if little to no degradation occurs in the aquifer, concentrations of organic contaminants are not expected to increase. However, organic contaminants might adsorb and accumulate in the aquifer and should hence be below the thresholds set in the Vietnamese guidelines for domestic water.

3.4. Ambient groundwater quality

The **DONRE Hau Giang** has been monitoring groundwater quality at eight different clusters from 2004 to 2021 for aquifers qh, qp₃ and qp₂₋₃. In 2022, four new monitoring clusters were installed (QT01, QT03, QT06, QT07new) and one of the previous cluster (QT13) remained, while the rest of the previous clusters were decommissioned (Figure 4).

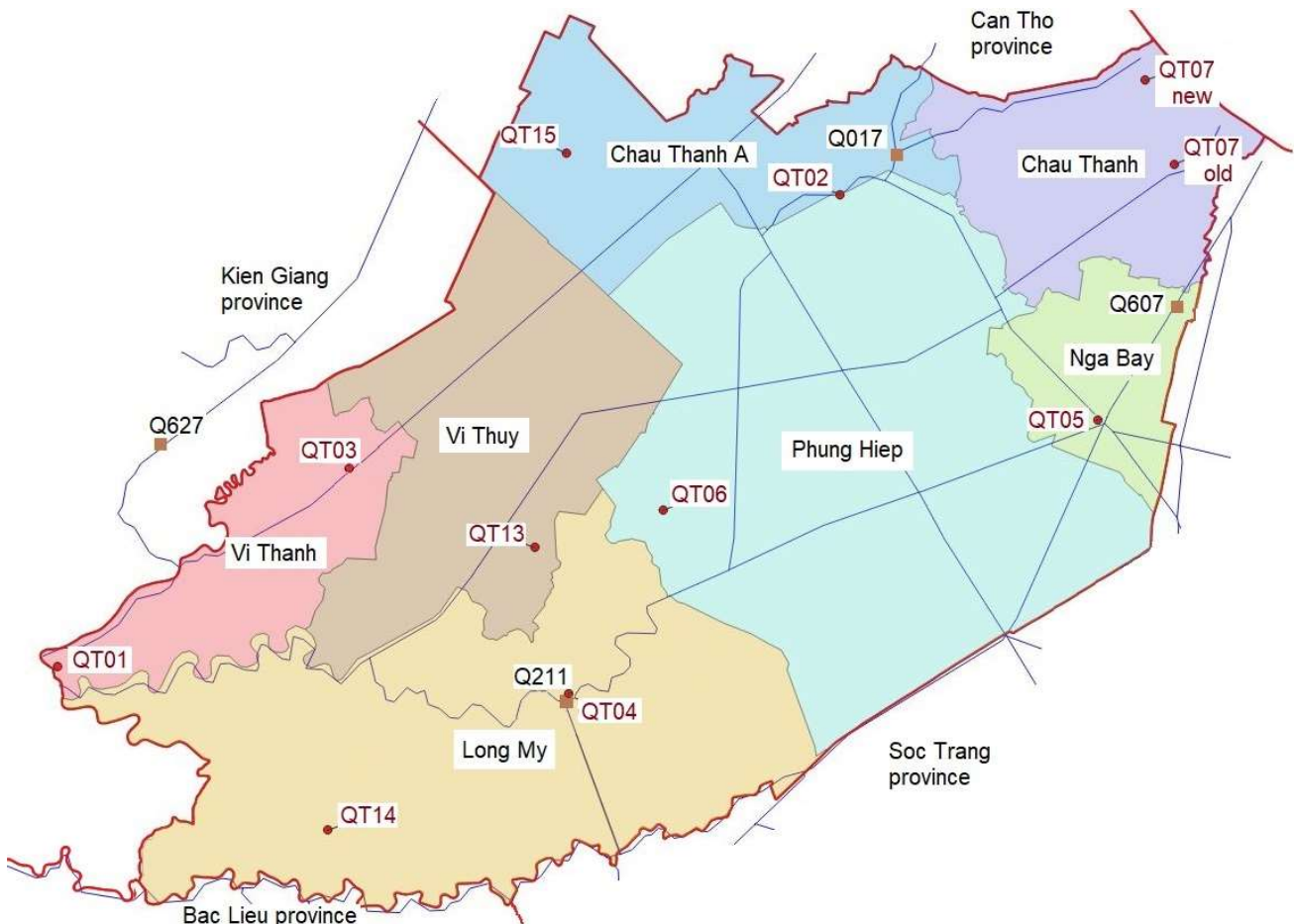


Figure 4: Groundwater monitoring clusters in Hau Giang from DONRE (QT) and NAWAPI (Q)

Groundwater quality in May 2021 closest to the proposed sites (Table 3) shows that the groundwater was fresh in all cases, but that Cl was elevated at three locations (*NOTE: it is not plausible that TDS < Cl for QT5A and QT14B; the quality of the data is questionable*). It also seems that salinity varies greatly over time as shown by the QT5 monitoring wells (Figure 5). For the shallow aquifer (qh), changes may be explained by different recharge amounts during wet and dry seasons as well as during dry and wet years. For qp₃ and qp₂₋₃, these variations might be due to groundwater flow resulting from local unlicensed abstractions. If these data were representative, it could mean that water quality at the selected MAR site might also change considerable within a short period of time.

Ammonium was elevated in all locations most of the time, probably due to the decomposition of natural organic matter contained in the aquifer. The permanganate index was highly variable from one sampling round to the next (e.g. KMnO₄ was 'not detected' for QT5B in Oct 2021 compared to 20.8 mg/L in May 2021) but also elevated in some locations during some years. Iron

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concentrations also vary significantly. While they are commonly below the threshold of 5 mg/L and often even below 1 mg/L, they could be > 20 mg/L at times. This could be a considerable problem for iron precipitation during the ASR pilot. Before this conclusion is drawn, it should be made evident that the observed variation of redox-sensitive parameters, such as permanganate index and iron, are indeed reliable, and are not the result of sampling and/or storage errors.

Table 3: Groundwater quality monitoring data for May 2021 from selected stations provided by DONRE Hau Giang compared to technical guidelines for groundwater quality QCVN 09-MT:2015/BTNMT. With light red background: values above the national guidelines.

Well ID		QCVN-GW	QT2A	QT4A	QT14A	QT5A	QT2B	QT4B	QT14B	QT5B
Aquifer			qp ₂₋₃				qp ₃			
Location			Thanh Hoa, Phung Hiep	Binh Thanh, Long My	Luong Tam, Long My	Nga Bay	Thanh Hoa, Phung Hiep	Binh Thanh, Long My	Luong Tam, Long My	Nga Bay
Well depth			142	144	119	111	48	76	65	76
pH		5.5-8.5	6.94	7.04	6.68	7.25	7.09	6.93	6.75	7.39
TDS	mg/L	1500	691	779	503	560	683	887	611	644
Cl	mg/L	250	114	35	44	780	121	57	816	277
SO ₄	mg/L	400	42	108	83	19	37	112	125	50
Hardness	mg/L	500	210	151	188	282	134	242	161	108
N-NH ₄	mg/L	1	1.68	2.48	1.38	2.63	1.89	2.76	1.5	2.63
N-NO ₂	mg/L	1	KPH	0.018	KPH	0.021	KPH	0.013	KPH	0.016
Fe	mg/L	5	2.24	4.91	0.35	12.19	1.71	0.28*	0.82*	0.53*
As	mg/L	0.05	nd	nd	nd	nd	nd	nd	nd	nd
KMnO ₄	mg/L	4	6.41	8.72	nd	19.7	1.25	11.0	nd	20.8

*values for May 2020 (May 2021 data for Fe had the same concentrations as Cl), nd = not detected.

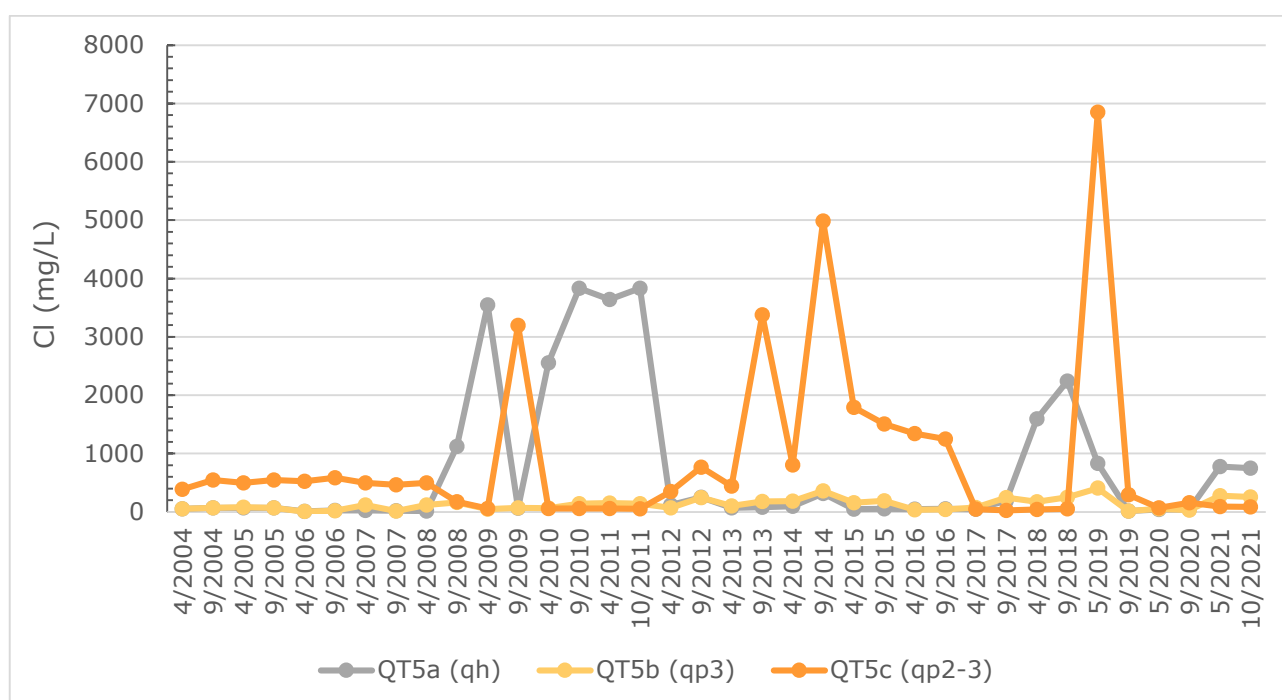


Figure 5: Chloride data from QT5 observation well cluster for shallow aquifers showing large variations over time by up to two orders of magnitude.

NAWAPI has a network of national groundwater monitoring stations with three clusters in Hau Giang: Q017 (2 wells aquifer qp_1 and n_1^3), Q211 (5 wells, aquifer qh , qp_{2-3} to n_1^3), and a new cluster Q607 (Nga Bay, 6 wells, aquifer qp_3 to n_1^3 , installed in 2019) (Figure 4). For qp_1 aquifer, salinity varies significantly across the province with fresh water (<1.5 g/L) in some areas and saline (>10 g/L TDS) in others. Aquifer qp_{2-3} is fresh in both monitoring clusters and brackish across the border to Kien Giang (Q627), but might also have some saline areas (Figure 6). However, the extent of saline groundwater is not very well known (compare the difference between Figure 6 and Figure 7) as monitoring wells are far apart. Local downward migration or upconing of saline groundwater is another possibility, as the upper aquifers qh and qp_3 or the deeper aquifer n_2^2 have saline areas (see Table 4). Migration of groundwater is possible through natural hydraulic windows or through the many improperly sealed wells and is driven by the increased hydraulic gradients from groundwater abstraction.

The saline groundwater samples are high in sodium and chloride, and often show elevated concentrations of iron and ammonium, while the fresh groundwater is a sodium-bicarbonate ($NaHCO_3$) water type (Table 4).

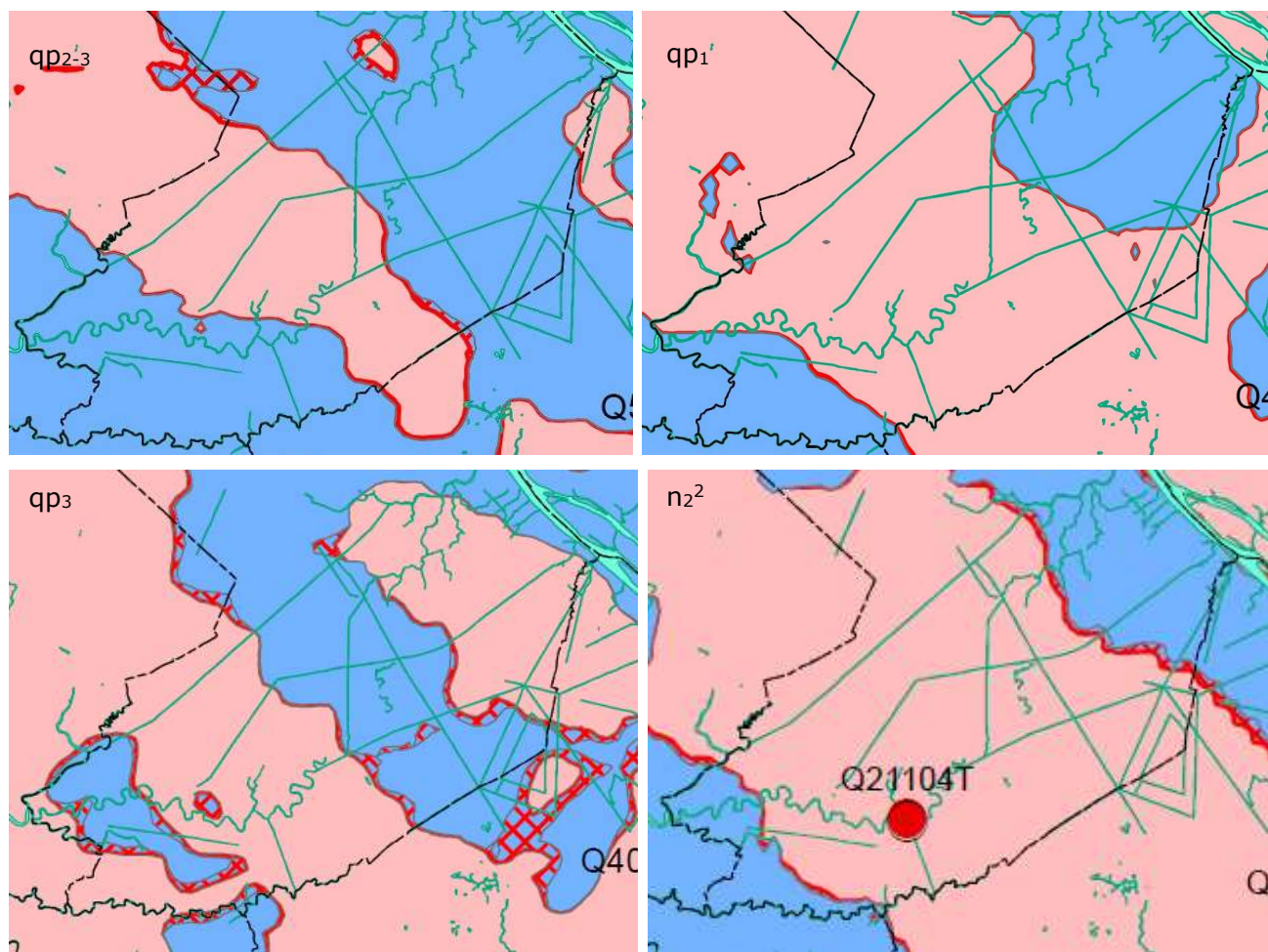


Figure 6: Salinity maps for qp_{2-3} (top left), qp_1 (top right), qp_3 (bottom left) and n_2^2 (bottom right) in Hau Giang with fresh areas (<1.5 g/L TDS) in blue and saline areas (>1.5 g/L TDS) in reddish. The modelled increase in saline areas in 2024 is given with red-hatched areas (NAWAPI Annual Report, 2019).

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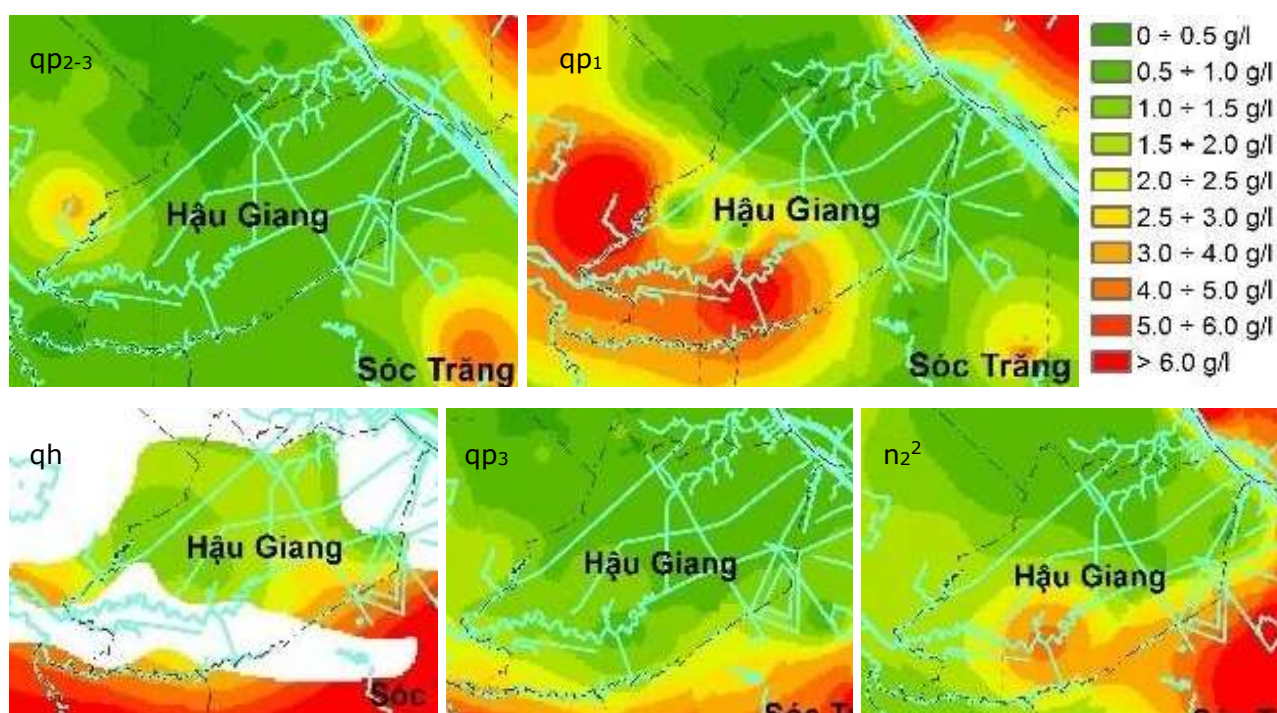


Figure 7: Salinity maps in mg/L TDS for qp₂₋₃ (top left) and qp₁ (right), qh (bottom left), qp₃ (bottom middle) and n₂² (bottom right) in Hau Giang (Quyên 2023).

Table 4: Groundwater quality monitoring data for Quarter 1-2021 from NAWAPI's national groundwater monitoring wells (received from NAWAPI in 2021). **Bold: concentrations exceeding the Vietnamese groundwater guideline.**

cluster	Q017	Q211				Q607			Q627		
aquifer	qp ₁	qh	qp ₂₋₃	qp ₁	n ₂ ²	qp ₃	qp ₂₋₃	qp ₁	qp ₂₋₃	qp ₁	n ₂ ²
pH	5.61	8.74	7.15	4.89	4.70	6.99	7.01	7.03	6.65	6.34	6.91
TDS	101	2812	672	13055	13489	1840	602	622	3083	10970	1768
Na	5.0	866.7	110.0	2857.1	4000.0	325.0	110.0	100.0	676.9	2.755.6	523.1
K	6.6	42.5	10.0	95.6	64.4	9.3	11.1	10.2	13.3	66.7	27.5
Ca	0.4	58.1	46.1	681.4	350.1	121.2	28.1	31.3	166.3	370.7	36.1
Mg	3.0	76.6	42.0	829.9	516.8	133.8	25.5	32.3	166.6	629.3	49.9
Cl	9.9	1199.6	81.2	7608.3	8370.8	888.0	18.4	19.9	1518.3	6167.9	843.7
SO ₄	1.0	5.4	119.0	814.6	26.7	72.3	161.7	163.4	257.2	811.8	44.5
HCO ₃	18.3	866.5	390.5	12.2	347.8	292.9	311.2	323.4	231.9	85.4	280.7
Fe ³⁺	0.36	0.94	0.59	23.18	7.45	4.00	1.77	0.92	5.41	21.52	1.26
Fe ²⁺	0.00	0.06	0.05	0.08	0.14	0.00	0.93	0.07	0.06	17.03	0.00
Mn	0.00	0.094	0.05			0.284	0.07	0.02	0.23	0.70	0.04
NH ₄	0.00	3.10	1.63	34.75	48.24	0.21	1.71	1.67	0.23	2.02	0.84
NO ₃	0.44	4.11	0.00	0.00	0.00	0.00	0.89	0.00	2.21	0.89	0.00
NO ₂	0.03	3.10	0.00	0.00	0.00	1.33	0.01	0.03	0.05	0.01	0.04
PO ₄	2.60	12.01	4.01			11.58	3.77	7.17	11.71	5.39	6.16
SiO ₂	50.5	35.09	36.7	40.4	21.3	63.03	62.5	59.7	79.9	43.1	42.6
Cu	0.019	0.019	0.019			0.019	0.019	0.019	0.019	0.019	0.019
Zn	0.019	0.928	0.019			0.019	0.019	0.019	0.016	0.019	0.019
As	0.002	0.012	0.004			0.003	0.005	0.006	0.002	0.002	0.013

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HAWASUCO also measured the ambient groundwater quality and provided some data for 2022 (Table 5). Here, none of the measured samples exceeded any thresholds, except one well in Dong Phu which had elevated concentrations of coliforms. Iron concentrations and permanganate index were much lower overall compared to the groundwater monitored by DONRE Hau Giang.

Table 5: Groundwater quality monitoring data for 2022 from wells of HAWASUCO compared to technical guidelines for groundwater quality QCVN 09-MT:2015/BTNMT. With light red background: values above the national guidelines.

Location	Unit	QCVN	Dong Phu			Luong Tam	Long Thanh	Cai Tac
Date			Apr 22	Jun 22	Aug 22	Jun 22	Jun 22	Jun 22
pH		5.5-8.5	6.56	7.24	6.57	7.42	7.20	7.36
TDS	mg/L	1500	926	1104	780			
Cl	mg/L	250	24	20	50	14	120	190
SO ₄	mg/L	400	236	290	340		195	
hardness	mg/L	500	86	80	90	54	170	240
F	mg/L	1	0.57	<0.1	<0.1	<0.1		<0.1
NH ₄ -N	mg/L	1	0.65	0.85	0.30	0.92		0.70
NO ₂ -N	mg/L	1	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005
NO ₃ -N	mg/L	15	<0.05	<0.05	0.07	<0.05	<0.05	0.06
Fe	mg/L	5	0.5	0.4	0.7	0.5	1.4	1.1
Mn	mg/L	0.5					0.17	
Zn	mg/L	3	<0.006	<0.006	<0.003	<0.006		<0.006
As	µg/L	50				<0.1	<0.1	<0.1
Hg	µg/L	1	<0.2	<0.2	<0.2	<0.2		<0.2
Pb	µg/L	10	<3	<0.6	<0.6	<0.6		<0.6
KMnO ₄	mg/L	4	<0.6	<0.5	<0.5	<0.5	<0.5	<0.5
Coliforms	MPN/100mL	3	75	<3	930	<3		<3
<i>E. coli</i>	MPN/100mL	nd	nd	nd	nd	nd	nd	nd

4. Water quality assessment prior to pilot ASR site development

4.1. Defining minimum criteria

There are no internationally defined minimum criteria as each scheme is different, but there are some overviews of existing MAR regulations (Demeau 2012, Fernández Escalante et al. 2022, Fernandez Escalante et al. 2020), with the USA and Australia having the most elaborate regulatory framework.

Within the **European Union**, all groundwater bodies should achieve a “good” quantitative and qualitative status, but the Groundwater Directive 2006/118/EC (Appendix I) (European Parliament and European Council 2006) only gives threshold values for nitrate (<50 mg/L) and pesticides incl. their relevant metabolites, degradation and reaction products (<0.5 µg/L total and <0.1 µg/L for individual pesticides). A list of main pollutants that should be considered during artificial recharge are given in Water Framework Directive (WFD) 2000/60/EC in Appendix VIII (European Parliament and European Council 2000) (e.g. persistent organic pollutants, metals, suspended matter etc.), but without any thresholds.

As a measure to prevent the input of pollutants to groundwater, **Germany** has prescribed threshold values for some parameters in the Groundwater Ordinance (GrwV) (Bundesministeriums der Justiz 2010) (Table 6). If no substance-specific information is available in the GrwV, the insignificance thresholds (“Geringfügigkeitsschwellenwerte” (LAWA 2016)) should be used, taking into account the principles of applicability and any geogenic background values for the groundwater body of concern.

Table 6: Threshold values for groundwater protection in the German Groundwater Ordinance (“Grundwasserverordnung” GrwV), Annex 2.

Parameter	unit	GrwV (Annex 2)	Parameter	unit	GrwV (Annex 2)
Nitrate NO ₃	mg/L	50	NH ₄	mg/L	0.5
total pesticides ¹	µg/L	0.5	NO ₂	mg/L	0.5
individual pesticides	µg/L	0.1	PO ₄	mg/L	0.5
As	µg/L	10	Cl	mg/L	250
Cd	µg/L	0.5	SO ₄	mg/L	250
Pb	µg/L	10	Total Tri- +	µg/L	10
Hg	µg/L	0.2	Tetrachloroethene		

The **French** National Agency for Food, Environmental and Occupational Health and Safety (ANSES 2016) recommends that the composition of the recharge water must be compatible with the constituent materials of the aquifer. This assessment shall be based on measurements of

¹ Based on European Commission Ordinance 1107/2009 including organic insecticides, organic herbicides, organic fungicides, organic nematicides, organic acaricides, organic algaecides acaricides, organic algaecides, organic rodenticides, antifoulings, slime control agents and related products (including growth regulators) and respective metabolites. For the pesticides aldrin, dieldrin, heptachlor and heptachlor epoxide heptachlor epoxide, the deviating limit value of 0.03 µg/L applies.

conductivity, pH, redox potential, calco-carbonic equilibrium and ion concentrations, in addition to the aquifer geochemical background. In general, water quality for recharge should be at least equal than the quality of the native groundwater and should respect the maximum values of TSS <10 mg/L, turbidity <5 NTU and TOC <10 mg/L.

The **Dutch** MAR guidelines are given in the Infiltration Decree (Infiltratiebesluit) (Ministry of Infrastructure and Waterworks 2009), which has merged into the Omgevingswet (Environmental Code) on 1 January 2024. Local authorities use these guidelines to formulate permits for MAR projects, applied to the local situation of available infiltration water and native groundwater.

The **Australian** guidelines (Natural Resource Management Ministerial Council et al. 2009) ensure a low risk of pollution by requiring the source water to meet: (a) quality requirement for groundwater environmental values, (b) quality requirements for the intended end-use of the recovered water and (c) quality requirements with respect to clogging (TSS <10 mg/L, TOC <10 mg/L and TN <10 mg/L). The Australian guidelines place a strong emphasis on monitoring to reduce any risk and assess system performance.

The **USA** (USEPA 1974, 2012) have the earliest and strictest regulations requiring recharge water to meet drinking water standards for all native groundwater with TDS <10 000 mg/L. Contaminants of concerns in MAR projects with injection wells (ASR) are nutrients, metals, pesticides, endocrine disruptor compounds, pharmaceuticals and personal care products, and microbes (USEPA 2019). Detailed technical guidelines for all steps of a MAR project are available (ASCE 2020).

SUMMARY

The success of Aquifer Storage and Recovery (ASR) schemes relies on defining appropriate design and operational parameters in order to maintain high rates of recharge over the long term (Page et al. 2014). The main risks are (a) operational risk (clogging), (b) risk of groundwater quality deterioration (pollution) and (c) risk of injected water quality deterioration due to hydrogeochemical reactions of the injected water with the aquifer matrix and ambient groundwater.

Main aspects to be considered are:

1. water source
2. hydrogeological conditions
3. MAR technology
4. monitoring (legal requirements and parameters)
5. final use of recharged water
6. risk and impact assessment.

To prevent **clogging** and the failure of the system in quantitative terms, the following preferred values are suggested by Dutch experts (Table 7).

Apart from the prevention of clogging, recharge water should also not pose a **risk to the ambient groundwater** (Zheng et al. 2023). Recharge water should hence meet at least the technical guideline for groundwater quality QCVN 09-MT:2015/BTNMT (MONRE 2015a). It provides thresholds for 32 parameters (pH, Permanganate, TDS, hardness, NH₄, NO₂, NO₃, Cl, F, SO₄, CN, As, Cd, Pb, Cr, Cu, Zn, Ni, Mn, Hg, Fe, Se, aldrin, benzene hexachloride (BHC), dieldrin, DDT, heptachlor & heptachlorepoide, total phenol, alpha and beta radioactivity, coliforms and *E.coli*).

In case the ambient groundwater is of higher quality than the regulation, it is suggested that recharge water should have at least the quality of the ambient groundwater. In the dry season, if salinity intrusion into the surface water reaches HAWASUCO water intake canals, this would mainly be a problem for the major ions and salinity which are not removed in the treatment process.

Table 7: Minimum criteria for recharge water suggested to prevent clogging (Beernink and van der Schans 2023).

Clogging process	Parameter	Preferred values
Chemical clogging (mineral precipitation)	Al (total dissolved)	<0.1 mg/L
	SI _{Ca}	<1
	Fe (III) / colloidal iron	<0.01 mg/L
Mechanical clogging	MFI (membrane filtration/fouling index)	<2 s/L ²
	TSS	<0.1 mg/L
	SAR (sodium adsorption ratio)	<3 / <6
Biological clogging	AOC (assimilable organic carbon/acetate)	<10 µg/L acetate-C
	DOC (dissolved organic carbon)	<2 mg/L
	ATP (adenosine triphosphate)	<10 ng/L
Physical clogging	Total gas pressure in groundwater	<atmospheric pressure

SUMMARY:

As we are planning to use treated water with domestic water quality, the recharge water quality will be guided by the technical regulations for domestic water, which are stricter than the regulations for groundwater quality. Hence, there should be no risk of contamination of the ambient groundwater, but Vietnamese guidelines for groundwater protection (GW-QCVN 09-MT:2015) must of course be complied with.

However, for the prevention of clogging, even higher standards for specific parameters (Table 7) might be required. The concentration of dissolved oxygen is likely to be the main concern for triggering geochemical reactions, as the water is oxygenated after the water treatment (estimated 7 mg/L).

4.2. Recommendations for water quality assessment

It is suggested to conduct a sampling campaign of **treated surface water** and **ambient groundwater** at the injection well from the selected pilot site and analyse it for

- field parameter: EC, pH, redox potential, temp, DO, turbidity, alkalinity
- major and cations and anions (Na, K, Mg, Ca, Cl, SO₄, HCO₃, F, Br, Al, As, Ba, Cd, Cr, Cu, Fe, Mn, Ni, Pb, Se, Sr, Zn),
- nutrients (NH₄, NO₂, NO₃, PO₄),
- TOC, DOC,
- coliforms, E.coli, enterococci and
- organic contaminants like organochlorine pesticide residues, herbicide residues, carbamate pesticides residues, triazole pesticides residues, atrazine residues (minimum would be to cover aldrin, benzene hexachloride (BHC), dieldrin, DDT, heptachlor & heptachlorepoxyde, total phenol as in technical guidelines for groundwater quality)

- as well as disinfection by-products in the chlorinated treated water

Ions, nutrients and DOC should be determined in water samples after filtration (0,45 µm).

No laboratory in Vietnam could be found for the analysis of PFAS and other emerging pollutants and these substances will hence not be analysed.

Sampling should take place on two different occasions (wet and dry seasons) with samples for treated surface water before and after chlorination and from each groundwater well (e.g. 2 wells at the site), giving a total of **eight samples**.

Based on the results, if minimum requirements are not fulfilled, further treatment steps might have to be added before the water could be used for injection.

4.3. Sediment sampling

To predict and model geochemical reactions and clogging likely to occur in the underground during the ASR cycles, it is beneficial to have detailed knowledge about the sediments present in the subsurface (Antoniou et al. 2012), (McCurry and Pyne 2022) (Lipperera et al. 2023). Before drilling the injection well, an exploratory drilling would be undertaken, which would be converted into a monitoring well. It is beneficial to take sediment samples during this drilling.

It is recommended to take samples from the overlying aquitard, the tapped aquifer and the underlying aquitard (if reached). A minimum of 3 samples per layer and a maximum of 6 samples (every 4-5 m depth) based on the thickness of the layers as known from previous logs and discovered during the drilling should be taken.

Disturbed representative sediment sample of about 500 g should be collected with a sampling tool and placed into well-labelled sealed plastic bags. Collected samples would be dried at 40–60°C and send to the BGR headquarters for analysis of:

- grain size analysis,
- XRD (X-ray diffraction; the XRD method is a mineralogical method for the phase analysis),
- XRF (X-ray fluorescence; the XRF method is a non-destructive method to determine the elemental composition of a sediment sample),
- (total and organic) carbon and (total) sulphur content, and
- cation-exchange capacity (CEC).

4.4. Data acquisition

Before implementation, the project should collect as much existing data as possible to get a better understanding of the groundwater and aquifer system:

- raw groundwater and surface water data available at HAWASUCO and DONRE,
- static and dynamic groundwater levels available at HAWASUCO and DONRE,
- well logs from nearby licensed wells available at the DONRE,
- sediment data from previous studies in the Mekong Delta,
- published literature data on other MAR project within a similar setting.

5. Monitoring during pilot implementation

ASR pilot monitoring and assessment: Monitoring and assessing various parameters, including but not limited to, such as pressure, groundwater table, well efficiency, infiltration rate, extraction rate, water quality etc. These parameters need to be monitored throughout the campaign to inform the final evaluation report.

There are two aspects during the implementation of an ASR system that could hinder its success: water quantity issues and water quality issues (ASCE 2020), (McCurry and Pyne 2022).

5.1. Water quantity monitoring

A suitable aquifer for ASR is defined by the amount of water that can be recharged to and recovered from the aquifer. The **storage capacity** depends on the extent of the aquifer, its permeability, transmissivity and its confinement. For confined aquifers, it is important not to over-pressurize the aquifer and cause rupture of the confining layer.

The **recovery efficiency** is defined as the percentage of water volume injected compared to the water volume that can be recovered within the target water quality criteria. As in our case we are using fresh high quality water in a fresh good quality aquifer, it might not be possible (or required) to distinguish between the recovery of recharged water and ambient groundwater.

The recovery efficiency can be reduced due to:

- high regional hydraulic gradients, which is in general not the case in the Mekong Delta, but local drawdowns around large production wells might influence local gradients,
- highly saline aquifers, which is not the case for our pilot,
- highly inhomogeneous aquifers (e.g. karst, fractured rock), which is not the case in the Mekong delta, or
- leaky aquifers resulting in loss to over- or underlying aquifers, which is likely in the Mekong Delta, but not necessarily a problem, as groundwater is extracted from many aquifers.

It also helps to use ASR schemes with different wells for recharge and recovery (which is planned for this pilot) or trickle flow during storage to compensate for potential density stratification.

The main concern with well performance is **clogging** (as described above). This reduction in porosity occurs mainly around the injection well and leads to a decrease in injection rate and an increase in hydraulic head. To remediate excessive clogging, periodical purging or backflushing of the well will be required.

Disposal of these back-flushed waters that are usually of low quality has to be accounted for in the planning stage (ASCE 2020).

The aim of water quantity monitoring is hence to assess:

1. infiltration rates
2. recovery rates
3. well efficiency (i.e. clogging)
4. costs and benefits (costs per m³)

The following parameters should hence be monitored:

- recharge volume (flow rate over time at injection well)
- recovery volume (flow rate over time at recovery well)
- groundwater table changes at recharge, recovery and monitoring wells
- pressure changes at injection well
- energy requirements for injection and recovery

All above parameters should be monitored with (semi-)automatic sensors to give high frequency readings. Hence, the following **sensors** are required:

- water level, temperature and EC at recharge well, recovery well and monitoring wells (number depends on the site)
- EC of recharged and recovered water; if feasible also in monitoring wells at screen depth
- flow meter at recharge well and recovery well
- pressure meter at recharge well
- energy consumption meter at recharge pump and recovery pump

Ideally, data from these sensors is logged and transmitted automatically. IoT-enabled devices are making it easier and cheaper than ever to capture real time monitoring data and deliver these data—in the form of graphical visualisations—to users in a matter of seconds. A low power wide area network (LPWAN) application, known as LoRaWAN, should enable the transfer of encrypted data over the whole premises of HAWASUCO (need to be checked if any building get in the way of the signal), while maintaining low-power consumption. LoRaWAN requires a local receiving gateway (base station) to transfer the incoming packets from the **IoT sensor nodes** to a network server to be decoded, after which a formatted (JSON) object (data) can be forwarded to a back-end database for storage and/or a front-end server for visualisation

Otherwise, logged data have to be retrieved manually at regular intervals.

In addition, it would be advisable to conduct **geodetic levelling** for all wells to be able to fully compare the measured groundwater levels with each other.

The detailed design for the injection well should consider how this monitoring equipment could be installed without compromising the ability to pressurize the water.

5.2. Water quality monitoring

The aim of the water **quality** monitoring is to assess the:

1. clogging potential (see above),
2. pollutants posing a risk for groundwater quality and domestic water quality (see above),
3. quality of the recovered water,
4. interactions taking place in the disturbed groundwater system (model geochemical reactions occurring from the infiltration of oxygenated surface water to anoxic groundwater and during passage in the underground).

Temporally, water quality changes are maximal during the initial ASR operating cycles. During consecutive cycles, the groundwater in the storage zone will be completely exchanged and geochemical reactions will adjust to the new equilibrium of the stored water, especially if not all injected water is recovered. However, in very heterogeneous aquifers diffusion processes might require several cycles to flush the residual ambient groundwater from lenses and layers of low hydraulic conductivity (Pavelic et al. 2006a).

Spatially, water quality changes are largest around the injection well, where the number of pore flushes, geochemical variations and the hydraulic gradient are highest. In the main storage zone, the number of pore flushes is reduced and water quality changes are less pronounced as the water has been pre-equilibrated around the well. Depending on the differences between injectant and ambient groundwater, the differences in water quality in the buffer zone furthest away from the well could nevertheless still be significant (Dillon et al. 2005), while a stable injectant water quality "bubble" will be established after a significant number of cycles.

For all schemes, biogeochemical interactions with the aquifer matrix like **ion exchange**, **redox reactions** and **dissolution/precipitation** will have an impact on the recovered water quality. Accordingly, attention should be paid to field parameters (pH, redox potential, temperature, EC), major ions as well as trace metals that could be released during dissolution. The frequency at which these parameters should be measured depends on the specific experimental setup. More specific information about this is given in Report 03.

The **organic matter** present in the aquifer may undergo oxidation and potentially release dissolved organic carbon, CO₂, H⁺, and nitrogen compounds.

When using potable water as injectant, **disinfection byproducts** are of concern and should be monitored. Hence, it is recommended to divert the treated water before chlorination for the ASR pilot.

When recovering water for potable use, trace organic pollutants in the recharge water like endocrine disruptors are of concern. However, these would already be a concern for drinking water supply without the ASR scheme. If they are mobile and are recovered, they may undergo a second treatment, as the recovered water would be treated again before distribution into the network. If they are absorbing to the aquifer matrix or degraded during the relatively long residence time, and not be recovered, recovered water quality would be improved by natural attenuation. In any case, it is not expected that the concentration of trace organic pollutants would be increased by the passage through the aquifer, as they are anthropogenic and not found in the aquifer sediments.

The following monitoring scheme is suggested (Table 8):

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Table 8: Suggested parameters for water quality monitoring during implementation.

Where	Why	Parameter
Recharge water quality	clogging potential	turbidity
		DO
		Fe, Al, Mn
		DOC
		gas pressure
	MFI	
	contamination	micropollutants, pesticides (if found during initial water quality assessment)
biogeochemical processes	pH, redox potential, temp, EC	
	Na, Ca, Mg, K, SiO ₂ , Cl, SO ₄ , HCO ₃	
Recovered water quality	biogeochemical processes	pH, redox potential, temp, EC
		Na, Ca, Mg, K, SiO ₂ , Cl, SO ₄ , HCO ₃
	metal mobilization	Al, As, Cd, Cr, Cu, Fe, Mn, Ni, Zn
	organic matter degradation	DOC
		NO ₃ , NO ₂ , NH ₄
disinfection-byproducts	total trihalomethanes (THMs) (if chlorinated water is used for injection)	

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Appendix I: Methods for sediment analysis

The **grain size distribution** is analysed using two methods: the small fraction (grain diameter $<63 \mu\text{m}$) of each sample is analysed with the Sedigraph method, whereas the large particles (grain diameter $> 63 \mu\text{m}$) is analysed with a Camsizer instrument. The Sedigraph unit semi-automatically determines the grain size distribution by measuring the absorbed x-rays on the particles during sedimentation. This method is based on Stokes' law describing the subsidence of particles in an aqueous solution. The Camsizer instrument provides an alternative for the traditional sieve analysis for high-resolution analysis of both the particle size and the form. The Camsizer operates by the principle of a dynamic digitally image analysis. The measuring range is for grains between $30 \mu\text{m}$ and 30mm . The results of the analyses are divided into the following fractions with their individual grain diameters: clay ($<2 \mu\text{m}$) – silt ($2 \mu\text{m} - 63 \mu\text{m}$) – fine sand ($63 \mu\text{m} - 200 \mu\text{m}$) – medium sand ($200 \mu\text{m} - 630 \mu\text{m}$) – coarse sand ($630 \mu\text{m} - 2 \text{mm}$) – gravel ($> 2 \text{mm}$). With the sum component of each grain size fraction, the typical grain size curve can be displayed.

For **XRD** analysis, the laboratory at the BGR uses a PANalytical instrument MPD Pro with double detectors for the XRD measurements. The measurements are performed with powdery samples. The calculation of the phase fractions is based on the detection of the present minerals with a specific software. This is, particularly for the swell able clay minerals, not always possible, because of structural disorders of the natural minerals.

For **XRF** analysis, the samples are analysed in the form of fused tablets. In a first step, the samples are homogenized by crushing and grinding. After weighting and estimating the volatile fraction (annealing at 1030°C), e.g. the loss of ignition, homogeneous fusing tablets are produced in an automatic melting furnace. In the last step, the quantitative XRF analysis is done with a wavelength dispersive sequence spectrometer.

The amounts of total **carbon** (C_{tot}), organic carbon (C_{org}) and total **sulphur** (S_{tot}) are analysed with the LECO CS 230 carbon sulphur analyser. Measurements are done in a high frequency heater after sample combustion in an oxygen stream at about 2000°C . Both the resulting sulphur dioxide and carbon dioxide are analysed by infrared detection, whereas carbon monoxide is catalytically oxidized before detection. Untreated samples were used for the determination of C_{tot} and S_{tot} , and decarbonized samples for the determination of C_{org} , respectively. For the removal of inorganic carbon, each sample was treated before incineration with a 10% hydrochloride acid (HCl) at 80°C . For the measurements, the maximal sample weight of 170 to 190 mg was weighed in.

The analysis of the **CEC** is performed in the laboratory of the BGR with the Cu(II) Triethylenetetramine (TETA) method. In a centrifuge tube, the milled sample material is mixed with 50 ml water, and a complex of both 0.01 mol CuSO_4 and 0.01 mol TETA. After centrifuging, the supernatant is analysed by a VIS spectroscopy and an inductively coupled plasma spectrometer (ICP), with regard to the content of Cu and cations.