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# D22.4 Evaluation of pre-potable water requirements for a safe injection in the Aquifer through ASR

## Results of Task 22.3: Increase the flexibility and resilience of Aquifer Storage and Recovery (ASR) in strategic groundwater reservoirs

December (2015)

Revised version, November (2017)

The research leading to these results has received funding from the European Union Seventh Framework Programme (FP7/2007-2013) under grant agreement no. 619039

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## D22.4 Evaluation of pre-potable water requirements for a safe injection in the Aquifer through ASR

RESULTS OF TASK 22.3: INCREASE THE FLEXIBILITY AND RESILIENCE OF AQUIFER STORAGE AND RECOVERY (ASR) IN STRATEGIC GROUNDWATER RESERVOIRS

**SUMMARY**


Results of Task 22.3 have been summarised in three separated reports, which are presented in this document for a comprehensive understanding. The reports present same structure, executive summary, background information and conclusions, so they can be read separately. Aquifer Storage and Recovery (ASR) is a site-specific technique which can be approached from several disciplines: hydrogeology, water quality regulations and recommendations, international experiences, clogging formation and evolution, potential pre-treatments, numerical modelling, etc. All of these issues have been studied and are presented in three main chapters.

All the findings have been obtained in the experimental tasks of the project (corresponding to RTD activities), such as literature review, historical data analysis, laboratory experiments and pilot tests to simulate ASR. Further work is being performed in the project at demonstration site, the Llobregat aquifer, and will be presented in the deliverable D35.1. Results of Task 22.3 are valuable background information to undertake demonstrative activities at field scale.

**Chapter A:**

**Description of the ASR system in the Llobregat Area and Water Quality Evaluation based on historical data**

The Aquifer Storage and Recovery System (ASR) in Llobregat is one of the oldest Managed Aquifer Recharge systems in Europe. Historically, drinking water has been recharged in the alluvial aquifer using injection wells. Despite the suitability of the aquifer and the good results achieved in terms of volume injected, the system has been operated fewer and fewer because the increase of potabilisation costs of the water to be injected. Therefore, in order to recover the sustainability and the economic feasibility of this ASR system, DESSIN project aims at demonstrating the injection of pre-potable water.

This chapter describes the Llobregat ASR system and presents the evaluation of the pre-potable water requirements to fulfil a safe injection in terms of aquifer quality and well operation. First results presented are the exhaustive literature review of recommendations and compilation of international experiences of ASR systems and their main operative parameters. Historical data of the sand filtered water produced has been plotted and analysed compared to quality standards and recommendations. The result of this evaluation has been useful to identify the strengths and weaknesses of sand filtered water being injected in the aquifer.

**Chapter B:**

**Evaluation of pre-treatments and pilot test results**

This chapter describes the experimental evaluation in real conditions that has been done in order to validate the sand filtered water as a pre-potable water to be injected in the aquifer and evaluate if it is needed any additional pre-treatment. As one of the sand filtered water drawbacks is the microbial load presence, as additional pre-treatments it were evaluated different disinfection methods.

Results will serve as robust conclusions of the consequences in the well of pre-potable water injection, and will give real conditions and recommendations for a correct future operation during DESSIN demonstration phase in a real well and also for a possible future implementation of complete ASR system.

**Chapter C:**

**Regional and local numerical modeling to simulate the flow and conservative transport in the Llobregat demo site**

This chapter summarises results of the application of numerical modelling to the Llobregat ASR system. The work corresponds to the first phase of the project, focused on the impact assessment of ASR in terms of groundwater volume infiltrated in the aquifer and the improvements and/or impacts in groundwater quality. The work has been divided in two parts: (i) MODFLOW-based numerical model to simulate the impact on injected water in the local piezometric network installed for the project (4 $km^2$) (ii) VISUAL TRANSIN-based numerical model to simulate the impact of ASR and ASTR at regional scale (129 $km^2$).

Results of this report correspond to the simulations carried out of Scenario 1 (Demonstration scale of the project) and Scenario 2 (application of ASR in the full system). Results of Scenario 1 conclude that the demonstration phase of the project will have a local impact in the aquifer, as the mixing ratio between injected water and native groundwater will be below 10% after 1.4 km of aquifer passage. Local model and regional model have been key information for the establishment of local control network (Pz1, Pz2 and Pz3) and the selection of external control points in the aquifer (P10, P13 and P03) to verify the impact in groundwater quality during the demonstration phase.

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## CHAPTER A: Description of the ASR system in the Llobregat Area and Water Quality Evaluation based on historical data

Cetaqua, April (2015)

Revised version, November (2017)

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# D22.4 Evaluation of pre-potable water requirements for a safe injection in the Aquifer through ASR

CHAPTER A: DESCRIPTION OF THE ASR SYSTEM IN THE LLOBREGAT AREA AND WATER QUALITY EVALUATION


**SUMMARY**

The Aquifer Storage and Recovery System (ASR) in Llobregat is one of the oldest Managed Aquifer Recharge (MAR) systems in Europe. Historically, drinking water has been recharged in the alluvial aquifer using injection wells. Despite the suitability of the aquifer and the good results achieved in terms of volume injected, the system has been operated fewer and fewer because the increase of potabilisation costs of the water to be injected. Therefore, in order to recover the sustainability and the economic feasibility of this ASR system, DESSIN project aims at demonstrating the injection of pre-potable water.

This report describes the Llobregat ASR system and presents the evaluation of the pre-potable water requirements to fulfil a safe injection in terms of aquifer quality and well operation. First results presented are the exhaustive literature review of recommendations and compilation of international experiences of ASR systems and their main operative parameters. Historical data of the sand filtered water produced has been plotted and analysed compared to quality standards and recommendations. The result of this evaluation has been useful to identify the strengths and weaknesses of sand filtered water being injected in the aquifer.


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# List of Acronyms and Abbreviations

| | |
|---|---|
| ACA | Catalan Water Agency |
| AOC | Assimilable Organic Carbon |
| ASPC | Agència Catalana de Salut Pública (Health Authority) |
| ASR | Aquifer Storage and Recovery |
| BDOC | Biological Dissolved Organic Carbon |
| BOM | Biological Organic Matter |
| CFU | Colony Formation Unit (unit used in microbiology) |
| EPA | Environmental Protection Agency (USA) |
| HAA | Halo Acetic Acid |
| DBP | Disinfection by-products |
| DWTP | Drinking Water Treatment Plant |
| LSI | Langelier Saturation Index |
| MC | Multicriteria Analysis |
| MCL | Maximum Contaminant Level |
| MFI | Membrane Fouling Index |
| NWQMS | Australia's National Water Quality Management Strategy |
| RF | Reliability Factor |
| RI | Ryznar Index |
| SFW | Sand Filtered Water |
| SJD | Sant Joan Despí |
| SS | Suspended Solids |
| TDS | Total Dissolved Solids |
| THM | Trihalomethanes |
| TOC | Total Organic Carbon |
| UIC | Underground Injection Control |
| UF | Ultra Filtration |
| UV | Ultra-Violet (disinfection method) |
| WFD | Water Framework Directive |



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# Executive summary

The Aquifer Storage and Recovery System (ASR) in Llobregat is one of the oldest Managed Aquifer Recharge (MAR) systems of its type in Europe. Historically, drinking water has been recharged in the alluvial aquifer using injection wells. Despite the suitability of the aquifer (alluvial aquifer with a high hydraulic transmissivity) and the good results achieved in terms of volume injected, the system has been operated fewer and fewer because the increase of the treatment costs of the potable water to be injected. Therefore in order to recover the sustainability and the economic feasibility of this ASR system, DESSIN project aim to demonstrate the injection of pre-potable water.

This report describes the Llobregat ASR system with a contextualization in the water management of Barcelona metropolitan area and its historical operation. International water quality standards for ASR have been reviewed regarding legal framework (in Europe, USA and Australia), contaminant injection and well operation issues. These standards have been compared with Llobregat pre-potable water quality. Moreover it is compared Llobregat aquifer characteristics with international examples of ASR.

All the evaluation has been done in order to know the strengths and weaknesses of the pre-potable water chosen in Llobregat site to forecast the operation of the demonstration phase that it is going to be implemented in following project tasks.

# 1 Introduction

Artificial groundwater recharge has been practised since the nineteenth century, at least in England, France, Germany, Scotland, Sweden and the USA. Nowadays it is used to a varying extent to produce drinking water as for example Belgium, Czech Republic, Denmark, Finland, Greece, The Netherlands, Poland, Spain and Switzerland. Due to the increasing need of drinking water and overexploitation of natural groundwater, the managed aquifer recharge has become an increasingly important means of solving water supply problems.

In spite of the long tradition of managed aquifer recharge (Wood and Bassett, 1975; Okubo and Matsumoto, 1979), many problems and uncertainties have not been totally solved. The deterioration of surface water quality has caused new problems. The water used for infiltration typically contains elevated concentrations of humic substances, industrial pollutants, microorganisms and inorganic solutes. Furthermore, the infiltration rate is much higher in managed aquifer recharge than under natural conditions. We therefore need to gather more information about purification processes and to improve MAR methods. The aim is to ensure the uniform quality of drinking water and the effective and sustainable capacity of water works, as well as to protect the quality of natural groundwater.

Aquifer Storage and Recovery wells (ASR) are a combination of recharge and pumping wells (Pyne, 1989; Bouwer *et al.*, 1990). They are used for recharge when surplus water is available, and for pumping when the water is needed. ASR wells typically are used for seasonal storage of drinking water in areas where water demands are much greater in summer than in winter, or vice versa. Drinking water treatment plants then are designed for mean annual capacity. ASR wells also can be used to avoid depletion of groundwater by recharging in the winter to compensate for excessive withdrawal in the summer.

However an increasing number of water managers are constructing ASR systems to ensure reliability of supply during emergencies such as floods, contamination incidents, pipeline breaks or to ensure supply during periods of maintenance. Because of the wide range in applications there is also a wide range in injection capacity for different ASR systems. ASR systems reported in literature range in scale from single well systems for domestic or horticultural irrigation to ASR well fields consisting of over 20 wells to meet water demand for urban areas or industrial use. Typical storage volumes for individual wells can hereby range from 0.04 $Mm^3$ for a small ASR plant to 2 $Mm^3$ for a large plant (Pyne, 2005). The largest ASR well fields in operation have design storage volumes in excess of 4 $Mm^3$, enabling a seasonal water supply of 30 to 280 ML/d (Pyne, 2005).

In the future, the ability to reduce pre-treatment of surface water to be stored in ASR aquifers may represent considerable cost savings for implementation of these systems. If natural conditions within the subsurface zone of discharge demonstrates an adequate attenuation of potentially harmful microbes (e.g., microorganisms are inactivated) and the systems are operated with proper monitoring and safeguards, the need for pre-treatment of storage water may be reduced.

# 2 Objectives

The objectives of the report are to summarize the findings in the task 22.4 of the DESSIN project regarding the evaluation of pre-potable water requirements for a safe injection in the aquifer. Specifically, the objectives of this report are:

- **Contextualize the ASR in Barcelona water supply system** to:
    - Describe why ASR system was constructed and why is important
    - Characterize the zone and the Llobregat Aquifer
    - Describe the DESSIN new scheme proposal

- **Analyse the literature and international experiences on ASR regarding:**
    - Legal framework in different world areas
    - Contaminant control of recharged water
    - Well operation parameters

- **Characterize the pre-potable water to be injected in Llobregat DESSIN site**
    - Describing clogging and contamination related parameters
    - Making multicriteria analysis with related software

- **Evaluate strengths and weaknesses** of pre-potable water in Llobregat ASR scheme

# 3 Overview of Barcelona water supply system

Barcelona is one of the most populated cities in the north bank of the Mediterranean. Barcelona's metropolitan area displays a physiographic pattern consisting of a narrow coastal plain, occasionally broadened by river deltas, and a series of mountain ranges roughly parallel to the coastline and separated by rolling plains. Climate is characterized by mild winters and hot summers and precipitation values oscillate around 500-700 mm a year but there is a pronounced inter-annual and intra-annual variation.

83% of the water input for potabilisation comes from surface sources, 40% (78 $Mm^3$) comes from the Llobregat River and 43% (83 $Mm^3$) from the Ter river. As it can be seen in Figure 2 these two rivers that mainly supply water to Barcelona metropolitan area have an average flow really smaller than big European rivers, implying that Barcelona system have bigger water stress and water scarcity risk than other European regions.

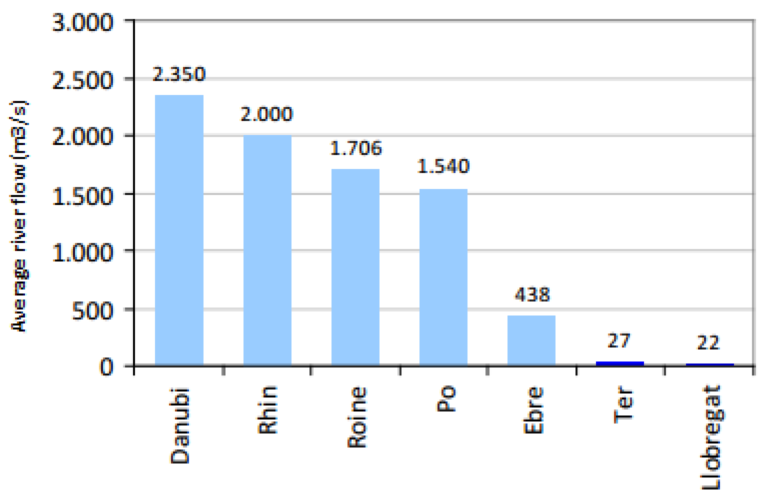



**Figure 1: Average flow of big European rivers and Ter and Llobregat rivers**

The water flow supplied by the Ter and Llobregat rivers are regulated by three and two reservoirs respectively and purified by one (Ter) and two (Llobregat) potabilisation plants (Figure 2). In addition, there is an input of 27 $Mm^3$ of groundwater, representing 14% of all the water resources supplied to the metropolitan area. Within the region, urban consumption shows a decline in the city itself and in neighbouring suburban areas due to population and industrial shifts, while it is growing rapidly in the periphery. Average daily household consumption in the metropolitan area is about 110 L/inhabitant (109.5 in 2011).

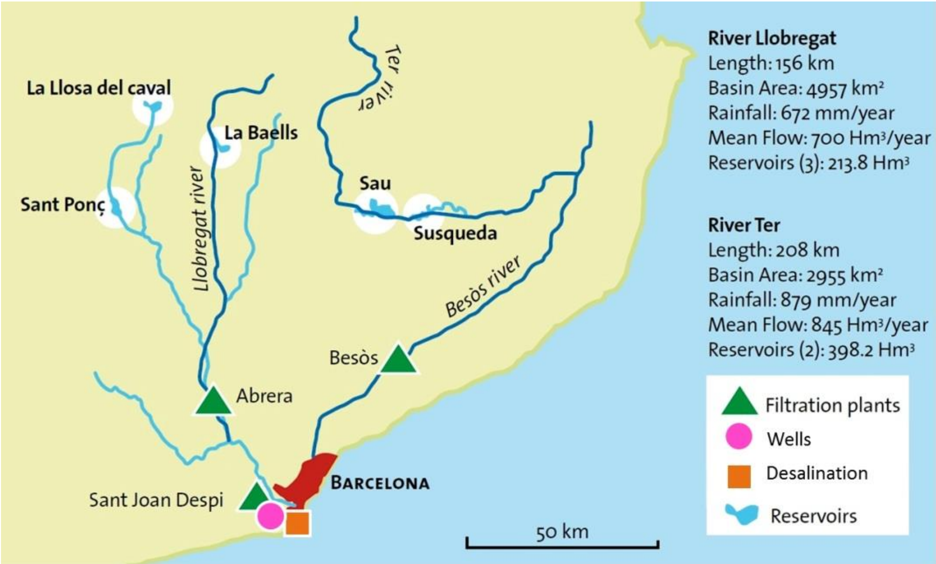



**Figure 2: Water supply to the Barcelona metropolitan area**

As it can be seen in Figure 3, it is important to point out that balance between supply and demand are lower than in other similar cities indicating that safety margins do not meet desirable levels and that the area is at risk of a water deficit that is likely to become significant. Actually, in the spring of 2008 for example, during the most acute drought since the 1940's the water shortage in the city of Barcelona was such alarming that the Catalan Water Agency (ACA), after many different restrictions (shutting off municipal fountains and beachside showers, prohibiting the filling of swimming pools, etc.) to no significant effect, decided to install the infrastructure needed to convey water shipped in from other countries (France) and Spanish locations (Tarragona, Almeria) from the industrial port of Barcelona to the purification plant (Sant Joan Despí). Transport by rail was also considered. However, shortly after work was completed, the rains fortunately came, and the installations were no longer used. Ultimately, few ships actually moored in the port of Barcelona to supply water.

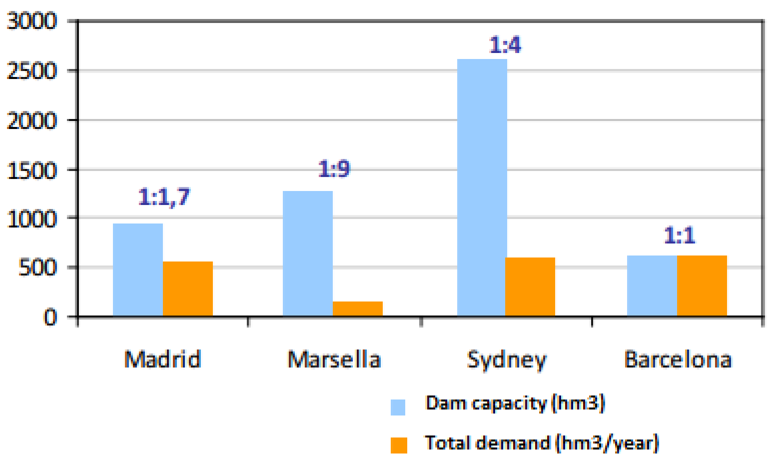



**Figure 3: Dam capacity and total demand of similar water supply systems**

Desalination plant was installed in Barcelona in 2009 with a capacity of 60 $Mm^3$/year. The plant assures the drinking water availability for the metropolitan area in case of drought periods or when there is an increase of the demand.  However since it was installed the city did not suffer any drought period and as desalinated water is more expensive than surface water, the desalination plant has been running just in the maintenance operation most of its lifetime.

In conclusion, all this data give an idea of the big importance of the Llobregat Delta Aquifer as a strategic water source for Barcelona.

# 4 Aquifer Storage and Recovery (ASR) system in the Llobregat area

The Llobregat basin has received the effect of human presence since historical times with hydraulic exploitation, water abstraction, channelization, and damming have added to mining exploitation, eutrophication, pollution, and the arrival of invasive species in successive steps of increasing pressures on the ecosystem (Sabater, 2012). So under all this pressures and interests, Llobregat basin has been converted in one of the best studied and monitored river basins in Europe and several groups from different universities and research institutions, both Spanish and European, have placed their efforts to understand the hazards and resilience of such a river system.

Overexploitation of the aquifer both for drinking water production and other purposes has resulted in a water table consistently below sea level, thereby fostering marine intrusion and groundwater salinization. Then, artificial infiltration of surface water through the riverbed and the deep recharge of treated water into the aquifer have helped ensure water availability at all times, even during the most extreme drought events.

Therefore DESSIN project aims to contribute to this recuperation of the Llobregat aquifer by boosting the deep recharge and also by demonstrating the valuation of the related ecosystem services.

## 4.1 Hydrogeological context

### 4.1.1 Hydrogeology

The Llobregat aquifer where ASR wells were drilled has two aquifer units in the Delta: the upper, phreatic unit, and the lower, semi-confined unit. This last one is the one basically targeted for water production, given the significant storage volume (110 $Mm^3$) and protection against polluting episode. Typical transmissivity values around 30,000 $m^2/d$ are usually obtained at the ASR wells, indicating that the aquifer is too transmissive if seasonal usage is desired. However, the objective of ASR is to store water underground and to be able to recover it at any moment as the whole system is subject to a very dynamic exploitation. Porosity is around 20%, while the average aquifer thickness is 20 m. The lower aquifer extends from 30 m to 50 m depth, up to a clayey layer. Sediments are coarse, as it was known thought the literature and as it was observed during the drilling works of the new boreholes for the DESSIN project (July 2014).

The water level in the alluvial gravels is normally several metres below the riverbed. However the alternating discharge/recharge regime operated by Agbar, and additional abstractions from the aquifer by industrial users, result in a complicated and dynamic groundwater regime. Since 1965 the water level measured in a piezometer at the Agbar pumping station has ranged between a maximum of +1 m a.s.l. (in 1997) to a minimum of —17 m a.s.l. (in 1990). This variation reflects the meteorological conditions and the consequent abstraction/ recharge operations by Agbar at Cornella.

### 4.1.2 Hydrochemistry

Both the main aquifer (deep aquifer) and the alluvial gravel aquifer (upper aquifer) have a similar Total Dissolved Solids (TDS) concentration, approximately 1300 mg/L, but there exist a clear distinction in their cation composition. The alluvial groundwater has a lower relative concentration of $Mg^{2+}$ and a higher relative concentration of $Na^{+}$ and $K^{+}$ than the plain groundwater. Furthermore, the alluvial groundwater has a lower $Na^{+}/K^{+}$ ratio (about 10) than the plain sands ($Na^{+}/K^{+}$ ratio range 30 to 90) (Martín, 2006).

Figure 4 shows the ionic water composition of 5 selected wells around the drinking water treatment plant of SJD. In general, groundwater samples from the aquifer are saturated with respect to calcite. Calcite has the important effect of buffering the pH of the groundwater within a observed range 6.8 to 7.8. Well 18 has a slight different water composition due to the influence of the Llobregat River compared to the other wells.

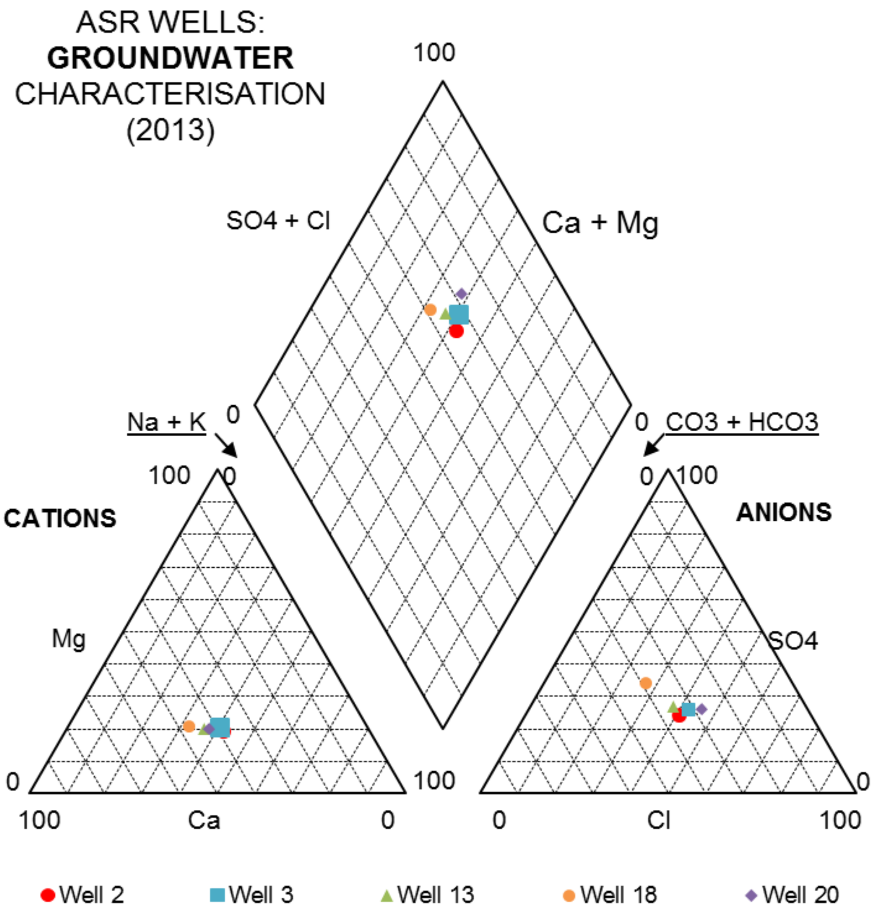



Figure 4: Groundwater characterisation in the deep aquifer in SJD

## 4.2 ASR system description

Until 1954, when the first treatment plant for surface water was built in Sant Joan Despí, all the water resources used in Barcelona and its metropolitan area came from groundwater of the aquifers in the lower valley and the delta of the Llobregat River. The growing demand for water in the zone of the Llobregat River delta, much greater than the aquifer's resources, brought about a progressive drop in the levels. In 1950, Sociedad General de Aguas de Barcelona started to force the aquifer recharge artificially by scarifying the riverbed in the zone where the aquifer connects with the river.

In 1966 Barcelona metropolitan area water supply capacity was increased with water from the Ter River to be added to the water resources coming from the Llobregat Delta (surface and groundwater). Therefore in some cases it was possible for production to exceed demand and in order to take advantage of this context, in 1969 seven extraction wells belonging to Aguas de Barcelona were adapted for recharging treated drinking water. The objectives of this action were to improve groundwater quality, to increase water table levels and to reduce the later pumping cost. Initially, Aguas de Barcelona started the recharge with drinking water in the seven existing wells being the design injection flow 50 L/s (Armenter, 2008). In the following years five new wells were constructed. Nowadays, there are 12 injection wells with a maximum recharge capacity of 75.000 $m^3$/d. New wells were equipped with a 15 to 20 m deep screen, which allowed increasing the injected water flow from 50 L/s to 100 L/s (Armenter, 2008). Optimum operation requires periodical unclogging by pumping a flow of 200 or 400 L/s, depending on the well, for 10 minutes, every 15 days of continuous operation. Figure 6 provides details of the ASR operation and maintenance.

It has to be pointed out that the distribution of the injection wells follows a transverse line through the aquifer. The hydraulic gradient decreases during recharge, and water table level increases. Quality parameters of the aquifer are monitored in order to assess the impact of ASR on groundwater quality. Petrovic *et al*. (2009) summarised in her article the benefits from managed aquifer recharge:

- Increasing of water reserves (for the Llobregat delta, a 3 m increase in level procures storage of almost 1 $Mm^3$ of water for each $km^2$ of surface area)
- Facilitating water transport and accessibility (because the delta of the Llobregat River's aquifer covers an area of about 110 $km^2$, water can be extracted at many different sites close to where it is used, with no need for long pipelines)
- Improving water quality (since the aquifer acts as a slow filter that retains suspended matter and associated contaminants in the infiltration zones)
- Increasing the phreatic level, which in turn saves energy expended in pumping and reduces the intrusion of brine.

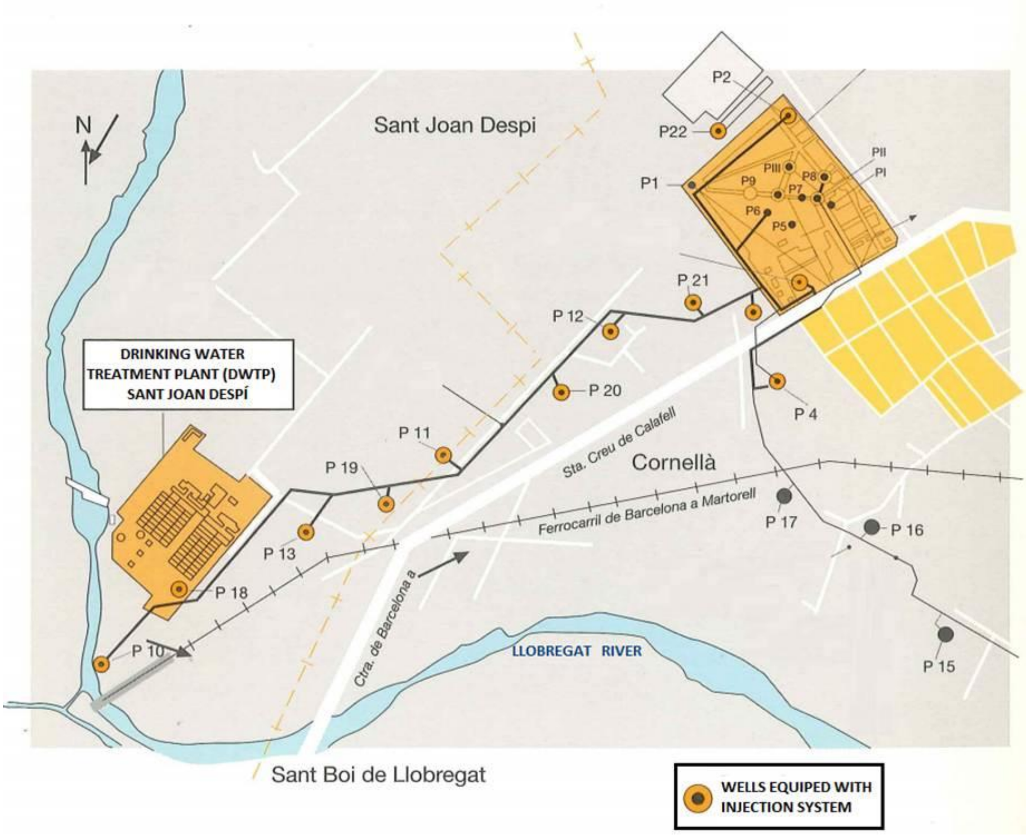



**Figure 5: Wells systems providing groundwater to Sant Joan Despí DWTP**

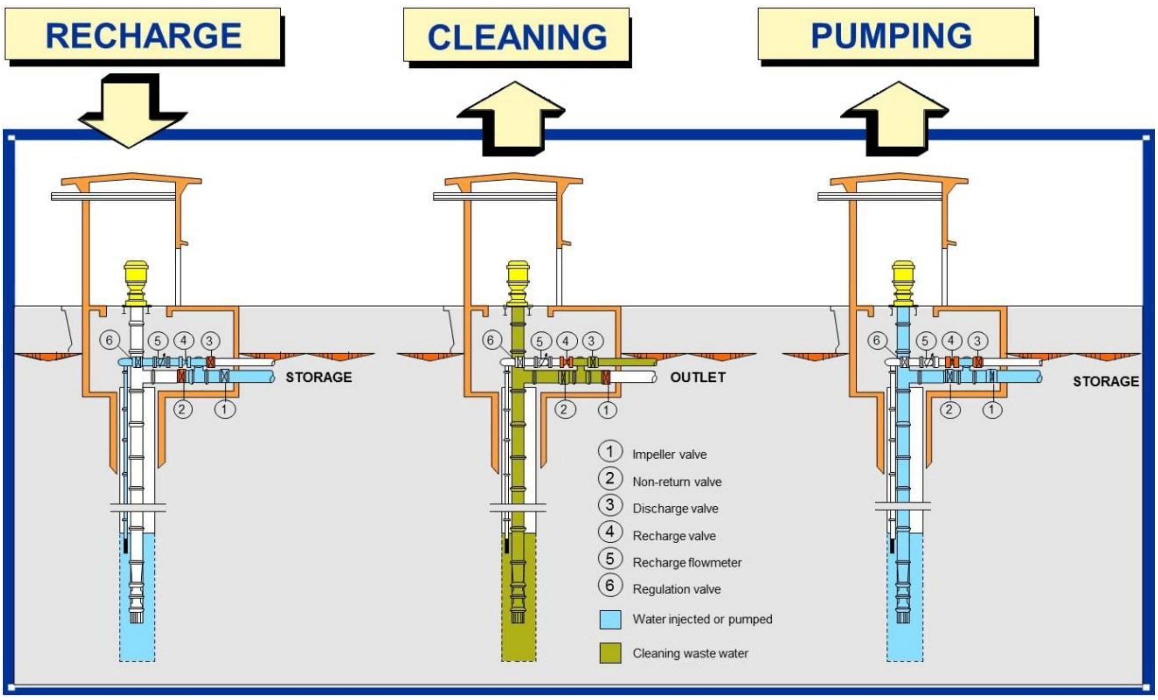



**Figure 6: Sant Joan Despí ASR maintenance operation**

## 4.3 Integration of the ASR system in the DWTP scheme

### 4.3.1 Water treatment evolution

The DWTP of SJD is one of the most complete plants in Europe. Since it was first commissioned in 1955, the plant has undergone several extensions and reforms, in all cases with the aim of allowing it to fulfil its supply commitments with every guarantee as regards health, as required by current legislation. At the beginning the plant had only a conventional treatment composed by a coagulation flocculation, sedimentation and sand filtration. In 1977 the sand filtration was replaced by a granular activated carbon filtration. In 1991 it was installed again the sand filtration in the conventional treatment and it was added also an advanced treatment with ozonisation and granular active carbon filtration. Finally in 2010 it was installed another advanced treatment that treats a fraction of the SFW by ultrafiltration, UV disinfection, Reverse Osmosis and Remineralisation. Figure 7 represents graphically the historical evolution of the treatment chain in SJD.

Historically the water injected in the ASR system of SJD has been always the potable water that came from the last treatment existing and always being disinfected with chlorine. So every time that the treatment plant was updated, the water quality has been improving but the treatment costs and the energy requirements have been increasing thus reducing the sustainability of the ASR system. As the cost increased the water production was adjusted with the demand and now no surpluses are generated to be injected in the aquifer. So despite the suitability of the alluvial aquifer and the good results achieved in the ASR the system, this has been operated fewer and fewer and since 2010 the injection is stopped.

The aim of the demonstration within DESSIN project is to study the effect of the injection of a pre-potable water in order to return the economic sustainability of the ASR system. With DESSIN scheme (Figure 8) SFW it is going to be injected in the aquifer and later it will be recovered and returned in the same point of the groundwater catchment input in the treatment plant. Actually, the treatment scheme from river water to injected water proposed is the same that it was in operation from 1969 to 1977 but without the disinfection process with chlorination. So one of the goals of the DESSIN project in Llobregat site will be to analyse if it is possible to inject SFW without disinfection without having clogging problems in the well and studding the effects of this kind of water in the aquifer.

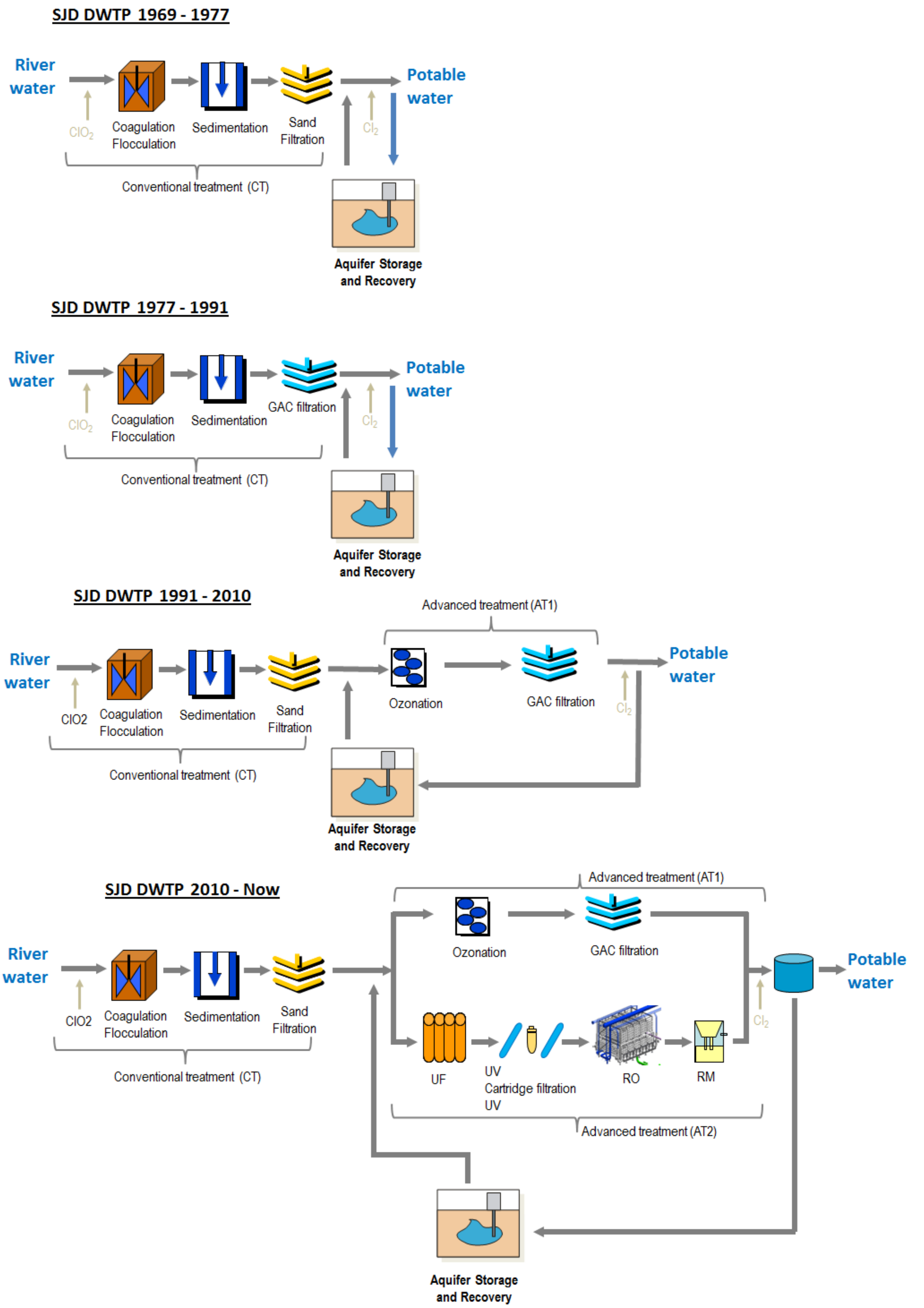



**Figure 7: Sant Joan Despí Drinking Water Treatment plant and ASR system evolution**

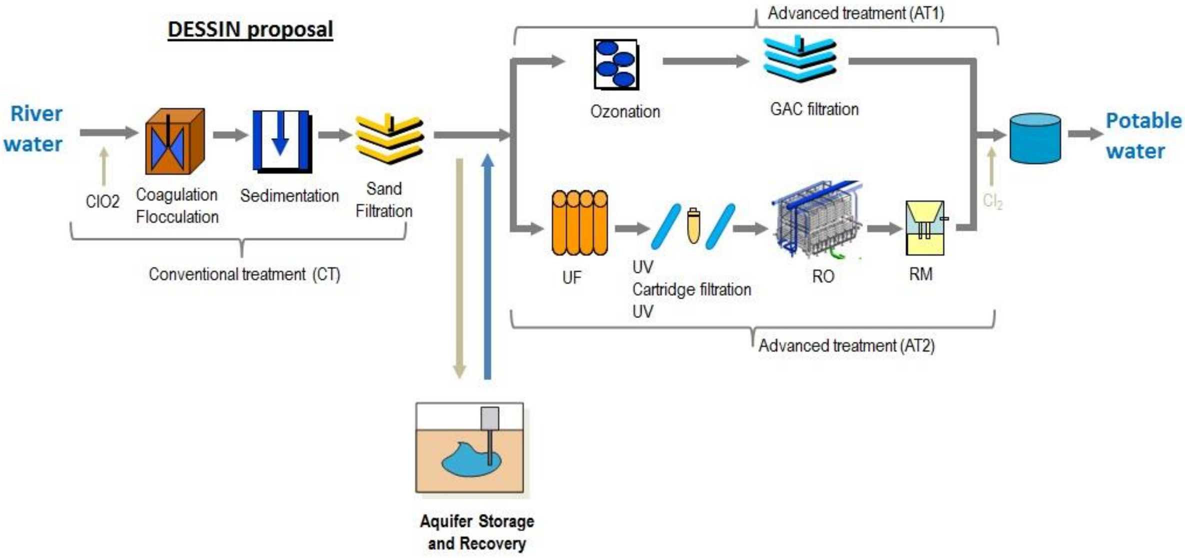



**Figure 8: DESSIN scheme proposal for ASR operation**

### 4.3.2 Historical ASR operation

As explained before, the operation of ASR in Sant Joan Despí system started in 1969 and until 1997 has been operating with values ranging from 1 to 14 $Mm^3$/year. Depending on environmental conditions as climate, precipitation, river water quality and water availability, the ASR system had been operating with different volumes of water injected. From 1969 until 1997 the system has been operating almost continuously with an average around 5 $Mm^3$/year. More recently, from 2001 until 2009 the system had been operating with a lower volume of around 1 $Mm^3$/year. Finally after the implementation of the second advanced treatment with reverse osmosis in 2010, the injection has been stopped and the system only worked extracting water from the aquifer.

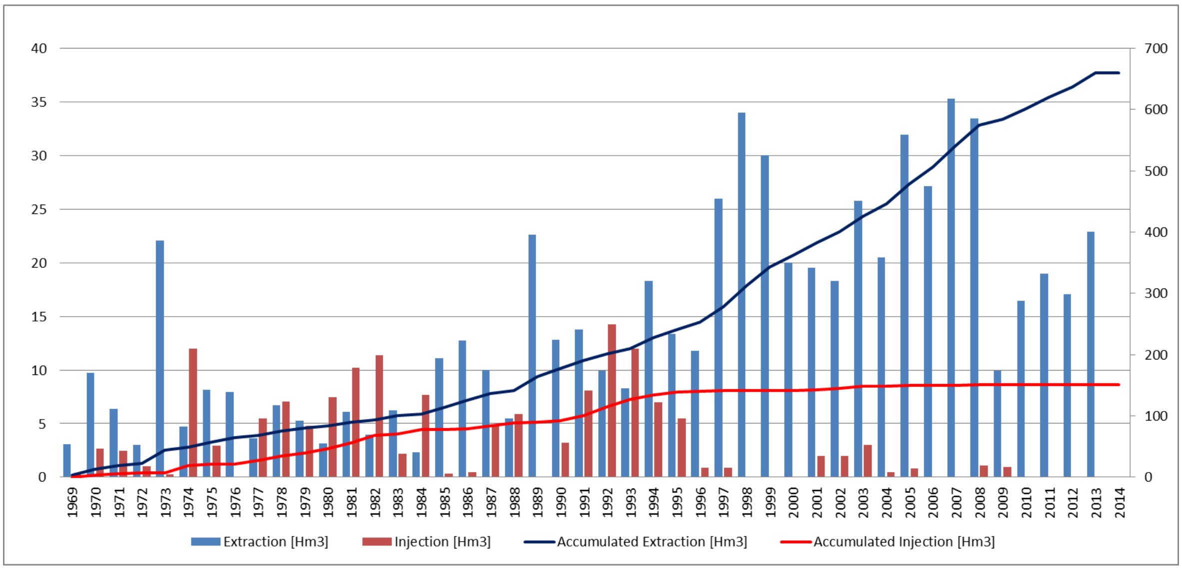



**Figure 9: Extraction and injection volumes in ASR system of Sant Joan Despí**

# 5 International water quality standards for ASR operation

## 5.1 Legal framework

### 5.1.1 European Union

Within European Union, the managed aquifer recharge (MAR) is regulated by the Water Framework Directive (WFD, 2000). The article 11 of WFD has a Programme of measures where it regulates the measures to control this managed recharge in the following terms: 3. 'Basic measures' are the minimum requirements to be complied with and shall consist of: *[...] f) controls, including a requirement for prior authorisation of artificial recharge or augmentation of groundwater bodies. The water used may be derived from any surface water or groundwater, provided that the use of the source does not compromise the achievement of the environmental objectives established for the source or the recharged or augmented body of groundwater. These controls shall be periodically reviewed and, where necessary, updated;*

Within the Annex II in the Groundwater part is it established the point 2.1 that: *Member States shall carry out an initial characterisation of all groundwater bodies to assess their uses and the degree to which they are at risk of failing to meet the objectives for each groundwater body under Article 4. In this analysis is included the managed aquifer recharge.*

Finally, WFD also establishes that it will be done an special characterization that will include: Anex II, point 2.3.e) *the rates of discharge at such points*, Anex II, point 2.3.f) *the chemical composition of discharges to the groundwater body*, and Anex II, point 2.3.g) *land use in the catchment or catchments from which the groundwater body receives its recharge, including pollutant inputs and anthropogenic alterations to the recharge characteristics such as rainwater and run-off diversion through land sealing, artificial recharge, damming or drainage.*

Later on each Member State transpose the directive in their legislation. From recharging water quality point of view there is no any specific requirements and it depends by the authorization of each groundwater body authority.

In the Spanish legislation there is only a specific quality requirement for managed aquifer recharge by direct injection just in the case of wastewater reuse, defined in the Royal Decree 1620/2007 (RD, 2007). There it defines parameters as Nematodes, *Escherichia Coli*, suspended solids, turbidity, total nitrogen and nitrate (Table1):

**Table 1: Water quality requirements for aquifer recharge by direct injection in RD1620/2007**

| | Intestinal nematodes | E. Coli | Suspended Solids | Turbidity | Other criteria |
|---|---|---|---|---|---|
| 5.2 Aquifer recharge by direct injection | 1 egg/10L | 0 CFU/100mL | 10 mg/L | 2 NTU | NT: 10 mg/L<br>NO3-: 25mg/L |

### 5.1.2 USA

In the USA, the water used for injection is usually treated to meet drinking-water quality standards for two reasons. One is to minimize clogging of the well-aquifer interface, and the other is to protect the quality of the water in the aquifer, especially where it is pumped by other wells in the aquifer for potable uses. Also, the water used for well injection in the USA is often chlorinated and has residual chlorine concentration about 0.5 mg/l that is injected in the recharge well. Thus, whereas secondary sewage effluent can readily be used in surface infiltration systems for soil-aquifer treatment and eventual potable reuse, effluent for well injection should at least receive tertiary treatment (sand filtration and chlorination). This treatment removes remaining suspended solids and protozoa like *Giardia s.p.* and *Cryptosporidium s.p.*, and parasites, like helminth eggs by filtration, and bacteria and viruses by chlorination, ultra violet irradiation or other disinfection. In the USA the tertiary effluent is often further processed with membrane filtration (microfiltration and reverse osmosis) to remove any pathogens that might have escaped the tertiary treatment and also nitrogen, phosphorus, organic carbon and other chemicals. Dissolved salts are almost completely removed.

However, different states address some issues differently (Pyne 2003). While in Florida recharge water quality must meet all primary drinking water standards at the wellhead prior to recharge, in Arizona same quality should be measured at the edge of a “compliance zone” around the ASR well, up to 200 m away. The main difference is that Arizona regulation considers full advantage of the demonstrated ability of aquifers to improve water quality due to natural treatment processes. In Arizona, ASR storage typically occurs in fresh water, unconsolidated sand aquifers that are utilized for drinking water supplies, whereas in Florida, ASR storage zones are generally brackish and are therefore unsuitable for potable water supply except following desalination treatment. In Wisconsin, compliance with water quality standards is measured either at the water treatment plant or in the distribution system during recharge. It is also measured at the ASR wellhead during recovery, in addition to compliance with state groundwater standards at a property line monitor well in the storage zone. However, an exemption for trihalomethanes (THMs) was implemented during 2001, providing a compliance zone radius of 350 m. ASR wells in Wisconsin are generally in sandstone aquifers. In North Carolina, water quality compliance with drinking water standards is measured at the edge of a mixing zone in a clayey sand aquifer around the ASR well, not at the wellhead prior to recharge.

While all four regulatory programs comply with federal law (1974 Safe Drinking Water Act), the United States Environmental Protection Agency (EPA) Underground Injection Control (UIC) regulations promulgated in 1981 pursuant to this law established that primary drinking water standards should be measured at the wellhead, not in the aquifer. As such, Pyne (2005) concluded that the federal law and the federal regulations are inconsistent. Arizona has followed federal law. Florida’s standards are in many ways more restrictive and more costly to achieve compared to those regulatory programs that evaluate compliance at a monitor well in the aquifer.

Other concerns have focused on potential leaching of metals such as arsenic, mercury and uranium from the limestone into the recovered water or into the surrounding aquifer; potential contamination of the aquifer with disinfection byproducts (DBPs); potential contamination with pathogenic microbiota such as bacteria, viruses and protozoa; and mixing with surrounding brackish water so that recovery efficiency is reduced to below acceptable levels.

Pyne (2005) concludes that a case like Florida with such a high level of pretreatment is quite conservative and would greatly increase the cost to taxpayers for capital investment in treatment facilities required to achieve these policy objectives. By comparison, taxpayers in Arizona, North Carolina, the Netherlands and Australia rely upon natural processes in the aquifer surrounding an ASR well to achieve these objectives at no additional cost.

### 5.1.3 Australia

The first Australian guidelines of stormwater and reclaimed water quality for injection into aquifers for recovery and reuse were made by Dillon and Pavelic in 1996. This replaced previous documents on managed aquifer recharge of reclaimed waters and was a first attempt (internationally) to provide a sound basis for the injection of non-potable waters into aquifers for a range of beneficial uses. These guidelines were an outcome of a two-year Urban Water Research Association of Australia study that reviewed international practice and guidelines for managed aquifer recharge of waters by injection. The principles, objectives, and guideline values for maximum contaminant levels in water for a range of beneficial uses (environmental values) were founded on Australia's National Water Quality Management Strategy.

While the guidelines adhere to internationally accepted principles they are quite different from those currently used to regulate ASR sites in other parts of the world for two reasons. They do not presume potability as an essential and sole objective, and they allow for demonstrated sustainable attenuation of contaminants by natural processes in aquifers. Currently there are pressures within the USA to adopt the principles embodied in these Australian guidelines, particularly in arid areas where alternative sources of supply are possible.  The guidelines covered licensing, pre-treatment, monitoring, guidance for maximum contaminant concentrations in injecting water, residence time prior to recovery and management of ASR operations. The report also made recommendations on revising the guidelines, identify knowledge gaps, concentrate research at selected sites and establish a national ASR research program to coordinate and conduct ASR research. Moreover it serves for collating all monitoring data and reports from Australian ASR sites and to produce a design manual for ASR.

These aquifer and storage and recovery operations have been successfully going since 1993 in South Australia, and number and size of sites are expanding where using limestone, fractured rock and alluvial aquifers. Stormwater has been seasonally injected into different aquifers and has been demonstrated that pathogen attenuation rates in aquifers were adequate for irrigation use and generally also meet local requirements for potable use of recovered water (Dillon and Pavelic 1996).

Regarding specific injecting water quality issues, Australian Guidelines expose that definitive guideline values are a highly desirable target for sustainable Australian managed aquifer recharge. This target is yet to be achieved and is an area of active research, so in the guidelines it just mentions the Netherlands (Olsthoorn 1982) and Pérez-Paricio (1999) recommendations pointing out that offer a useful starting point, but this single guideline values are inappropriate for Australia because it does not consider the soil or aquifer conditions, which are known to be important. Finally the guidelines conclude that to meet operational requirements, site-specific evaluation to assess clogging potential and identify water-quality targets suited to the aquifer is needed at all sites.

## 5.2 Risk of groundwater contaminants in ASR and recent findings

Experience has shown that very close to the ASR well, typically within a radius of a few meters, there is a development of a treatment zone where ambient microbial activity is accelerated, geochemical changes are more prevalent, and water quality changes occur (Pyne 2003). Changes in water quality have been generally minor, so that treated drinking water quality standards that are met during recharge are also generally met during recovery process, involving transport along the aquifer.

However, subtle changes in some constituent concentrations have been noted at several ASR sites, and some of these are the subject of considerable public interest. Most of these subtle water quality changes are beneficial, improving recharge water quality during storage. In particular, significant reductions in nitrogen, phosphorus, microbiota, DBPs and other constituents have been observed during ASR storage. Where high concentrations of some water quality constituents are naturally present in the storage zone, such as iron, manganese and hydrogen sulphide, it has been possible to leave these constituents in the aquifer and not produce them in the recovered water. Where constituent concentrations have increased in the recovered water, this effect has generally proven to be transitional, reflecting natural subsurface physical, geochemical and microbial treatment of the recharge water around the well during early cycle testing.

This section lists the potential risks occurring in ASR operation. They can be due to their presence in the injection water, or the chemical reactions occurring in the aquifer.

### 5.2.1 Disinfection by-products

Disinfection by-products is one of the most regulated parameters in USA and EPA has established primary drinking water standards that limit its concentrations in public drinking water supplies in order to protect public health.

DBPs such as THMs and haloacetic acids (HAAs), which are cancer-causing constituents at elevated concentrations, are formed when water containing natural dissolved organic carbon is chlorinated for disinfection[1]. Other treatment processes are available to provide adequate disinfection of public drinking water supplies but which may provide better control of THM and HAA formation, such as chlorination followed by dechlorination, chlorammoniation, ozonation and ultraviolet (UV) radiation. However, chlorination still represents a widely used disinfection treatment process. From long-term experience in Peace River ASR wellfield in Desoto County, Florida, presented in the 1996 AWWARF report, it is evident that the microbial and other processes contributing to DBP attenuation are sustainable. Supplemental research has shown that HAAs disappear within a few days, primarily due to aerobic microbial reactions occurring underground in the ASR storage zone (Pyne et al, 1996). THM concentrations are eliminated over a few weeks, primarily due to anaerobic microbial reactions that typically become established within a few days after ASR recharge. This occurs once the chlorine in the recharge water dissipates underground.

[1] The pre-treatment of the sand filtered water in SJD consists in dioxichlorination. This treatment generates fewer amounts of DBPs than the direct chlorination.

Reducing conditions are re-established in the aquifer due to subsurface microbial activity, geochemical changes, and the effects of mixing and dilution in the buffer zone surrounding the ASR well. Where anaerobic conditions do not exist in the storage zone, such as may be expected in a surficial aquifer, THM reduction is minimal or absent (Fram *et al.*, 2003).

### 5.2.2 Pathogens

During the past few years, extensive microbial research has been conducted by CSIRO in Adelaide, Australia, in brackish, limestone, confined aquifers. This research has shown that native microbiota naturally present in the aquifer is effective in attenuating pathogenic microbiota that are introduced with the recharge water. In these researches it has been defined time periods for pathogen attenuation on the order of a few days for each log cycle, or 90 per cent reduction in concentration. It is also pointed that other factors that attenuate microbiota concentrations include temperature, salinity, and probably other mechanisms.

D.E. John (2004) made a literature review of survival studies on microorganisms in groundwater with a total of 19 studies. Data on inactivation were extracted and combined to analyse trends and central tendencies. The inactivation rate was obtained by plotting of the decrease in numbers of microorganisms over time and examining the slope of line (k values which are inactivation rates, expressed as a decrease in log10 number/day [log/d]). These rates were taken from the publication or were developed from the original data in the paper. In addition many survival studies use the number of days for 90% reduction or 99% reduction, instead of using k values (similar to using the half-life of chemicals). The median value for inactivation rates across all temperatures for coliphage (0.079 log/d, n=72), poliovirus (0.081 log/d, n=41), echovirus (0.079, n=15), and coliform bacteria (0.071 log/d, n=22) were almost identical. In addition, the Q1 values (lower boundary of the middle 50% of data points) of data sets for coliform bacteria, enterococci, Salmonella spp., coliphage, poliovirus, and echovirus were all similar, ranging from 0.032 to 0.057 log10 /day inactivation (1, 3, 9, 16, 19, 20). These rates in the literature correspond to days for 90% reduction in a range between 17.5 to 31.3 days.

Temperature has always been known to act as one of the key variables affecting microbial survival. Several investigators observed that virus inactivation increased with increasing temperature, but similar trends for bacteria were observed less consistently. Other studies also described an increase in inactivation rates in non-sterile vs. sterile water sources; however, the opposite was also observed in some cases. In others still, no effect of sterilizing the environmental water source was observed.

In the Australian Guidelines for water recycling in managed aquifer recharge are defined pathogen inactivation rates in aquifers taken from studies of some sites where in situ decay studies were used. Observed for microorganisms in situ in Australian aerobic and anoxic aquifers, it defines maximum inactivation times for 1-log reduction of 3 days for *E. Coli* and 6 days for *Salmonella typhimurium* and for Bacteriophage MS2. Martin and Dillon (2002) in “Future Directions for South Australia” recommend for faecal coliforms a maximum of 10,000 colony-forming units per 100 mL. They also point that allowing for 1 log cycle removal per 10 days (which is conservative), faecal coliforms would be depleted to the irrigation water quality guideline after 10 days and to near potable standards after 50 days residence in the aquifer.

### 5.2.3 Nitrogen

In "Future Directions for South Australia" (Martin and Dillon, 2002) it is recommended a maximum of 10 mg/L for potable reuse, subject to ammonia concentrations being less than 0.5 mg/L. Swiss guidelines restrict ammonia to 0.5 mg/l. Denitrification within the aquifer should not be relied upon to attenuate nitrate concentrations as this may cause gas binding. For irrigation reuse the nitrogen concentration in recovered water should be sufficiently low (typically less than 10 mg/L) that the nitrate concentration in irrigation leachate is environmentally sustainable.

### 5.2.4 Arsenic

Injection of non-native water in an aquifer can result in a release of chemical constituents from the receiving aquifer into the injected water that is detrimental to water quality of the recovered water or, alternatively, condition the aquifer to retain or control a natural contaminant. In certain regions of the world, elevated arsenic concentrations are present in water recovered from ASR Systems. Understanding the mechanism of arsenic dissolution and mobility in an aquifer upon the introduction of recharge water is crucial in order to design pretreatment and storage protocols to circumvent this dissolution. In the US, EPA's lowering of the Drinking Water MCL for arsenic from 50 to 10 µg/L has added increased incentive to further investigate the complex relationships between storage zone mineralogy, injection water chemistry, and arsenic mobility. Although there are many forms of organic As, the most common organic species are monomethylated and dimethylated As(III) and As(V) under natural conditions. More information about the arsenic mobilisation can be found in Bhattacharya *et al*. (2007), Welch and Stollenwerk (2003) and O'Day *et al.*, (2005).

## 5.3 Well operation considerations in ASR: clogging

Within the operation of an ASR process, the most important issue to control is the clogging that it could be developed in the well. Clogging may be defined as the reduction of available pore volume or reduction of available filtration area due to a combination of physical, biological and chemical processes. Consequently, the immediate effect of clogging is a diminution in the intrinsic permeability of a system, leading to a drop in infiltration rate or in specific flow. As it can be seen in Figure 10, clogging could be found in different places of the well as borehole wall or well screen. Perez-Paricio (1999) defined that clogging is a rather complex phenomenon, but the following causes can be identified:

- Physical retention of particles during their passage through the medium, termed physical clogging.
- Formation of a viscous phase caused by biological reactions, mainly bacteria in the aquifer and algae in surface water: bioclogging.
- Chemical reactions that provoke the precipitation of minerals, or chemical clogging.
- Generation of gas and air bubbles that reduce the available volume for water, gas generation.
- Compression of the clogging layer itself because of an excess of water height in surface systems: compaction.

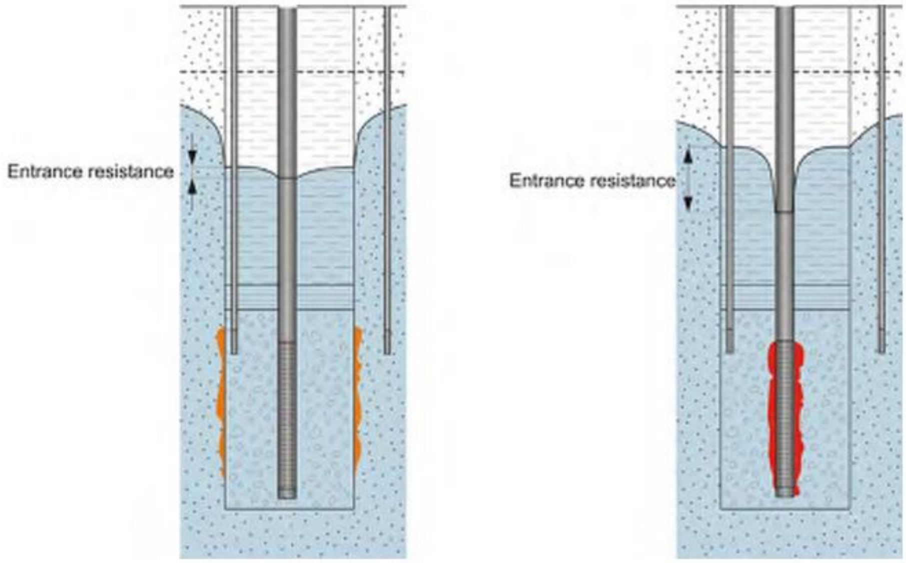



**Figure 10: Clogging in borehole wall (left) and in well screen (right)**
Source: Zwart (2007)

Clogging is a site-dependent phenomenon and is influenced by the natural heterogeneity and by water type and climatic conditions. This implies important variability in the field conditions so extrapolate site empirical relationships could lead to errors. Then while comparing different site data is interesting as a reference, each company or water manager try to obtain their own operational protocol for clogging management.

If clogging produces a remarkable reduction in the effectiveness of infiltration systems then some re-development methods must be used causing the stopping of the system and needing an investment of energy to restore the initial capacity.

Decades of operational experience have shown that clogging prevention is a better option than renovation, particularly for well-injection systems. While various renovation measures that can yield excellent results exist, ensuring that recharge water meets the appropriate water-quality target through adequate pretreatment is a key factor in ensuring successful and sustainable long-term MAR operations. Although all forms of clogging can be avoided through pretreatment, in practice there is a trade-off between the costs associated with pretreatment, and the degree of clogging that is acceptable in terms of the type and frequency of renovation that would be required.

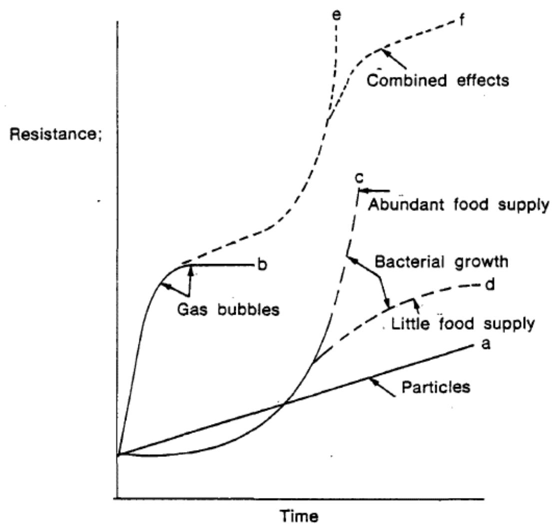



**Figure 11: Typical clogging evolution**
(a) Free suspended matter(b) Gas or air bubbles (c) Bacterial growth with large food supply (d) With a limited food supply (e and d) Various simultaneously forms.
Source: Olsthoorn (1982)

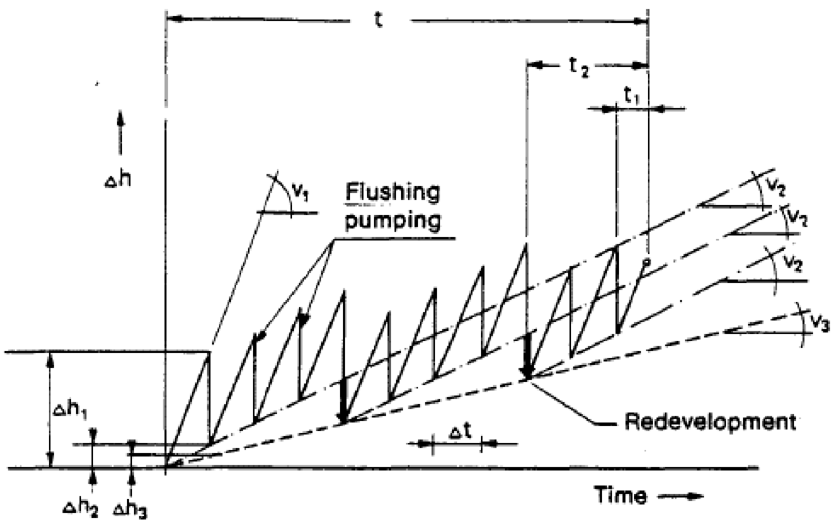



**Figure 12: Schematic variation of water head in an injection well as a result of clogging, flushing pumping and redevelopment**
Note: The time scale can be varied at option by choice of injection rate and water quality. Source: Olsthoorn (1982)

### 5.3.1 Physical clogging

Perez-Paricio (1999) described physical clogging as the mechanisms that affect the movement, deposition or detachment of inert suspended particles in recharge water, whose consequence is a diminution of porosity. Depending on their size, suspended particles are usually classified under three categories:

- Colloidal particles: Their diameter is lower than approximately one micron; this implies that they are subject to physicochemical surface forces, owing to their great specific surface
- Intermediate particles: Their diameter varies between 1 and 30 µm, so that they are subject to both surface and volumetric forces.
- Large particles: Their diameter is larger than 30 µm and large particles are only affected by volumetric forces, such as sedimentation, inertia, direct interception (straining) and hydrodynamic effects.

The major factors affecting the particle deposition are solids concentrations of the recharging water, soil size and distribution, and porosity. Solids concentrations can be measured as suspended solids (mg/l), turbidity (NTU) or Modified Fouling Index (MFI).

For suspended solids (SS) experimental results have shown that recharge water used for ASR should have levels of SS < 2 mg/l to avoid physical clogging problems (Okubo and Matsumoto, 1983). This parameter was obtained in one experiment using 40 cm column of sand composed of 0.25-0.42 mm diameter and with saturated hydraulic conductivity of column about 43 m/day. In other experimental study (Rinck-Pfeiffer, 2000) three columns were packed with aquifer material from a core of the target sandy limestone aquifer and filled for a period of 22 days with synthetic wastewater. With levels of suspended solids between 3±4 mg/L hydraulic conductivity (K) decreased from 0.78 m/day to 0.062 m/day in 7 days. However previous research in the same aquifer system has shown that SS levels in excess of 25 mg/l have not caused clogging in a calcareous aquifer at a stormwater ASR site in South Australia (Pavelic *et al*., 1998). Actually Martin and Dillon (2002) say in the future directions for South Australia that values of 30 mg/L of suspended sediments have been acceptable in a variably cemented limestone aquifer. For total dissolved solids a maximum of 500 mg/L is recommended for potable reuse and 1000 mg/L is desirable for non-potable reuse.

Modified Fouling Index (MFI) was developed by Schippers and Verdouw (1980) to determine the fouling characteristics of reverse osmosis membranes, but is also one of the most used parameters to characterize physical fouling in managed aquifer recharge. MFI is the slope of the straight part of filtration time divided by filtrated volume (t/V) versus filtrated volume (V) curve. It is necessary to plot t/V as a function of V in order to obtain the MFI, although experimental apparatus and conditions should be considered too. The volume of filtrate is measured with a measuring cylinder every 30 seconds for a maximum of 20 minutes at a pressure of 210 kPa. Time is counted when the desired pressure has been achieved. Olsthoorn (1982) evaluated that for injection wells, MFI-values less than 3 are good and over 10 - 15 are bad.

In the Recycled Water Quality Standards Study of West Basin Municipal Water District in California is set that the MFI standard for injecting water should not exceed 1.25 s/L$^2$ (average) and 2 s/L$^2$ (maximum). Turbidity values around 1 to 5 NTU are acceptable. The guidelines of The Netherlands and Germany recommend values of Turbidity below 1 NTU.

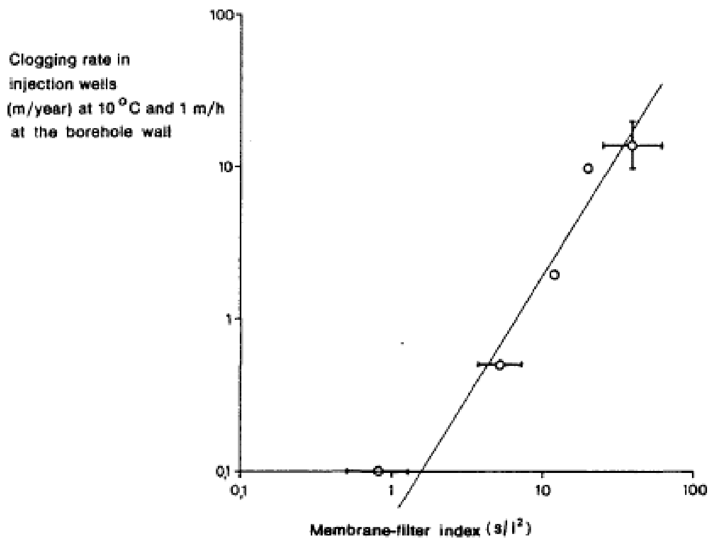



**Figure 13: Relation between injection-well clogging and membrane-filter index of Dutch injection wells**
Source: Olsthoorn (1982)

### 5.3.2 Bioclogging

Although protozoa and viruses are present in groundwater and in fact are receiving a great deal of attention because sanitary concerns, bacteria play the most relevant role regarding biological clogging. Baveye *et al.* (1998) point out at four different mechanisms of bacterial clogging: (1) the accumulation of cell bodies in the porous medium, (2) the production of bacterial extracellular polymers, basically polysaccharides, (3) the entrapment of gaseous products, especially nitrogen (denitrification) and methane (methanogenesis), and (4) the microbially mediated accumulation of insoluble precipitates.

Many bacteria form a biofilm, which is composed by cells and extracellular material, basically polysaccharides. Biofilm is treated as a separated phase by many researchers. The formation of a biofilm attached to the medium reduces the porosity, thus causing clogging of recharge devices.

Bacteria are also essential because of their catalytic capacity with respect to some fundamental chemical reactions. Bacteria need external electron acceptors for catabolism, and carbon (organic if heterotrophic and inorganic if autotrophic) for synthesis. Oxygen is the most effective electron acceptor in terms of energetic yield, so that its availability is crucial for bacterial development. Generation, transport and consumption of certain basic species depend strongly on bacteria and aquifer conditions.

The most important parameters to monitor bioclogging are assimilable organic carbon (AOC), dissolved organic carbon or chlorine. AOC refers to a fraction of the total organic carbon (TOC), which can be utilized by specific strains or defined mixtures of bacteria, resulting in an increase in biomass concentration that is quantified. AOC typically comprises just a small fraction (0.1– 9.0%) of the TOC (van der Kooiji, 1990). AOC represents the most readily degradable fraction of BDOC/BOM.

Hijnen (1991) investigated the influence of the concentration of AOC in water on clogging with filter beds operated under well-defined laboratory conditions using acetate as a model substrate. It was concluded that acetate concentrations in the water as low as 0.01 mg C/l promoted clogging with the main head loss, caused by bacterial growth, in the first centimeter of the sand bed.

AOC is determined microbiologically by plating out and incubating a water sample for growth of bacteria of the type Pseudomonas fluorescence counting the bacterial colonies, and expressing the results in terms of the carbon concentration of an acetate solution producing the same bacterial growth. AOC can be less than 1 % of dissolved organic carbon (DOC). If a residual chlorine level is maintained before recharge higher AOC levels are probably tolerable. Rather than AOC, biodegradable organic carbon or BDOC is often preferable as a biological clogging parameter, especially for higher organic carbon concentrations. BDOC is easier to determine than AOC, because BDOC is based on degradation of organic carbon by passing the water fine gravel, water is applied to the surface of the backfill, through laboratory soil columns or in batch tests with soil slurries.

Okubo and Matsumoto (1983) trials concluded that DOC should be < I0 mg/L to maintain a high infiltration rate during a long inundation period. Swiss guidelines, set maximum DOC concentration of 2 mg/l.

In the Recycled Water Quality Standards Study of West Basin Municipal Water District in California recommend that an adequate total chlorine residual should be maintained throughout the barrier system, maintaining total chlorine residuals between 2 and 4 mg/L throughout the barrier to minimize biofilm formation in the distribution system, well casings, and aquifer material.

### 5.3.3 Chemical clogging

Chemical reactions leading to precipitation of minerals can occur because of incompatibility between recharge water and groundwater causing an immediate reduction in porosity (Perez-Paricio, 1999). Then chemical clogging is more susceptible in pumping and recharge wells because oxidation can cause precipitation of iron compounds if iron-laden groundwater exists. Moreover in many real situations is difficult to differ between chemical and biological processes because many chemical reactions are catalysed by bacteria, especially redox ones. Precipitation is controlled by chemical composition of recharge water and groundwater and by aquifer mineralogy and physical variables like temperature and pressure. Bacterial metabolism, addition or depletion of oxygen and carbon dioxide, and presence of catalysts also affect mineral precipitation and dissolution.

Perez-Paricio (1999) stated that optimal value of pH=7.5 is assumed to minimise iron bacteria growth, chemical precipitation and corrosion of metallic parts. Lower values enhance iron bacteria growth and corrosion, while higher ones are likely to cause encrustation. In Clogging Handbook (Perez Paricio, 1999) it was described literature parameters regarding chemical clogging:

- Lucas et al. (1995) show that a serious risk for iron oxyhydroxides precipitation exists when concentration of ferrous iron in groundwater is between 11.2 to 13 mg/l.
- Hills et al (1989) use the Langelier saturation index (LSI) to study the potential for calcite precipitation. This index provides the pH of equilibrium based on main ion concentrations of groundwater, and then if measured pH is greater than equilibrium pH precipitation can

occur, while dissolution is likely when measured pH is lower. Temperature is very important as regards LSI.

- Ford (1990) concludes that a Ryznar index below 7, TDS higher than 150 mg/l, pH above 7.5 and high concentrations of calcium and carbonates should be avoided.

The Ryznar stability index (Driscoll 1986) was introduced to reflect more accurately the encrusting or corrosive tendencies. It is based on the LSI, and is widely used for predicting the reactions of metal objects in saturated subsurface environments. In summary, it can be said that water is corrosive if the Ryznar index is larger than 7, and encrusting if lower.

### 5.3.4 Corrosion control

Corrosion defines the process of natural reversion of electrochemically produced metals to their original former state. It is a physicochemical phenomenon that tends to destroy a material that is not in equilibrium with the surrounding liquid. In order to simplify the description, two main corrosion types can be distinguished (Driscoll 1986):

a) Chemical corrosion. A component in water causes rapid removal of material from the metal alloy. Components are, in general, the following: carbon dioxide, oxygen, hydro-sulphidric acid, hydrochloric acid, chlorine and sulphuric acid.

b) Electrochemical corrosion. Flow of an electric current facilitates corrosion: redox potential variations take place in metal surfaces and water acts as an electrolyte. Bimetallic corrosion is typical, because the less noble metal gets the anode and suffers from corrosion

Degalier (1987) makes an extensive review of corrosion causes. Corrosion can be originated by: (1) the existence of a corrosive water inside or outside a well; (2) galvanic effects between parts of a metal in contact with differently composed water or between different non-isolated metals; (3) the activity of specific bacteria (e.g., sulphate reducing bacteria); and (4) electric currents caused by moving water.

Pérez-Paricio (1999) exposes that although in general corrosion is not relevant, assuming that recharge wells and pipes have been properly constructed, it can be very problematic if design is deficient. Moreover, redevelopment techniques that make use of corrosive acids can aggravate this situation. Design guidelines consist in avoiding work-hardened materials, stressed joints, high recharge temperatures and flows, or the production of several harmful gases, such as dissolved oxygen, carbon dioxide, methane and hydro-sulphidric acid. Redevelopment methods must take into account these considerations and opt by an appropriate solution.

According to Ford (1990) corrosion of ferrous metals is accelerated when the following conditions are accomplished: Ryznar index is above 9, pH below 7, oxygen concentration greater than 2 mg/l, total of suspended solids (TSS) greater than 1000 mg/l, carbon dioxide above 50 mg/l, chloride concentration greater than 500 ppm, and if there exists hydrogen sulphide.

## 5.4 Summary of international quality standards

Experience has shown that a wide range of parameters (MFI, AOC, DOC, TDS, Turbidity, pH, etc) are useful parameters for comparing relative clogging potentials of various waters, but that they cannot be used to predict clogging and declines in injection rates for actual recharge wells, which also depend on well construction and aquifer characteristics. Thus full-scale studies on recharge test wells are still necessary to determine feasibility and design and management criteria for operational recharge wells.

### 5.4.1 Clogging control

Perez-Paricio (1999) compiled some recommendations and classified water quality according the potential effect on clogging during ASR. Table 2 has been extracted from this work to highlight the importance of controlling TSS, turbidity and TOC during ASR operation:

**Table 2: Design guidelines to prevent ASR clogging**
Source: Clogging handbook, Perez-Paricio (1999)

| Clogging in ASR systems | Recharge water quality | Well redevelopment |
|---|---|---|
| Slight | TSS < 1 mg/L<br>Turbidity < 1 NTU<br>TOC < 5 mg/L | Frequent pumping<br>Surging/jetting: once a month |
| Notable | 1 mg/L < TSS < 10 mg/L<br>1 NTU < Turbidity < 10 NTU<br>5 mg/L < TOC < 15 mg/L | Pumping once a day<br>Surging/jetting: once a week |
| Dangerous | TSS > 10 mg/L<br>Turbidity > 10 NTU<br>TOC > 15 mg/L | Daily pumping<br>Adapted protocol |

### 5.4.2 Compilation of water quality recommendations for ASR injection water

Table 3 summarises the values of recommendations and quality standards established worldwide for ASR. Specific references are listed in ANNEX 1 of this report.

**Table 3: Water quality recommendations for ASR injection water**
Source: own elaboration based on literature review

| Parameter | Unit | Maximum value (Reference) |
|---|---|---|
| TSS (Total Suspended Solids) | mg/L | 2 (1), 30 (2) |
| MFI (Modified Fouling Index) | [s/l²] | 3-5(3), 2(4) |
| Turbidity | [NTU] | 1(5,6), 0,2(4) |
| DOC (Dissolved Organic Carbon) | mg/L | 2 (7,8) |
| TDS (Total Dissolved Solids) | mg/L | 150(9),100(10),500(2) |
| AOC (Assimilable Organic Carbon) | µg acetate-C/L | 10 (11) |
| TOC (Total Organic Carbon) | mg/L | 5(5), 10(12,13) |
| E. Coli | NMP/100ml | 10.000 (2) |
| Ammonium | mg/L | 0,5 (2) |

# 6 Characterisation of pre-potable water in the Llobregat ASR System

## 6.1 Sand Filtered water characterisation

Sand filtered water seems to be the best pre-potable water available to be injected in the ASR system. It covers main requirements established by Aigües the Barcelona, as the operator or the DWTP in SJD. Sand filtered water corresponds to the last stage of the conventional treatment, and was in fact the potable water for the Barcelona area during lot of time (1969 – 1977, see Figure 7). Thus, sand filtered water has been chosen as the best option to inject in the demonstrative phase of the project. This section analyses existing data of water quality (2011 – 2014) and compares it with international recommendations reported in literature. Parameters and compounds have been selected according to what has been previously reported in other ASR experiences and critical aspects identified by Aigües de Barcelona, ACA (Water Authority) and ASPC (Health Authority). Parameters are presented in 3 categories: Physical clogging related parameters, bioclogging related parameters and groundwater pollutants.

### 6.1.1 Physical clogging related parameters

<u>MFI (Modified Fouling Index)</u>

MFI values were taken from LIFE UFTEC project [2] from 2011 to 2103, and from DESSIN analysis in 2014. 60% of MFI values are below 10 s/L$^2$ that is value recommended by Olsthoorn in order to avoid physical clogging. West Basin Quality standards have a more stringent limit of MFI recommended value.

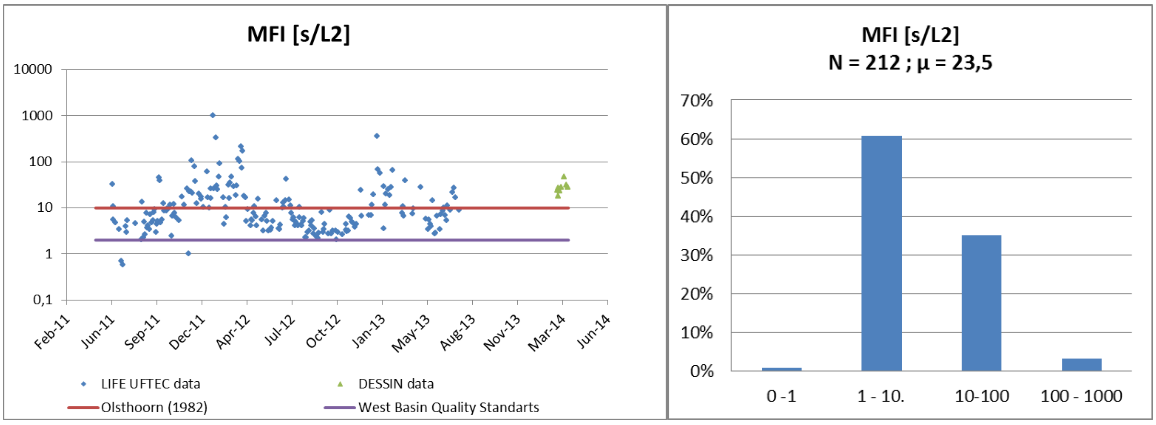



**Figure 14: MFI values (2011-2014) of SFW**

Left chart: punctual values, right chart: aggregated percentages of MFI ranges

[2] http://www.life-uftec.eu/

Turbidity

Figure 15 shows turbidity values from 2010 – 2014. More than 85% of turbidity values are between 0.1 – 0.4 NTU, while there are punctual peaks overpassing 1.0 NTU.

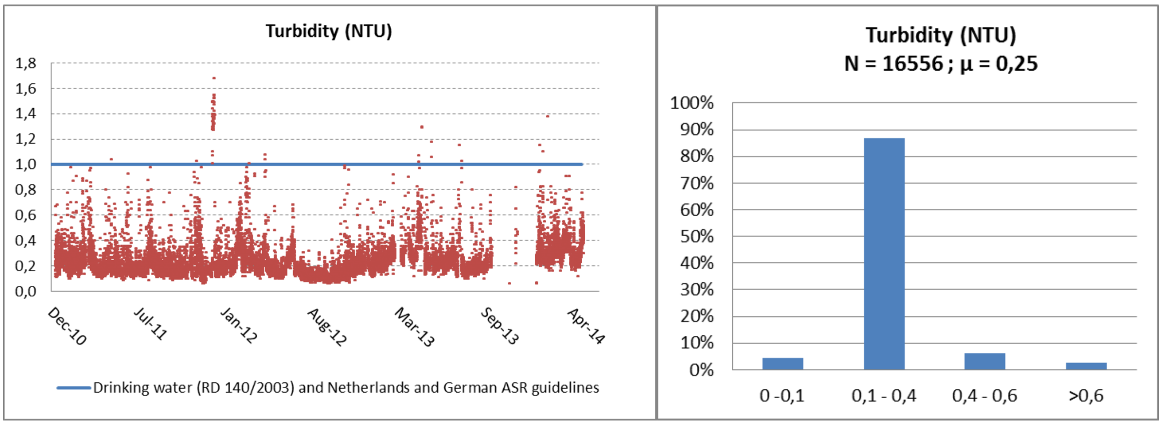



**Figure 15: Turbidity values (2010-2014) of SFW**
Left chart: punctual values, right chart: aggregated percentages of turbidity ranges

### 6.1.2 Bioclogging related parameters

AOC (Assimilable Organic Carbon)

As exposed before, AOC is determined microbiologically by plating out and incubating a water sample for growth of bacteria of the type Pseudomonas fluorescence. The analytical method is very time consuming and needs a well-prepared technicians so the analysis is very expensive and few laboratories in Europe are able to do it. So it was decided not to quantify AOC in SFW at this stage of the project. Instead of quantifying it, scientific literature has been reported: *Ribas et al.* (1997) determined the AOC levels in different parts of the SJD DWTP plant, determining 0.96 mgC/L (0.69 of coefficient of variance) in raw water and 0.31 mgC/L (1.74 of coefficient of variance) after prechlorination and sedimentation, previous to sand filter. Although the water quality of Llobregat River could not be the same and AOC data does not include sand filtration, that values give information of the range that we will have. On the other hand as AOC could be between 0.1 and 9% of TOC, and Toc is 3.6 mg/l in average, it could be estimated a possible AOC concentration between 0.0036 mg/L and 0.32 mg/L.

In column studies, Hijnen (1990) concluded that acetate concentrations in the water as low as 0.01 mgC/L. Estimations done in SJD overpass this limit, so the biological clogging will be specially assessed in the demonstration phase.

TOC

The average TOC in SFW is 3.6 mg/L. As seen in Table 2, TOC values below 5 mg/L should represent a slight clogging formation so from TOC point of view in DESSIN Llobregat site shouldn't be problems with bioclogging.

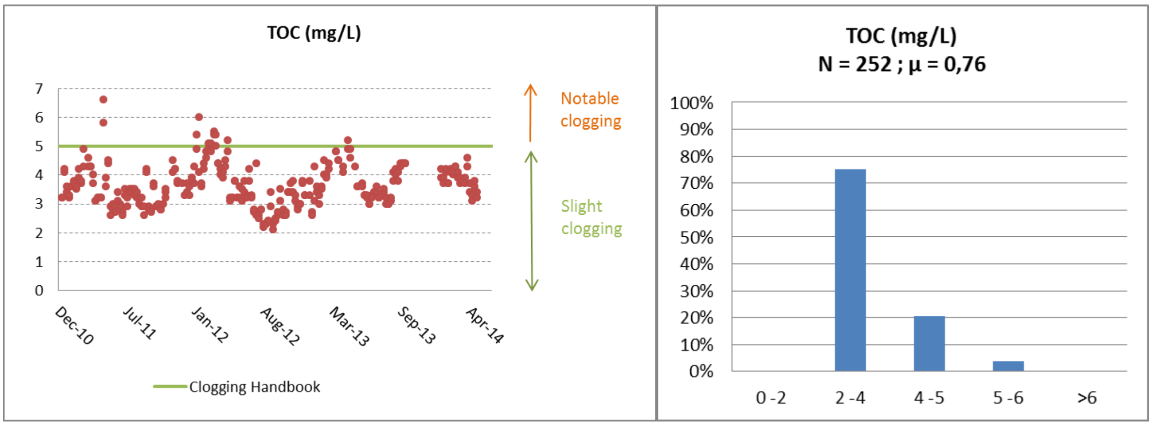



**Figure 16: TOC values (2010-2014) of SFW**

Left chart: punctual values, right chart: aggregated percentages of TOC ranges

<u>Microorganisms in injection water and groundwater</u>

Bioclogging formation could be caused from a lot of different microorganisms. Available information regarding microorganisms in SFW is the analysis of pathogens microorganisms used as control indicators of treatment processes. Pathogens will give an order of magnitude in terms of presence of microorganisms in injection water. Microorganisms analysed in the SFW are Total Coliforms, E. Coli, Clostridium Sulphite Reducing, Clostriduim Perfringens, Colony Count at 22ºC, Aeromonas, Pseudomonas, Cryptosporidium and Giardia. In the following charts are represented the data from December 2010 until April 2014.

According to the data of microbiology in the SFW (Figure 17 to Figure 23), it can be seen that the water as is not disinfected it has a microbial load that pass thought de sedimentation, prechloritartion and sand filters. In the case of Total Coliforms SFW have around 18% of samples without detection and the other 82% are mainly between 0 and 1000 MPN/100ml. for E.Coli have around 64% of samples without detection of E. Coli but the rest it have detection between 1 and 10.000 cfu's. In the Future directions for South Australia report, is it recommended a faecal coliform value below 10.000 cfu, so DESSIN Llobregat site fulfil this recommendation.

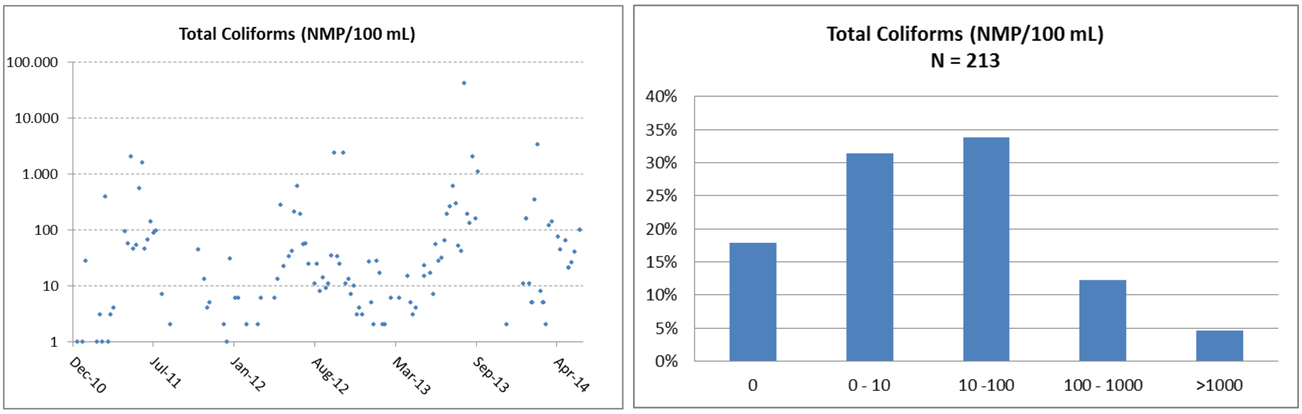



**Figure 17: Total Coliforms values (2010-2014) of SFW**

Left chart: punctual values, right chart: aggregated percentages of Total Coliforms ranges

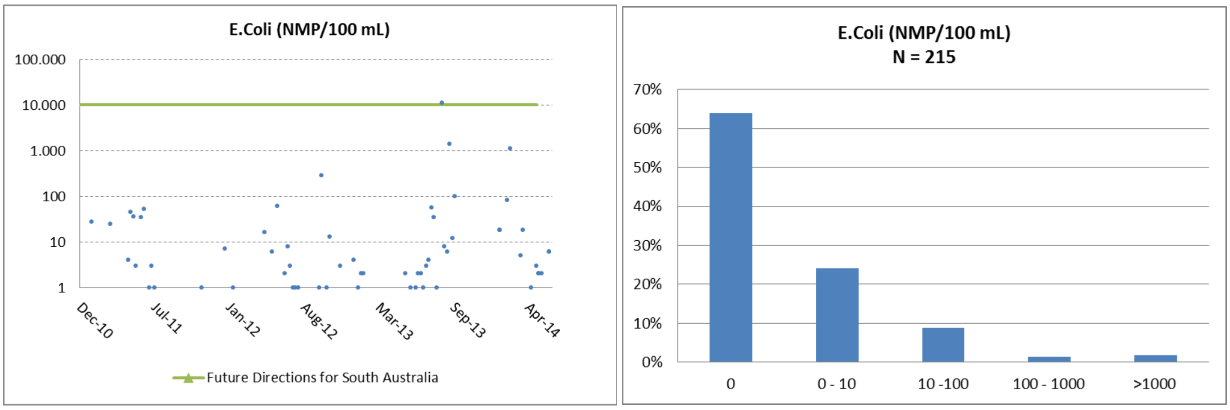



**Figure 18: E.Coli values (2010-2014) of SFW**

Left chart: punctual values, right chart: aggregated percentages of E.Coli ranges

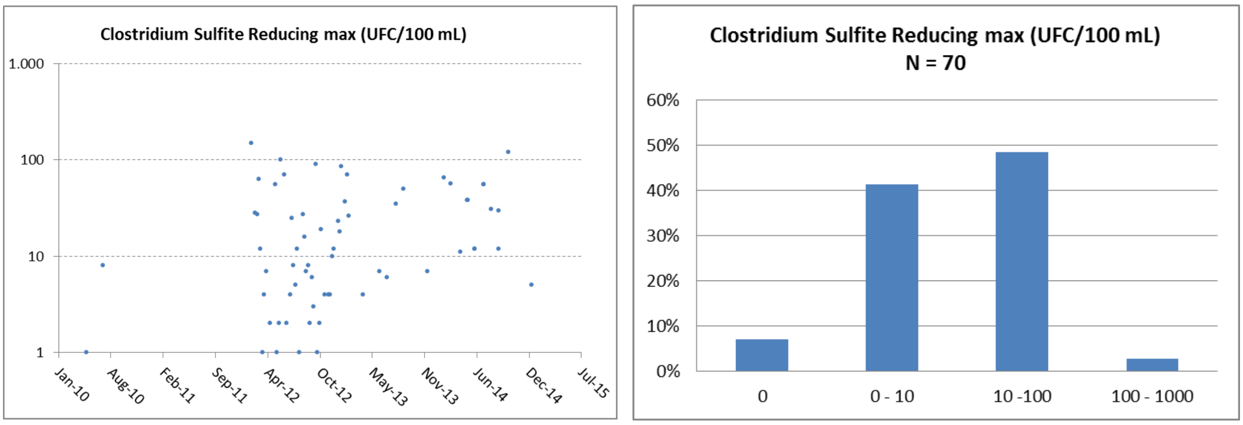



**Figure 19: Clostridium Sulphite Reducing maximum values (2010-2014) of SFW**

Left chart: punctual values, right chart: aggregated percentages of Clostridium Sulphite Reducing ranges

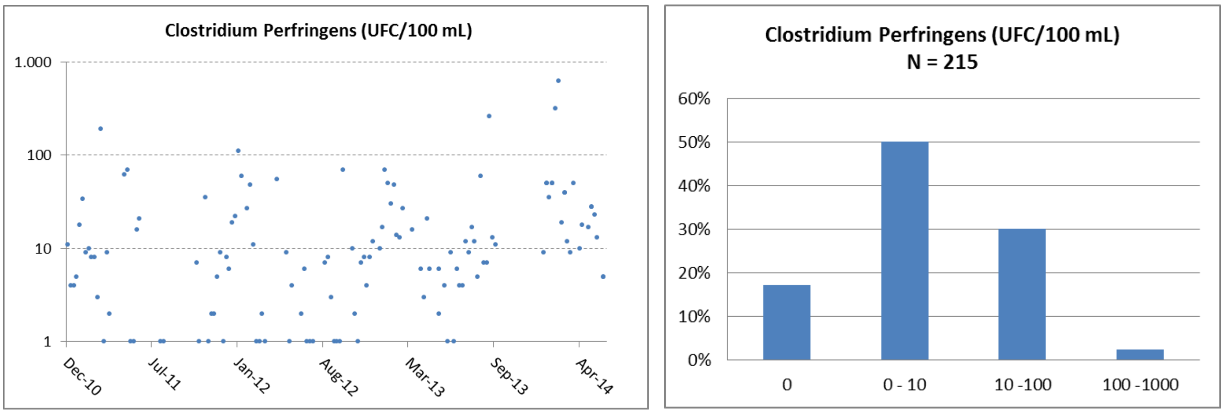



**Figure 20: Clostridium Perfringens values (2010-2014) of SFW**

Left chart: punctual values, right chart: aggregated percentages of Clostridium Perfringens ranges

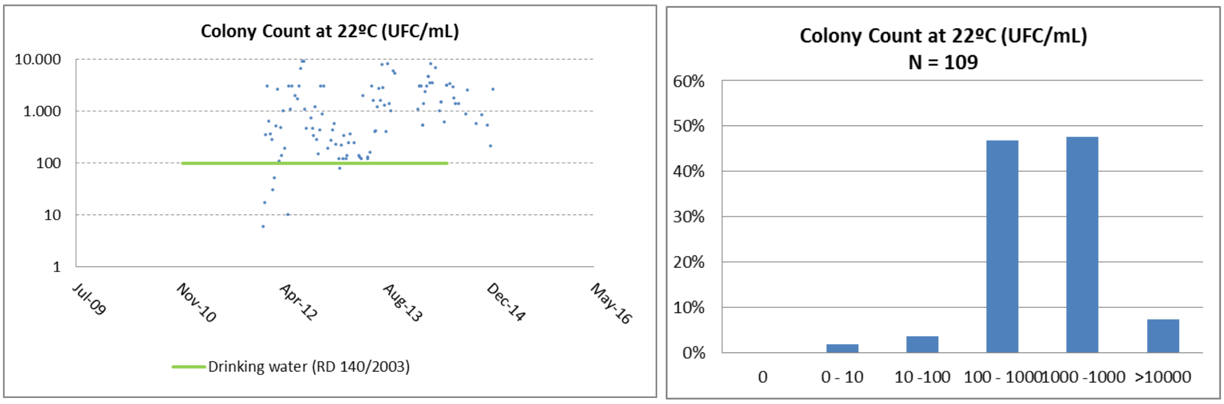



**Figure 21: Colony Count at 22ºC values (2011-2014) of SFW**

Left chart: punctual values, right chart: aggregated percentages of Colony Count at 22ºC ranges

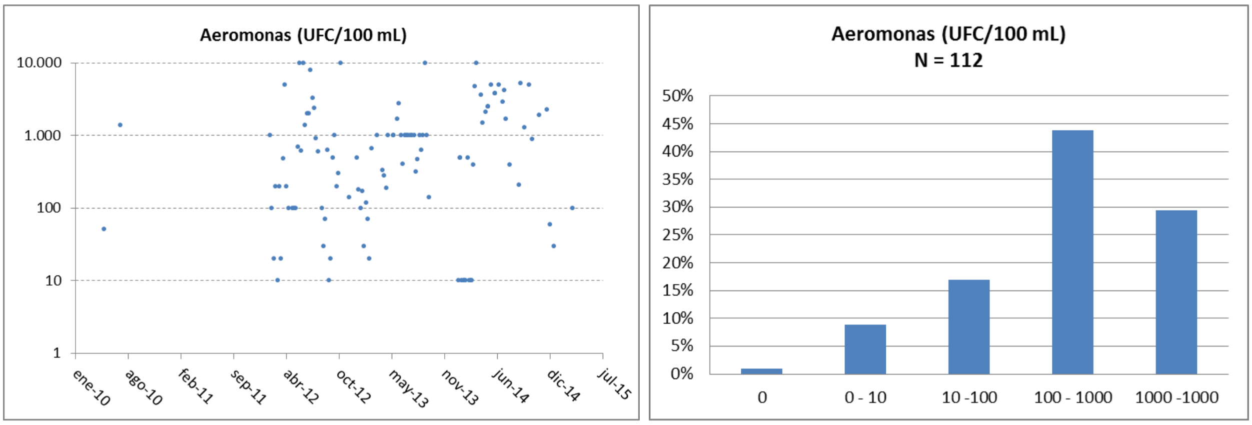



**Figure 22: Aeromonas values (2010-2014) of SFW**

Left chart: punctual values, right chart: aggregated percentages of Aeromonas ranges

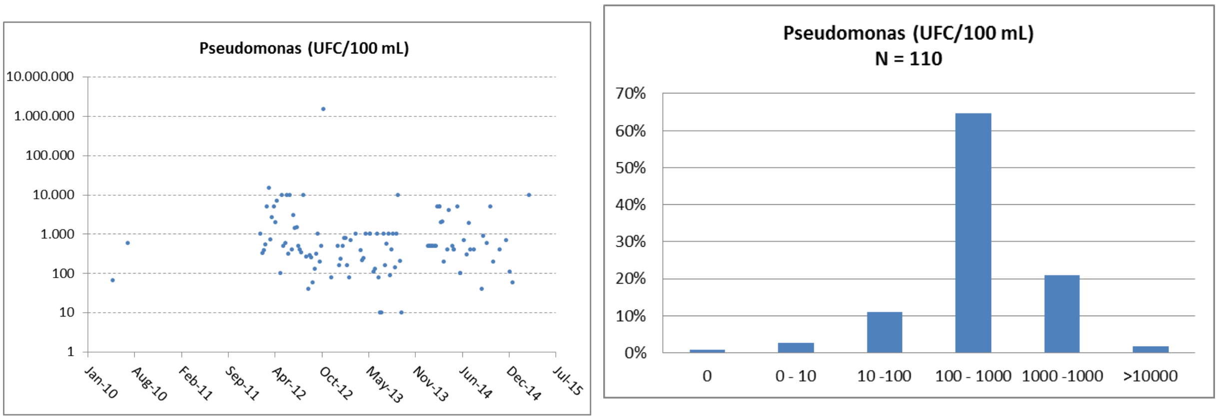



**Figure 23: Clostridium Perfringens values (2010-2014) of SFW**

Left chart: punctual values, right chart: aggregated percentages of MFI ranges

### 6.1.3 Groundwater pollutants

This section includes those compounds considered not desirable in groundwater. It's a misceslaneous chapter including chemical compounds that are considered toxic, or are specifically regulated in groundwater to get a good quality status in a groundwater body. Other, as metals, are also added in the section and the SFW values are compared to drinking water quality standards.

<u>THM (Trihalomethanes)</u>

Trihalomethanes are chemical compounds in which three of the four hydrogen atoms of methane (CH4) are replaced by halogen atoms and are formed as a by-product predominantly when chlorine is used to disinfect water for drinking. They result from the reaction of chlorine or bromine with organic matter present in the water being treated. The THMs are considered an environmental pollutants and have been associated through epidemiological studies with some adverse health effects, and many are considered carcinogenic.

As is not chlorinated, SFW does not have concentration of trihalomethanes, avoiding a contamination in the aquifer of this species.

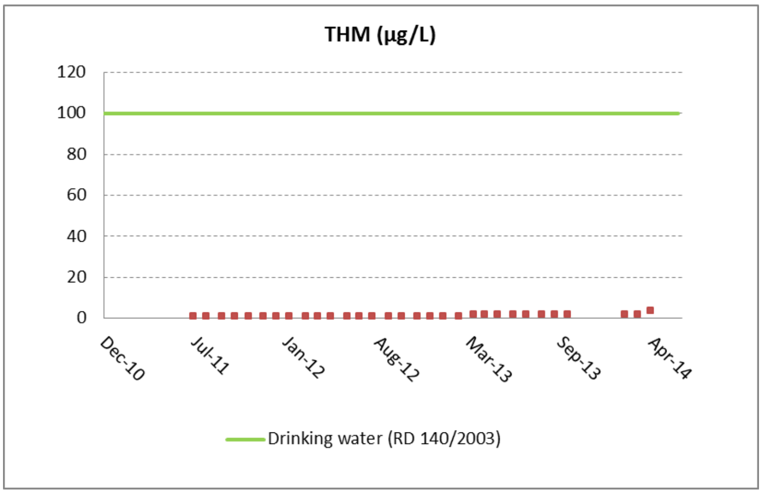
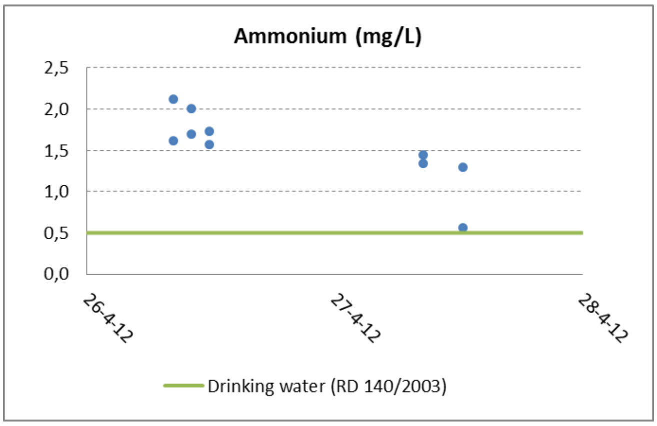



**Figure 24: Total Trihalomethanes values (2011-2014) of SFW**

<u>Nitrogen cycle: ammonium and nitrate</u>

As it was available few data regarding ammonium (Figure 25) nitrate and sulphate in SFW, it was taken into account historical data of Llobregat River, that actually would be always higher than SFW because its treatment. As it can be seen in (Figure 25), ammonium in river water is highly variable and seasonal, having peaks around 10 mg/L and with an average below 2 mg/L being above of drinking water standards. The injection of ammonium in the aquifer will be monitored during demonstration in order to detect how is behaving this compound as a part of nitrogen cycle.



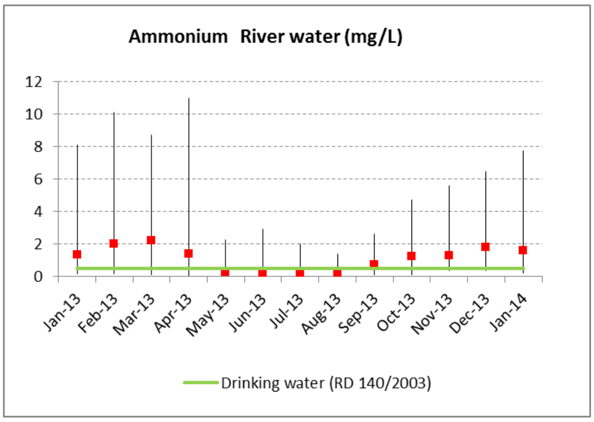




**Figure 25: Ammonium values (2012) of SFW and Ammonium values (2013) of Llobregat River water**

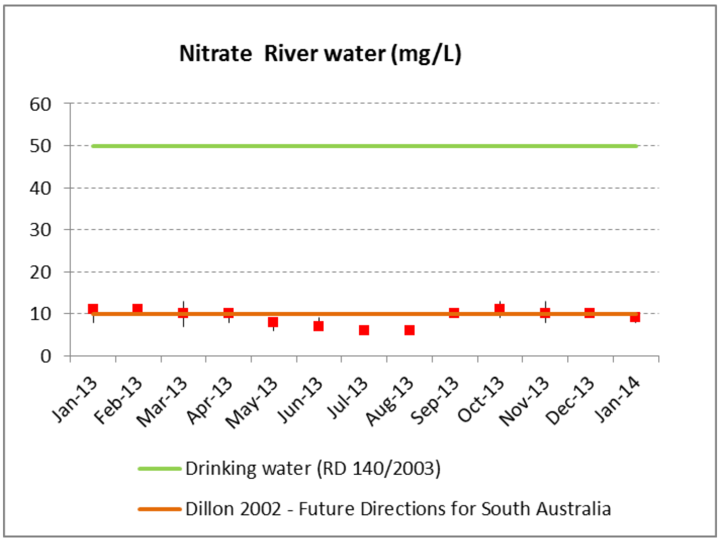



**Figure 26: Nitrate values (2013) of Llobregat River water**

River Water has nitrate concentrations around 10 mg/L, above drinking water standards and in the same level of the Future Directions of South Australia recommendations. Sulphate concentrations in Llobregat River are below drinking water standards. Aluminium salts are used as coagulant in sedimentation process of river water, so the concentration of Aluminium in SFW is higher than the concentration in river water. However 71 % of historical samples and the average concentration of 175 µg/L so are below drinking water standard of 200 µg/L. Nickel and iron values in SFW are below drinking water standards.

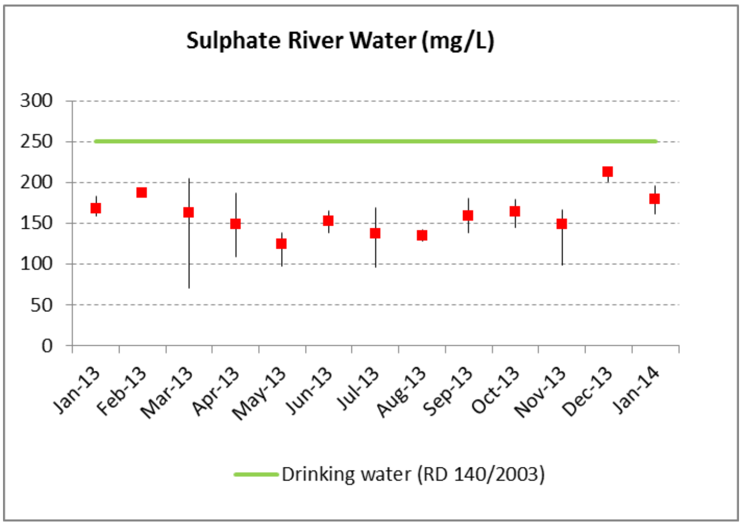
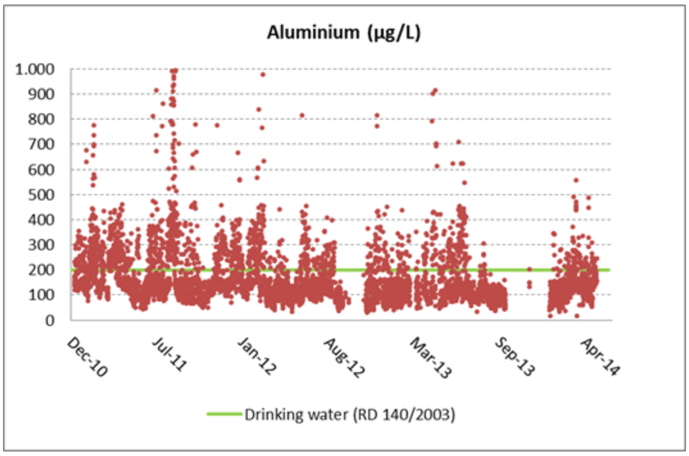



Figure 27: Sulphate values (2013) of Llobregat River water



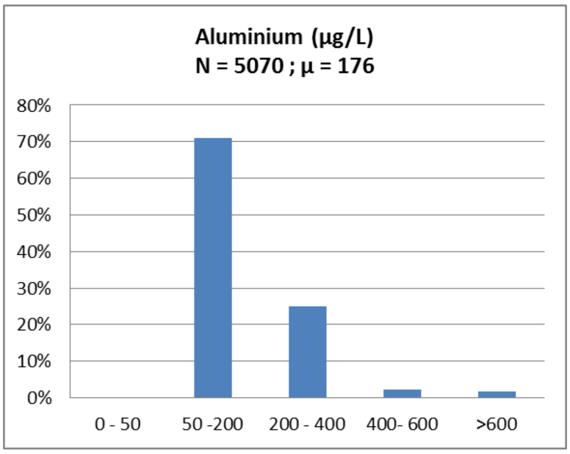




Figure 28: Aluminium values (2010-2014) of SFW

Left chart: punctual values, right chart: aggregated percentages of Aluminium ranges

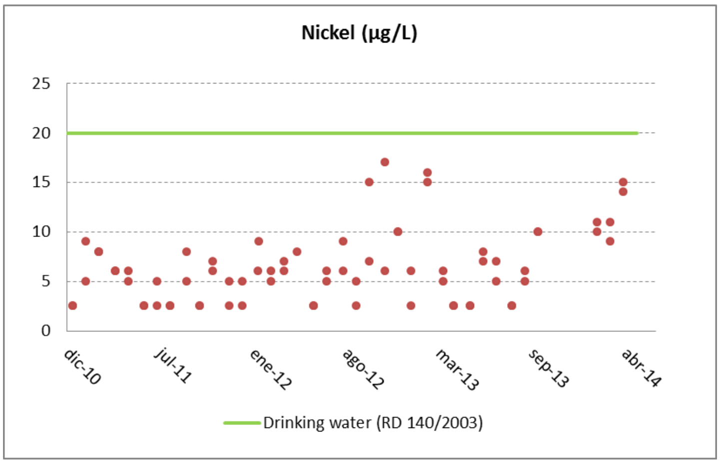



Figure 29: Nickel values (2010-2014) of SFW

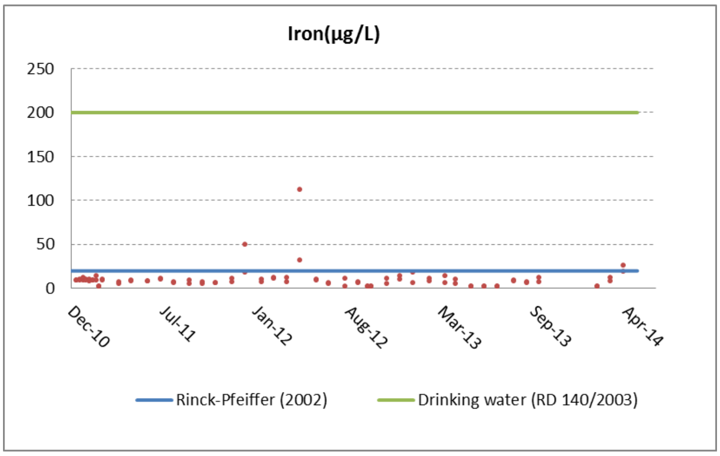



Figure 30: Iron values (2010-2014) of SFW

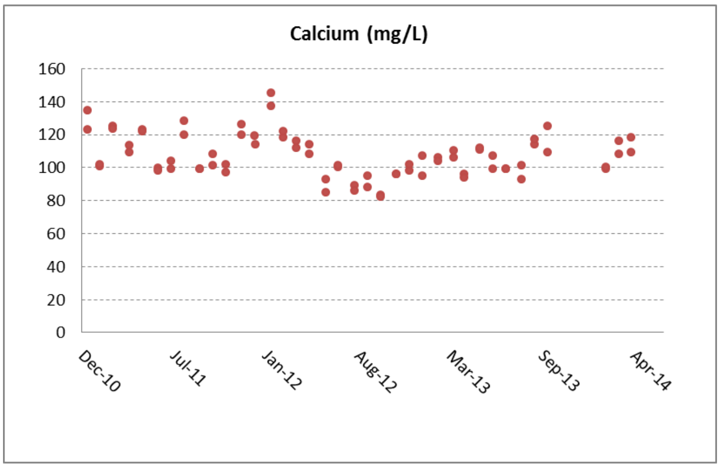



Figure 31: Calcium values (2010-2014) of SFW

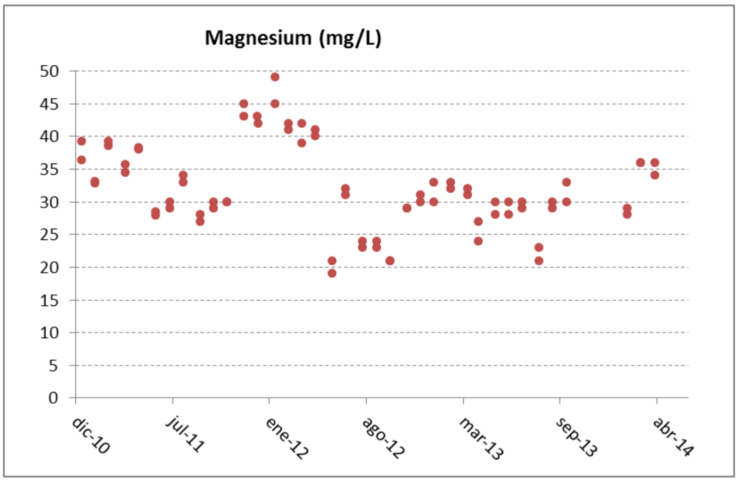



Figure 32: Magnesium values (2010-2014) of SFW

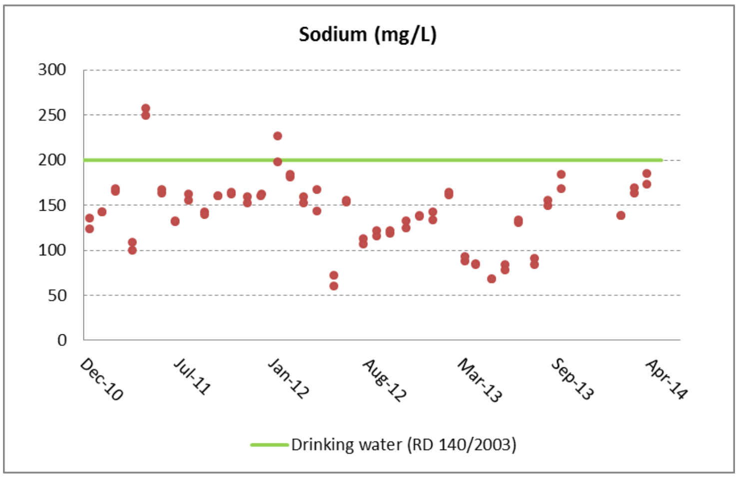



Figure 33: Sodium values (2010-2014) of SFW

## 6.2 Groundwater characterisation (native groundwater)

Figure 34 represents chemical water composition of 5 selected wells of the ASR system and the sand filtered water (SFW) using a Piper diagram. Content on $Na^{+}$ and $K^{+}$ is higher that $Mg^{2+}$ concentration, while anions concentration are more equilibrated. All the samples present a very similar pattern, which means that no significant differences between native groundwater and sand filtered water are found in ionic bulk chemistry.

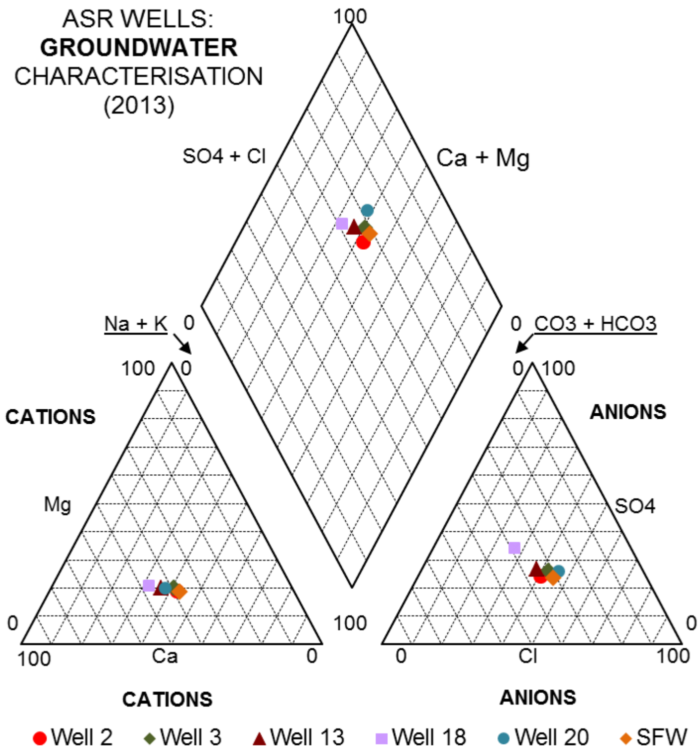



Figure 34: Piper diagram of groundwater from different wells and from SFW (2013)

**Table 4: Quality comparison between SFW and Groundwater of LLobregat Site.**
NOTE: Data are averages between samples taken from 2011 to 2014.
(*) As it was not available SFW data it was taken River data.

| Parameter | Sand Filtered Water | Groundwater |
|---|---|---|
| Turbidity [NTU] | 0.24 | 0.13 |
| Colour [mg Pt/L] | 4.6 | 1.4 |
| Electrical conductivity [µS/cm] | 1279 (*) | 1753 |
| Chloride [mg/L] | 229 | 297 |
| Total Organic Carbon (TOC) [mg/L] | 3.6 | 1.2 |
| Calcium [mg/L] | 105 | 164 |
| Magnesium [mg/L] | 27 | 48 |
| Sodium [mg/L] | 116 | 173 |
| Sulfate [mg/L] | 160 (*) | 244 |
| Nitrate [mg/L] | 9.1 (*) | 10.3 |
| Ammonium [mg/L] | 1.11 (*) | 0.07 |
| Fe total [µg/L] | 10,1 | 96.4 |
| Aluminium [µg/L] | 179 | 13 |
| Nickel [µg/L] | 6,9 | 2.5 |
| Total manganese [µg/L] | 9 | 12 |
| Total phosphorus [µg/L] | 32 | 10 |
| Total THM [µg/L] | <2 | <2 |
| *E.coli* [MPN/100mL] | 120 | 0 |
| Total coliform [MPN/100mL] | 731 | 0.5 |
| *Enterococcus* [CFU/100mL] | 2 | 0 |
| *Clostridium perfringens* [CFU/100mL] | 35.7 | 0.1 |
| Colony count at 22ºC [CFU/100mL] | 4024 | 31 |

In Table 4 it can be seen the comparison of quality parameters between groundwater and the sand filtered water (SFW). In general terms, the two types of water are in the same range in most of parameters. Ionic parameters as calcium, magnesium, sodium, sulphate, nitrate or chloride are similar between two types of water. Total trihalomethanes are in both cases below quantification limit of 2 µg/L.

Most relevant differences are:

- Turbidity of SFW is in the same range than groundwater being just slightly higher (0.24 NTU vs 0.13 NTU respectively) so it could be predicted no problems regarding physical clogging.
- Electrical conductivity is lower in the SFW so its injection will decrease the aquifer salinity.

- Total organic carbon is higher in SFW (3.6 mg/L vs 1.2 mg/L respectively) but not exceeding some bioclogging recommendations of 5 mg/L.
- Aluminium of SFW is higher than aluminium in groundwater. However, aluminium in SFW is still below drinking water standards (200 µg/L).
- Ammonium concentration in SFW is higher than the concentration in groundwater. During the demonstration phase ammonium will have to be analysed in accurately in the monitoring network to know how the behaviour within nitrogen cycle is.
- Microbiology As SFW is not disinfected, microbiological parameters (E.coli, Total coliforms, Clostridium perfringens and Colony Count at 22ºC) are 2 or 3 orders of magnitude higher than groundwater. While groundwater has almost not microbial load, the conventional treatment is not able to eliminate all the bacteria of the river so SFW has a significant load. One of the goals during demonstration phase in Llobregat site will be to determine the microbial inactivation within the aquifer and to evaluate if the microbial load could produce a problematic bioclogging.

## 6.3 Multicriteria analysis

In order to use a multicriteria analysis comparing the benefits of injecting sand filtered or potable water it has been used AquaStoRe, a Decision Support Tool capable of evaluating the suitability of potential ASR or ASTR sites developed by Cetaqua in the framework of an R+i Alliance project (internal founds of SUEZ ENVIRONNEMENT Group[3]). The objective of AquaStoRe is to assess the potential feasibility of an ASR/ASTR system for new implementations by evaluating the information introduced by the user and maintaining objectivity on the final result.

AquaStoRe structure, which was developed on a Microsoft Excel environment, drives the user through 3 stages that evaluate separately socioeconomic, hydrogeological and hydrochemical aspects of ASR systems. The evaluation methodology of those aspects is based on the assessment of 63 indicators, carried out through a multicriteria analysis (MC) that gives to each indicator a value weighted by a specific factor for every indicator.  The main feature of AquaStoRe is the possibility of comparing one or more different alternatives to implement an ASR or ASTR in a tested site.

Quality aspects enclose indicators that represent physical and chemical variables of the native groundwater in the aquifer and also of the recharge water, introducing a relative comparison between them. Within this group, there are some indicators evaluating the relation between recharge water and native groundwater quality which assess through comparison the blending process of recharge water with groundwater. For indicators, within each category, the best rank value is assigned according to the importance in the ASR system. In addition, this value is multiplied by a weight value, which assesses the information representation used in each indicator rank. Subsequently, the final result for each category is a value representing the suitability of the ASR location for that category.

The quality of the data introduced in AquaStoRe has been taken into consideration by assigning a reliability factor (RF), which evaluates its source and ranks it for a better classification. RF transfers a level of veracity to the results obtained through multicriteria analysis. Thus, a high value of RF is necessary to trust initial information and, moreover, permits comparison and differentiation between case studies.

[3] http://www.ri-alliance.com/site/RIA/homepage

The section of AquaStoRe's results mainly visualizes a summary of the individual score given to each indicator of the case studies to be compared. This information is generated by the tool and drawn in table format with its values on a colour scale that indicates the best or worst scored indicators. This colour scale classifies the values on the table, indicating whether they are considered positive or negative values by AquaStoRe, and also evidences the reliability and origin of the data available for each indicator. This functionality of the tool is very useful after the analysis of the main results showed on the overall evaluation, addressing the user to the source of the differences between case studies.

**Table 5: Possible results of multicriteria analysis and reliability factors**

| MC | | RF | |
|---|---|---|---|
| No data introduced | BLACK | No data introduced | BLACK |
| Low score indicator | RED | Low reliability | RED |
| Regular score | ORANGE | Average reliability | ORANGE |
| High score indicator | GREEN | High reliability | GREEN |

### 6.3.1 Sand filtered vs Drinking water multicriteria hydrochemical comparison

For the Llobregat DESSIN site it has been used the AquaStoRe tool to assess the suitability of SFW that will be injected during the project in front of drinking water that has been injected historically. Only module 3 consisting in hydrochemistry has been applied. After the introduction of all the data available in the tool regarding physical and chemical indicators of SFW and drinking water of Sant Joan Despí DWTP, Figure 35 shows the overall evaluation of the two types of water.

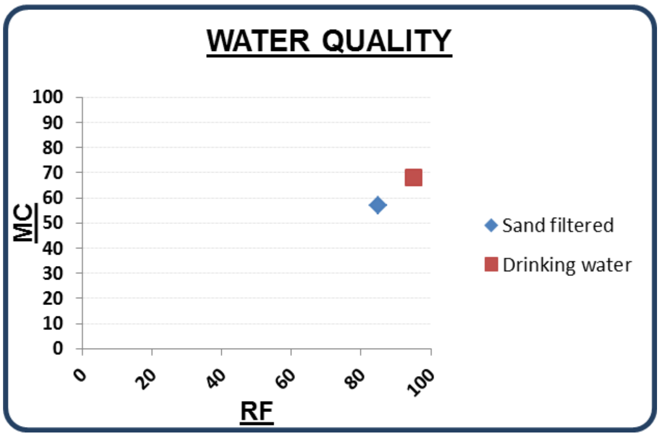



**Figure 35: Overall evaluation of sand filtered and drinking water of Llobregat ASR site**

The overall evaluation of the tool gives an idea of the positive or negative expectations regarding water quality and the reliability of the data sources. The maximum MC and RC are calculated out of a 100%, which indicate the maximum score a hypothetical site could have in case it has the ideal conditions. As it can be seen in Figure 35, the multicriteria analysis result gives values of 57% for SFW and 68% for drinking water, with a RF of 85% and 95% respectively. This indicates that both types of water are in the medium-high level of chemical quality for ARS applications, having the SFW an 11% lower score than drinking water.

Going in detail into the multicriteria analysis, Table 6 lists each single indicator representing physical and chemical variables. Each indicator has a different score for the two different types of water and represents the quality value multiplied by the weight given to each indicator.

AquaStoRe tool have in total 63 indicators that analyse site conditions for ASR implementation. From 1 to 25 these indicators are related with ASR management and site hydrogeology and from 26 to 63 are related with water quality. For DESSIN Llobregat site the objective was to compare two types of water to be injected in the same aquifer, so it only has been used the water quality part of the tool.  This group of indicators approaches the chemical parameters that could affect an ASR system.  Indicators considered for AquaStoRe tool are:

- Indicators from 26 to 29, both included, represent the physical variables of the recharge water.
- Indicators from 30 to 36, both included, gather the chemical variables of the recharge water.
- Indicators from 37 to 39, both included, characterize microbiologically the recharge water.
- Indicators from 40 to 63, both included, correlate aquifer native groundwater with the recharge water. Being from 40 to 50 non toxic substances, and toxic substances from 51 to 63.

**Table 6: Multicriteria indicators of the quality comparison between SFW and potable water**

Source: AquaStoRe tool

| Indicador | SFW | | Potable water | |
|---|---|---|---|---|
| | MC | RF | MC | RF |
| 26. Temperature of recharge water | 30 | 10 | 30 | 10 |
| 27. Arsenic mobilizing capability | 5 | 0.5 | 5 | 10 |
| 28. Total Suspended Solids | 12.5 | 0.5 | 12.5 | 10 |
| 29. Turbidity | 50 | 10 | 50 | 10 |
| 30. Dissolved oxygen | 0 | 10 | 0 | 10 |
| 31. Residual chlorine | 30 | 10 | 30 | 10 |
| 32. Ryznar Index | 7.5 | 0.5 | 7.5 | 0.5 |
| 33. Trihalomethanes | 40 | 10 | 40 | 10 |
| 34. Pesticides | 40 | 10 | 40 | 10 |
| 35. Priority organic micropollutants | 20 | 10 | 20 | 10 |
| 36. Non-regulated emergent pollutants | 10 | 10 | 10 | 10 |
| 37. Enterococcus | 0 | 10 | 20 | 10 |
| 38. Total coliforms | 0 | 10 | 40 | 10 |
| 39. Escherichia coli | 0 | 10 | 40 | 10 |
| 40. pH | 50 | 10 | 50 | 10 |
| 41. Calcium | 6 | 10 | 4 | 10 |
| 42. Magnesium | 6 | 10 | 4 | 10 |

| Indicador | SFW | | Potable water | |
|---|---|---|---|---|
| | MC | RF | MC | RF |
| 43. Sodium | 18 | 10 | 12 | 10 |
| 44. Iron | 0 | 10 | 0 | 10 |
| 45. Aluminium | 0 | 10 | 30 | 10 |
| 46. Beryllium | 10 | 10 | 10 | 10 |
| 47. Manganese | 40 | 10 | 8 | 10 |
| 48. Phosphate | 4 | 10 | 10 | 10 |
| 49. Dissolved Organic Carbon | 10 | 10 | 20 | 10 |
| 50. Electrical conductivity | 40 | 10 | 40 | 10 |
| 51. Fluoride | 2.5 | 10 | 2.5 | 10 |
| 52. Bromide | 2.5 | 0.5 | 10 | 10 |
| 53. Cadmium | 10 | 10 | 10 | 10 |
| 54. Total chromium | 10 | 10 | 10 | 10 |
| 55. Copper | 10 | 10 | 10 | 10 |
| 56. Lead | 10 | 10 | 10 | 10 |
| 57. Nickel | 2 | 10 | 10 | 10 |
| 58. Zinc | 10 | 10 | 10 | 10 |
| 59. Chloride | 30 | 10 | 30 | 10 |
| 60. Sulfate | 30 | 10 | 30 | 10 |
| 61. Arsenic | 10 | 10 | 10 | 10 |
| 62. Nitrate | 40 | 10 | 40 | 10 |
| 63. Ammonium | 0 | 10 | 25 | 10 |

Regarding physical variables (indicators 26-29) of the recharge water, both waters have a high score meaning that have a good conditions of temperature and turbidity.

Regarding chemical variables (indicators 30-36) all have a high score except dissolved oxygen. There is no trihalomethanes, pesticides, priority organic micropollutants and non-regulated emergent pollutants detected in none of the two types of water, so this results in a high score in these parameters. Storage of treated recharge water, which typically has higher dissolved oxygen levels and higher oxidation-reduction potential than the groundwater in confined aquifers can result in pyrite oxidation.  So if dissolved oxygen in the waters is above 8 mg/L, the tool considers this parameter with low score.

Regarding microbiology (indicators 37-39), SFW has a low scores because it is not disinfected water and have a positive analysis of enterococcus, total coliforms and Escherichia Coli. Drinking water has a high score because is a disinfected water and microbiology analysis are always negative.

Most of the non-toxic substances (indicators 40-50) correlation between aquifer native groundwater and recharge water is scored by the Aquifer Indirect Influence method. The point of this methodology aims to allocate the best score to the most similar concentrations to the host aquifer's groundwater, independently if they are a slightly above or below. The worst score is given to the most distant concentrations.

- pH, beryllium and manganese has same values in injecting waters than groundwaters, so all of them have a high score.
- Calcium, magnesium and sodium of both injection waters have a regular score considering that the concentration of that species are slightly different than groundwater.
- Dissolved organic carbon (DOC) has a low score for sand filtered because it has 195% more DOC than groundwater and drinking water has a regular score because it only have 27% more DOC than groundwater.
- Drinking water have a high score for aluminium but for SFW there is a concentration of 145 µg/L of aluminium coming from the coagulant of the sedimentation phase that produce a low score in this indicator.
- Although SFW has a low concentration of 32 µg/L it has a relative big difference with groundwater concentration, so it has a low score.
- Drinking water and SFW have a lower concentration of iron than groundwater, so both have a low score in iron indicator.

For conductivity is applied the Aquifer Direct Influence method, that is applied to the components that directly influence the groundwater quality, e.g. they can generate and/or increase the pollution of the aquifer. For this kind of components the ideal solution would be to inject water of the same or better quality than the native water in the aquifer. The higher the conductivity of the water recharge the greater the impact on groundwater. So based on the premise that water cannot be recharged with poorer quality than in the aquifer, all those values whose difference is positive are evaluated with the lowest mark. In our case as injected waters have a lower conductivity than groundwater; there is a high score in this indicator.

Correlation of toxic substances (indicators 51-63) between aquifer native groundwater and recharge water is also scored by the Aquifer Direct Influence method.

- Cadmium, Chromium, Copper, Lead, Zinc, Chloride, Sulfate, Arsenic and Nitrate have high score given that they are in the same concentrations or lower in the injected waters respecting the groundwater.
- SFW has a 79 % bigger concentration in Nickel than groundwater (7.1 µg/L vs 4.0 µg/L) so it has a low score. Ammonium in the SFW has an average value of 1.11 mg/L that is relatively bigger than the concentration in the groundwater of 0.07 mg/L, so the score is lowest one.

# 7 International experiences of ASR with non-potable water

In order to define the water quality of ASR sites around the world and compare it with Llobregat DESSIN site, it has been done a compilation of as much as possible existing data from literature and described case studies. The result is the following tables (Table 8):

Table 7a: Description of site and operational parameters of international ASR experiences (Australia)

| Site | | Aquifer | | | Infiltration water quality | | | | | | | | | Clogging issues | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Country | Site | Number injection wells | Aquifer type | Transmiss. (m2/day) | Water origin | TSS (mg/l) | TDS (mg/l) | AOC (µgC/L) | MFI (s/l2) | Turb (NTU) | TOC (mg/l) | DOC (mg/l) | Fecal coliform (cfu/100 ml) | Clogging | Comments clogging | Well redevelopment | Water treatment | Disinfection | Q (l/s) per well | Ref. |
| Spain | DESSIN site (Llobregat) | 12 | Sand and gravels. Quaternary alluvial | 30000 | River water (llobregat) | | | 310,0 | 23,5 | 0,3 | 3,6 | 3,6 | 731 | None | None | periodical backwashing | Coagulation, floculation, Sand Filtration | Cl2 | 50 | |
| Australia | Adelaide, South Australia | 50 | | | Reclaimed | 20,0 | | | | | | | 100 | | | Occasional pumping | coagulation, air, filtration | UV | 6 | [2] |
| Australia | South Australia | | Tertiary | 800 | River | 17,0 | 19250 | | | | | | | Low | | Pumping and Injection with HCl (5000 L at 15%) | | | 40 | [2] |
| Australia | South Australia | 14 | Carbonaceous fossils | 1000 | | 45,0 | 750 | | | | | | | Yes | | Pumping, Recharge only if TDS < 1000 mg/L | Sedimentation, removal of fines | | 30 | [4] |
| Australia | South Australia | 1 | Sands with carbonaceous fine cementation | 180 | Stormwater | | 190 | | | | | 4,3 | 293 | Yes | Caused by Suspended Solids; Microorganisms zooplancton | Periodical airlift, Long term: feasible dissolving carbonaceous cementation | | | 20 | [10] |
| Australia | South Australia, Andrews Farm | | Sands with carbonaceous fine cementation | 180 | Stormwater | 99,0 | | | 1500,0 | 50 | 8,0 | 4,3 | 620 | Yes | | Periodical airlift | Detention basins | | 11 | [11] [17] |
| Australia | South Australia, Bolivar | | Sandy limestone | 150 | Reclaimed | 6,0 | | | | | | | | Moderate | Biological + physical | | secondary plus advanced treatment plus chlorination | Cl2 | | [17] |
| Australia | South Australia, Willunga | | Sandy limestone | 80 | | 1,0 | | | | | 3,5 | | | Low | Physical | | secondary plus advanced treatment plus chlorination | Cl2 | | [17] |
| Australia | Carrum, Victoria | | Intercalated basalt | 570 | | | | | | | | | | Severe | Biological + physical | | secondary treatment plus chlorination | Cl2 1,5 Clresidual | | [17] |
| Australia | Urrbrae, South Australia | | Unconsolidated sand | 6 | Stormwater | | | | 240,0 | 27,9 | 8,5 | | | Severe | Biological + physical | | Wetland & rapid sand filtration | | | [17][18] |
| Australia | Jandakot, Leederville aquifer | | Sandstone beds | 790 | Surface | 5,0 | | | 70,0 | 0,8 | | 2,0 | | Low | Physical. (From 350 m3/d/m to 50 m3/d/m) | | Filtration (50 µm) | | | [19] |
| Australia | Mirraboka, Leederville aquifer | | Sandstone beds | 650 | Groundwater | 5,0 | | | 5,0 | 0,3 | | 2,0 | | Low | Removilization and compaction of aquifer fines. (From 550 m3/d/m to 200 m3/d/m) | | Filtration (50 µm) | | | [19] |
| Australia | Beenyup, Leederville aquifer, | | Sandstone beds | 2100 | Reclaimed | 1,1 | | | | 0,5 | | 1,0 | | No | | | UF, RO, UV | UV | | [19] |

## Table 8b: Description of site and operational parameters of international ASR experiences (Europe)

| Site | | Aquifer | | | Infiltration water quality | | | | | | | | | Clogging issues | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Country | Site | Number injection wells | Aquifer type | Transmiss. (m2/day) | Water origin | TSS (mg/l) | TDS (mg/l) | AOC (µgC/L) | MFI (s/l2) | Turb (NTU) | TOC (mg/l) | DOC (mg/l) | Fecal coliform (cfu/100 ml) | Clogging | Comments clogging | Well redevelopment | Water treatment | Disinfection | Q (l/s) per well | Ref. |
| Spain | DESSIN site (Llobregat) | 12 | Sand and gravels. Quaternary alluvial | 30000 | River water (llobregat) | | | 310,0 | 23,5 | 0,3 | 3,6 | 3,6 | 731 | None | None | periodical backwashing | Coagulation, floculation, Sand Filtration | Cl2 | 50 | |
| Israel | | | Sand, limestones, basalts | | Lake water mixed with groundwater | 1,0 | 640 | | | | | | 2 | Yes | Bacteria (important clogging) | Pumping | | Cl2 | 150 | [14] |
| Spain | Vergel | | Alluvial | 4100 | River | 4,0 | | | | | | | | Low | | | | | 40 | [9] |
| Switzerland | | | Sands and gravels | | River | | | | | | | 2,0 | | Yes | 60% capacity reduction in 2.5 years caused by bacteria and rganics | | Sedimentation + filters (sand and gravel) | | 10 | [5] |
| The Netherlands | St. Jansklooster | 2 (ASTR) | Coarse and fine sand | 3870 | Drinking water from local groundwater | <1 | 435 | | | | | 5.6-7.2 | | Minor (0.5 m/year) | Infiltration well hardly clogged. Recovery well could clog within 100 years by mixing of O2+NO3 with Fe(II) | Not needed | aeration and RSF | Not needed | 8.3-16.6 | [1] |
| The Netherlands | Someren (DIZON) | 2 (ASTR) | Tertiary sands | 510 | Channel water (>50% Meuse River) | <1 | 407 | 6.7-9.5 | 1.8-7.3 | | <1 | 0,8 | | Yes | Steady increase in resistance in injection well (2.5 m/2.5 years) | High rate backpumping, 6 times in 3 years | microstraining, flocculation, DAF, RSF, active carbon, SSF (SSF removed after | | 10 | [3] [15] [20] [21] |
| The Netherlands | Langerak | 2 (ASTR) | Pleistocene sands | 975 | Drinking water from local groundwater | <1 | 481 | 5,0 | 4,0 | 0,3 | <3.5 | 3,0 | | Yes | By iron flocs generated during aeration (pre-treatment) of anoxic groundwater, after 10 months | | aeration and RSF | Not needed | 10 | [16] [26] [27] |
| The Netherlands | Nieuwegein | 2 (ATR) | Pleistocene sands | 2100 | pretreated Rhine River | <1 | 528 | <10 | 3,0 | 0,1 | <2.5 | 2,2 | | No | | | coagulation, sedimentation, RSF and active carbon filtration | | 11 | [16] [26] [27] |
| The Netherlands | Waalsdorp | 46 (ATR) | Pleistocene sands | 1050 | Pretreated Meuse River | < 0,2 | 430 | <10 | <3 | | <2.5 | 2,3 | | Hardly | First years: 6 cm/m3/h,Later: 40 cm/m3/h. Likely cause = iron flocs from coagulation | Periodical infiltration stops (in winter), few times juttering | Coagulation, microstraining, RSF, Flotation, coagulation, RSF and active carbon filtration | | 5 | [28] [30] [31] |
| The Netherlands | Watervlak | 32 (ATR) | Pleistocene sands | 1350 | Pretreated Rhine River or Lake IJssel | <1 | 452 | 25,0 | 5-10 | | <3.5 | 3,2 | | ? | 40 cm/m3/h | Daily, short, high rate backpumping. After 16 years CBL and H2O2 were applied | Coagulation, RSF, active carbon filtration, UV+H2O2 | UV+H2O2 | 7 | [29] [30] [31] |
| The Netherlands | Herten | 1 (ASR) | Tertiary sands | 332 | Drinking water from local groundwater | 0,3 | 380 | | | | 0,7 | | 0 | No | | | Aeration and rapid sand filtration | Not needed | 13 | [20] [21] [22] |

Table 9c: Description of site and operational parameters of international ASR experiences (USA)

| Site | | Aquifer | | | Infiltration water quality | | | | | | | | | Clogging issues | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Country | Site | Number injection wells | Aquifer type | Transmiss. (m2/day) | Water origin | TSS (mg/l) | TDS (mg/l) | AOC (µgC/L) | MFI (s/l2) | Turb (NTU) | TOC (mg/l) | DOC (mg/l) | Fecal coliform (cfu/100 ml) | Clogging | Comments clogging | Well redevelopment | Water treatment | Disinfection | Q (l/s) per well | Ref. |
| Spain | DESSIN site (Llobregat) | 12 | Sand and gravels. Quaternary alluvial | 30000 | River water (llobregat) | | | 310,0 | 23,5 | 0,3 | 3,6 | 3,6 | 731 | None | None | periodical backwashing | Coagulation, floculation, Sand Filtration | Cl2 | 50 | |
| USA | Tucson, Arizona | 10 | | | Potable | 0,5 | | | 10,0 | | | | | Yes | Due to Suspended Solids | Redevelopment frequency depends on estimated clogging rates | | | 25 | [6] |
| USA | Texas | 1 | | | Reclaimed | | 680 | | | 0,5 | | | | Yes | | Pumping and surging | Sedimentation filtration., Ozonation, AC | Ozone | 50 | [7] |
| USA | Phoenix | 1 | Alluvial | 3100 | Surface | 4,2 | | | | 2,3 | | | | No | | | Micro-screen | Chlorination (7 ppm Cl2 at the beginning; 1.5 ppm later on) | 19 | [8] |
| USA | Colorado | | Partially cemen. Sandstone | 100 | Surface | 0,2 | | | | | | | | Yes | 32,3 m /month norm. Rate | Backflushing | Chem. addition floccul., filtr., Cl2 | Cl2 | 16 | [12] [13] |
| USA | Las Vegas | | Sands and gravels | 2800 | Surface | 0,1 | | | 16,1 | | | | | Yes | 0,8 m /month norm. Rate | Backflushing | Chem. addition floccul., filtr., Cl3 | Cl2 | 92 | [12] [13] |
| USA | Calleguas | | Sands and gravels | 1100 | Surface | 0,4 | | | | | | | | Yes | 11,3 m /month norm. Rate | Backflushing | Chem. addition floccul., filtr., Cl4 | Cl2 | 38 | [12] [13] |
| USA | Pasadena | | Sands and gravels | 2800 | Surface | 0,0 | | | | | | | | Yes | 0,6 m /month norm. Rate | Backflushing | Chem. addition floccul., filtr., Cl5 | Cl2 | 98 | [12] [13] |
| USA | Seattle | | Sands and gravels | 4200 | River | 1750 | | | 79,2 | | | | | Yes | 0,03 m /month norm. Rate | Backflushing | Cl2 + lime (to avoid corrosion) | Cl2 | 44 | [12] [13] |
| USA | Sonoma | | Sands and gravels | 230 | River | 0,1 | | | | | | | | No | | Backflushing | Chem. addition floccul., filtr., Cl2 | | 37 | [12] [13] |
| USA | Tucson, Arizona | | Basin filled with interbedded | 741 | Groundwater | 0,7 | | | 0,1 | | | | | Yes | 9 m /month norm. Rate | Backflushing | Cl2 | Cl2 | 77 | [12] [13] |
| USA | Orange Country, California | | Sands and gravels | 5145 | Reclaimed | | | | | 0,7 | 2,1 | | | Low | Biological + physical | | Tertiary | | | [17] |
| USA | Palo Alto Baylands, California | | Sands and gravels | 110 | | | | | | | 3,0 | 10,0 | | Low | | | secondary plus advanced treatment plus chlorination | Cl2 | | [17] |
| USA | Texas, El Paso | | Sands and gravels | 1240 | | | | | | | 0,5 | | | Moderate | | | Tertiary including disinfection (to near-potable standards) | | | [17] |
| USA | Sun Lakes, Arizona | | Sands and gravels | 26000 | | 5,0 | | | | | | | | Moderate | Biological | | Tertiary including disinfection (to near-potable standards) | | | [17] |
| USA | Waimanalo, Hawai | | Fine-medium sand | 90 | | 15,0 | | | | | 23,5 | | | Moderate | Biological (long-term) + physical (short-term) | | secondary treatment plus chlorination | Cl2 (1,5 (chlor res)) | | [17] |

From all data collected from the different ASR sites it has been done a comparison with DESSIN Llobregat site between single parameters as transmissivity, MFI, Turbidity, TOC, AOC or faecal coliform. It has to be noted that not all the sites listed in Table 8 have information regarding all single parameters. This information compiled has been really valuable to compare the status of DESSIN Llobregat site with the injection of SFW with other ARS sites and their injected water quality.

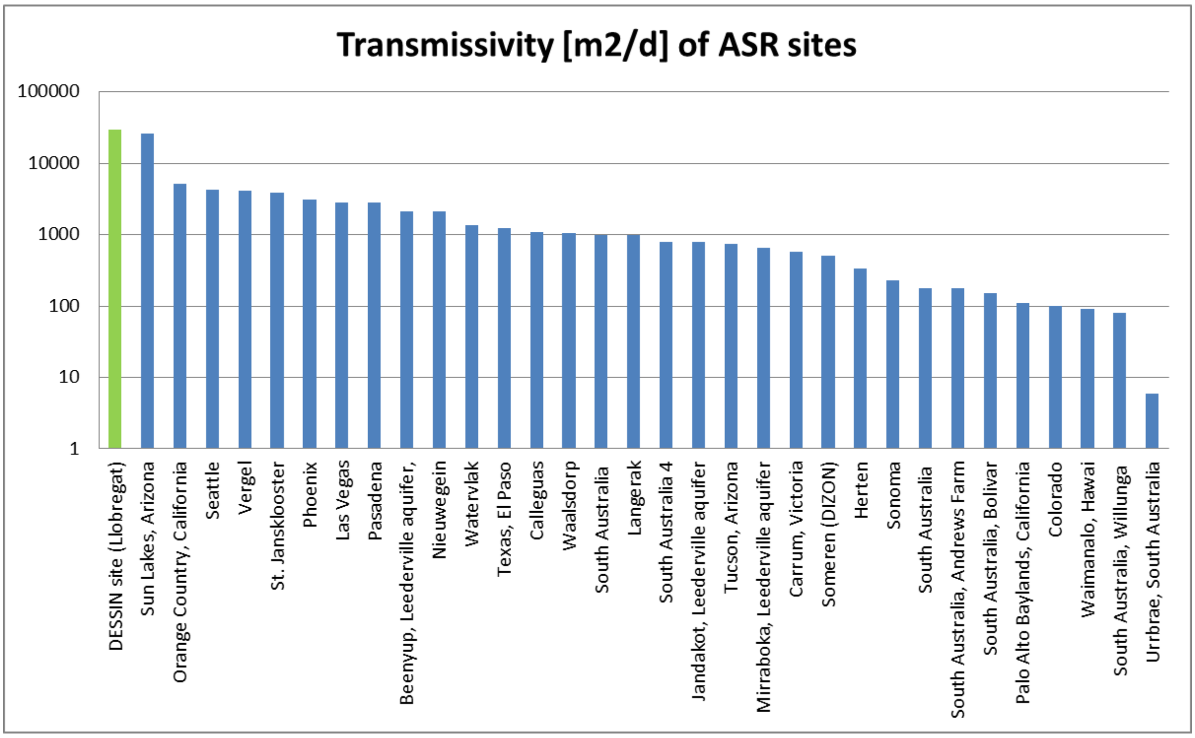



**Figure 36: Transmissivity of aquifers where ASR is applied**

Transmissivity of ASR sites range from 6 $m^2$/day of Urrbrae in South Australia to the DESSIN site that has the bigger transmissivity with 30,000 $m^2$/day, but the majority have it from 100 to 5000 $m^2$/day. DESSIN Llobregat site is placed in the Llobregat delta aquifer that is known to be very transmissive, implying that the injection wells should accept large flows and that there is less probabilities of having problems with physical clogging.

MFI values (Figure 37) ranges from less than 0.1 s/$L^2$ in Tucson to 1,500 s/$L^2$ in Andrews Farm. Olsthoorn (1982) stated that injection water should have MFI values less than 10-15 s/$L^2$. SFW from DESSIN site has 23.5 s/$L^2$ of average MFI, so it is the upper range of sites list and above recommendations. MFI is indicator of potential clogging, thus it could appear in Llobregat demo site. In the other hand, as the aquifer is really transmissive, it could minimize the physical clogging.

Turbidity range from 0.1 to 50 NTU, and DESSIN Llobregat site has lower turbidity than the majority with 0.25 NTU value (see details in Figure 38). Total Organic Carbon (Figure 39) ranges from 0.5 to 23.5 mg/L, and DESSN Llobregat site is the medium of the list. It is recommended to have TOC value less than 10 mg/L to avoid bioclogging, so except Hawaii site all the others are below this value.

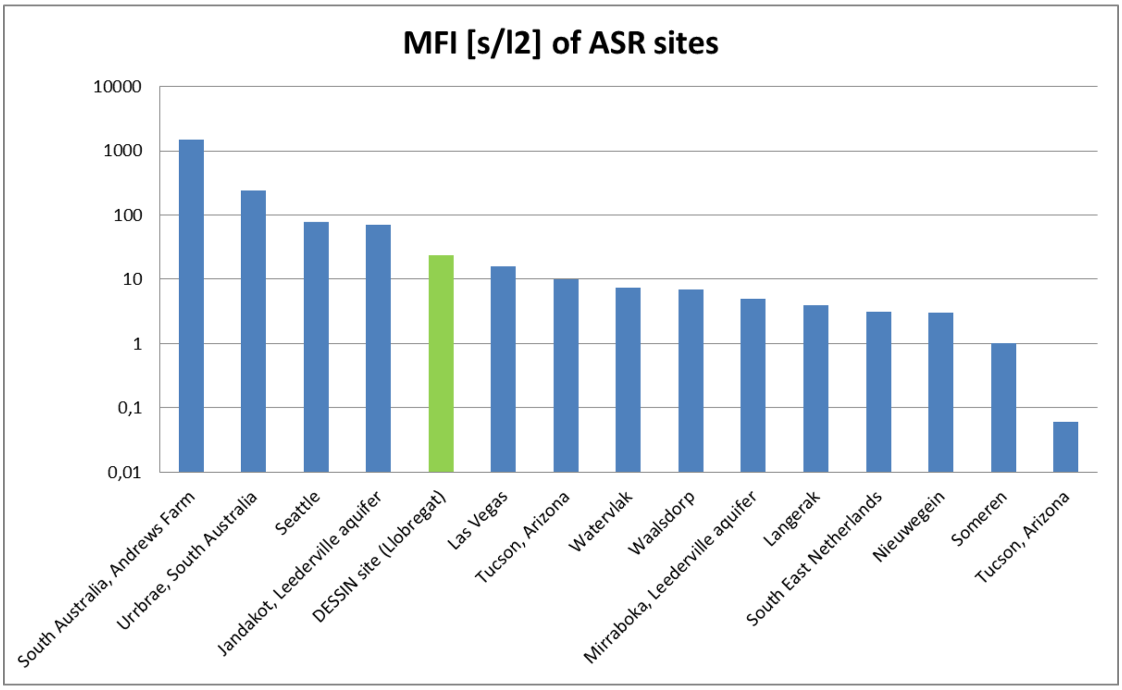



Figure 37: MFI of injection water reported for ASR sites

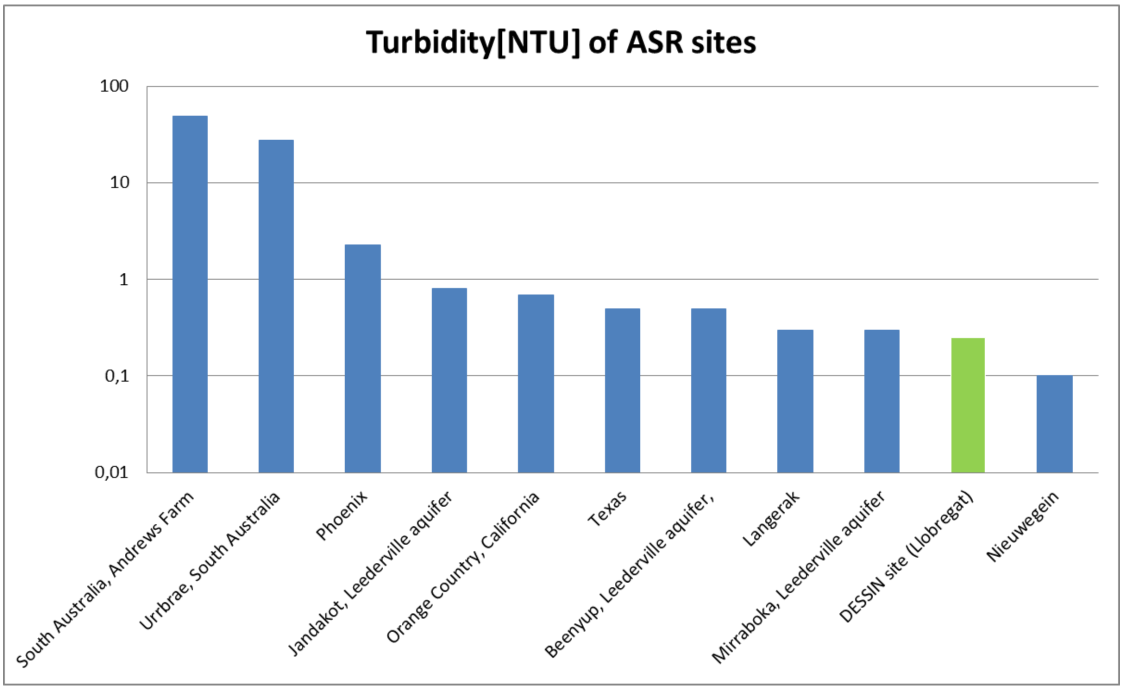



Figure 38: Turbidity of injection water reported for ASR sites

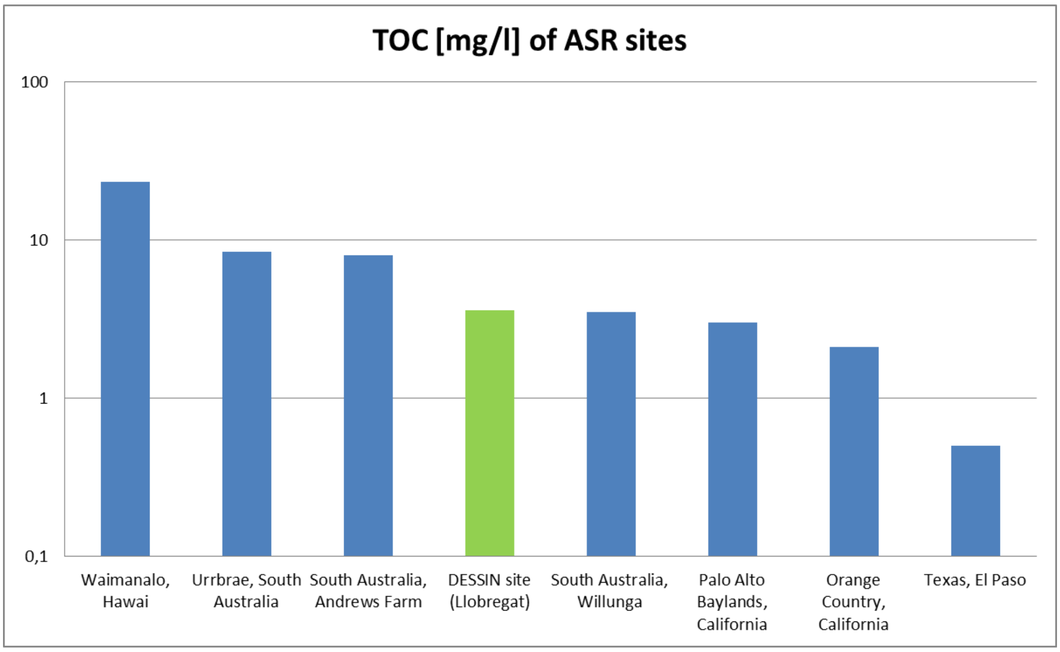



**Figure 39: TOC of injection water reported for ASR sites**

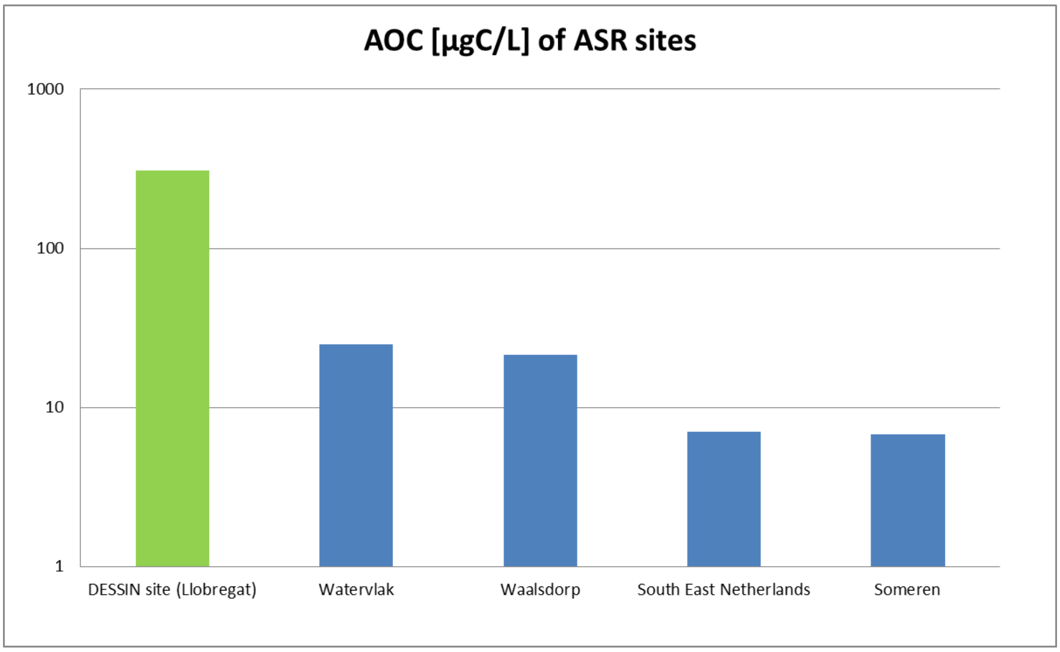



**Figure 40: AOC of injection water reported for ASR sites**

There is found only five data of ASR sites with AOC value and these ranged from 7 to 25 µgC/L. DESSIN Llobregat site value was taken from other studies and calculated indirectly using the TOC concentration, and maximum estimation has a value of 310 µgC/L. Hijnen (1991) recommended values minor than 10 µgC/L so in the upper range of AOC estimation DESSIN Llobregat site would have really higher values, meaning that there is a relative high potential of bioclogging formation.

There is also few data regarding microbiology indicators of injected waters. From the four data founded about faecal coliform this ranges from 100 UNF/100 mL to 600 UNF/100 mL. SFW of DESSIN site, as is not disinfected water it has an average of 730 UNF/100 mL.

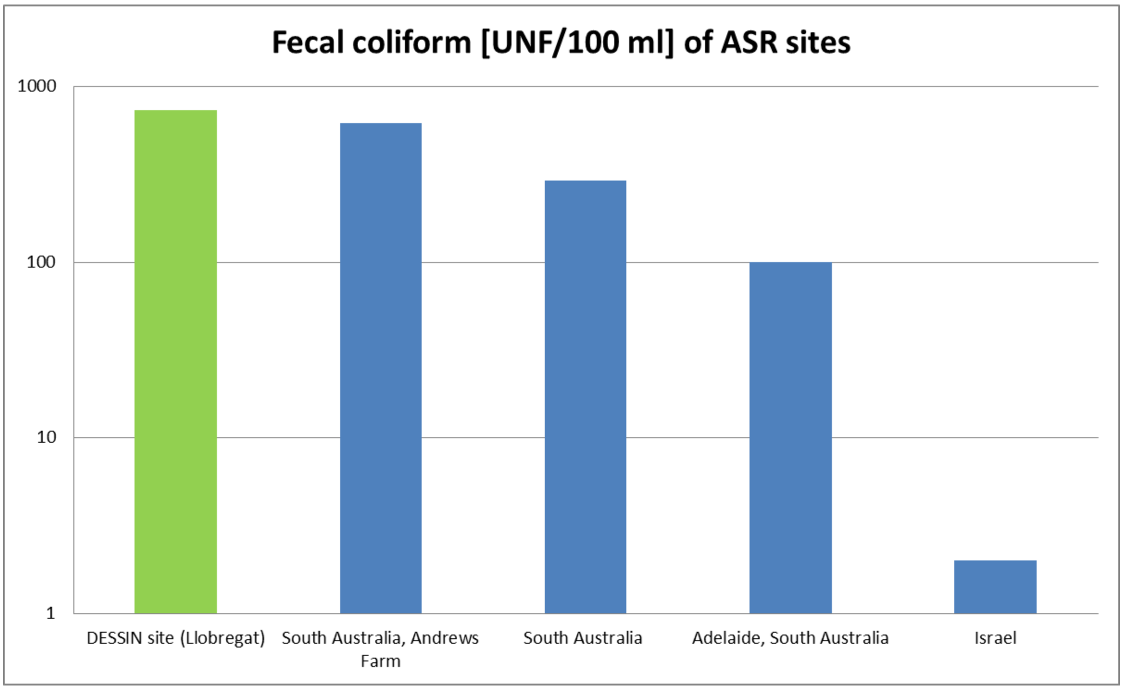



**Figure 41: Faecal coliform concentration in injection water of ASR sites**

# 8 Conclusions: identification of the strengths and weaknesses of ASR with pre-potable water in the Llobregat area

This report summarises an exhaustive review on ASR literature focused in two aspects: real ASR experiences with non-potable water and guidelines and recommendations for a successful and safe ASR projects implementation. This review has been practically applied to the case study of the Llobregat, where sand filtered water (SFW) coming from the potabilisation process of the drinking water treatment plant of Sant Joan Despí is intended to be done.

Before doing this work, a lack of knowledge existed in Aigües de Barcelona (operator of ASR system in the Llobregat) and local stakeholders about the existence of previous initiatives in injecting non-potable water in the aquifers to increase subsurface resources. Identified worldwide experiences illustrate the diversity of scenarios of ASR projects. 37 references have been selected and analysed from literature (data cited in articles, scientific papers and books). Most references have been obtained from Australia, The Nederlands and USA. The synthesis of the information have been presented in a table of characteristics allowing the quick comparison of the conditions of the aquifer, the dimensions of the projects and the pre-treatment and maintenance tasks carried out in each system.  This work will help Aigües de Barcelona and future potential operators implementing ASR schemes to position their system in terms of hydrogeology, water quality and expected risks of non-potable water injection.

Llobregat system is classified in a mean value of physical parameters causing clogging. Modified Fouling Index (MFI) and turbidity of injection water are not extreme values compared to other ASR projects. In contrast, the Llobregat aquifer seems to be really suitable in terms of hydraulics, as the reported transmissivity (about 30,000 $m^2/d$) is the highest transmissivity value reported in the receiving aquifer.

One of the important differences of SFW in front of other ASR injection waters is that SFW is not disinfected. This implies that there is a presence of microorganism load in the aquifer. This report evaluated the indicators microorganisms that are present in SFW: E. coli, Total Coliforms and Clostridium Perfringens. Their concentrations are in the range of 2-3 logs that could be attenuated by the aquifer in the range of days, according to literature. During the demonstration phase of SFW injection will be important to analyse the inactivation of microbiota indicators within the piezometers located near the injection well. The other important impact of non-disinfected water could be the formation of bioclogging within the borehole wall or well screen. Indicators as total organic carbon (TOC) and assimilable organic carbon (AOC) also suggest that bioclogging formation could happen during the demonstration, but is it very difficult to predict only with these indicators.

As it is no possible to access and see directly the well screen during the demonstration phase and evaluate bioclogging formation, in the subsequent tasks of the project will be done a piloting with a simulation of the well screen and the aquifer material. The pilot tests will operate in same conditions than the injection well. Therefore with this piloting conclusions will be evaluated if SFW needs any additional pre-treatments or not. The fact that the injected water is it not disinfected could give also positive impacts as the absence of disinfection by-products like trihalomethanes (THM's) or bromates in the injected water that will be also assessed.

The review of guidelines and recommendations reported in literature has been also applied to the SFW characterisation. Data from 2010 to 2014 of SFW have been plotted and aggregated in ranges to evaluate the frequencies and mean values of the bulk chemistry. TSS, MFI, Turbidity, DOC, TOC, AOC, E. coli and ammonium are the parameters mainly reported as clogging and pollution control in ASR injection. Other ions, metals and microbiological parameters have been analysed as SFW characterisation.

Finally, a multicriteria analysis has been applied to determine the most critical compounds and parameters that can be different in SFW and native groundwater. An own MS Excel Tool name AquaStoRe has been applied to evaluate the critical parameters of ASR injection in Llobregat aquifer using non-potable water. Review of guidelines and recommendations and AquaStoRe results converged on identifying the following weaknesses of the injection water: ammonium, pathogens and turbidity.

To sum up, the work presented in this report helped operators of Sant Joan Despí DWTP to identify critical aspects of the demonstration phase and orientate them about how to design pilot test and laboratory experiments to bring some light to what is expected at real scale in the injection well. The list of parameters evaluated in this report will be extended with additional requirements of local stakeholders. Compliance with Water Framework Directive and considering the risks of non-regulated or emerging pollutants will be also taken into account in the design of the analytical plan in the demonstration phase.

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# D22.4 Evaluation of pre-potable water requirements for a safe injection in the Aquifer through ASR

## CHAPTER B: Evaluation of pre-treatments and pilot test results

Cetaqua, December (2015)

Revised version, November (2017)

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# D22.4 Evaluation of pre-potable water requirements for a safe injection in the Aquifer through ASR

## CHAPTER B: SELECTION OF PRE-POTABLE WATER AND ADDITIONAL PRETREATMENT REQUIREMENTS


**SUMMARY**

In the previous deliverable D22.4(a) the ASR system of SJD DWTP was described and the pre-potable water injection was evaluated by reviewing literature recommendations, international experiences and historical data of SJD DWTP sand filtered water quality parameters. It was concluded that sand filtered water fulfils the different quality requirements for the injection in the SJD ASR system, but an experimentation and demonstration phase would be needed in order to validate the new DESSIN scheme.

This report describes the experimental evaluation conducted in real conditions in order to validate the use of sand filtered water as a pre-potable water to be injected in the aquifer and additionally, to evaluate if any additional pre-treatment is needed. As one of the sand filtered water drawbacks is the microbial content presence, tests of disinfection techniques have been tested as potential pre-treatment.

Results obtained will serve to extract robust conclusions of the consequences of pre-potable water injection in the well, and will give real conditions and recommendations for a correct future operation both during DESSIN demonstration phase in a real well and for a possible future implementation of complete ASR system.


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# List of Acronyms and Abbreviations

| | |
|---|---|
| AOC | Assimilable Organic Carbon |
| ASR | Aquifer Storage and Recovery |
| CFU | Colony Forming Unit |
| DOC | Dissolved Organic Carbon |
| DWTP | Drinking Water Treatment Plant |
| EDX | Energy Dispersive X-Ray |
| EPS | Extracellular Polymeric Substances |
| HAAs | Haloacetic Acids |
| LOI | Lost On Ignition |
| MCL | Maximum Contaminant Level |
| MFI | Membrane Fouling Index |
| MPN | Most Probable Number |
| MRDL | Maximum Residual Disinfection Level |
| ONORM | National standard published by the Austrian Standards Institute |
| SEM | Scanning Electron Microscope |
| SEM/EDX | Scanning Electron Microscope and Energy Dispersive X-Ray |
| SFW | Sand Filtered Water |
| SJD | Sant Joan Despí |
| TDS | Total Dissolved Solids |
| THMs | Trihalomethanes |
| TOC | Total Organic Carbon |
| WHO | World Health Organisation |



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# Executive summary

This report summarises the main findings obtained in the experimental phase performed to test Aquifer Storage and Recovery (ASR) with pre-potable water before its practical application in a real injection well in Sant Joan Despí DWPT. The objective was double (i) to gain knowledge about the clogging processes occurring along water injection with pre-potable water and (ii) to evaluate the potential disinfection methods to reduce the microbial load. To this end, an experimental methodology has been applied in Sant Joan Despí DWTP, taking advantage of the access to sand filtered water for the performance of the experiments.

To simulate the injection of pre-potable water (sand filtered water), an innovative experimental methodology has been developed to simulate the conditions of the aquifer based on column test. A column test simulating the well screen and the surrounding aquifer material has been assembled in SJD DWTP, being fed with water coming from the channel of sand filtered water. The fed water has been continuously circulated to the column with a constant flow of 3.7 L/min. The experiment has been conducted during 8 months, and nowadays the column is still operative to evaluate the long term effects. Pressure has been continuously measured at the bottom of the column to evaluate the head loss along the time in continuos conditions of operation. Pressure measurements in the bottom of the column have reported a 20% of head loss after 75 days of continuous operation. This increase in pressure does not mean a limitation in the aquifer capacity of infiltration of injection water, but an ascent in piezometric level is expected in the well.

The selected demonstration well (named P18 and located in SJD DWTP) was constructed in 1973, and the steel screen has a known design (grid openings and distribution). The pilot simulates the distribution and size of the screen holes using a system of a 4 x 4 embedded parts. This scheme allows to obtain 16 removable pieces to perform specific analysis. The extraction and substitution of these pieces along the time of the experiment allow the performance of specific measurement to better characterise the nature of the sediment and the evolution of bioclogging.

The composition of the muddy sediment settled in the pieces was analysed, being 11% organic and 89% inorganic content.  This is quite similar to the composition of the sludge obtained from the Llobregat River raw water. Scanning Electrode Microscopy (SEM) and elemental composition determinations identified some isolated bacillus and hifas, while most of the ubiquitous material observed with the microscope corresponds to biological mass aggregates, presumably extracellular polysaccharides (EPS). EPS have been quantified from the starting point of the experiment, observing a rapid increase in EPS concentration (from 0.3 to 48.9 µg glucose-equivalents /cm$^2$ in 44 days' time). Peaks of 60 µg glucose-equivalents /cm$^2$ have been quantified in the 140 day of the infiltration. EPS study reveals that there is a decrease on microorganisms growth and thus of microbiological activity after 5 months that the injection  begans.

Reviewed literature on ASR systems concluded that there is not a standard method on how to simulate ASR systems at laboratory scale, and results regarding hydrogeological changes and water chemistry are only qualitative and not directly applicable to field scale. These references encouraged researchers and operators to test as much as possible these ASR systems at field scale, devoting their efforts at these stage.

Disinfection methods have been tested using the calculation of logarithm reduction with increasing doses of disinfectant. Selected methods tested have been: chlorination, dioxichlorination, UV and ozonation. Sand filtered water presented from 1 to 3 logarithms of *E. coli, C. perfringens* and total coliforms, which are not very high pathogens loads compared to waste water disinfection studies. This represented a limiting condition to determine the effectiveness of disinfection methods. Besides calculating the optimal dose of each method, it was assessed the apparition of disinfection by-products, as bromates or trihalomethanes (THMs) in the disinfected water. THMs appeared with the disinfection methods test. The column experiments results joined to the low pathogen loads in injection water and the generation of disinfection by-products led to the decision of not appling disinfection as pre-treatment in the demonstrative phase of the DESSIN project (located in one single well, named P18). This will lead to test the depurative capacity of the natural system along the aquifer passage.

# 1 Introduction

Managed aquifer recharge operations using injection wells are often limited by clogging of well screens and the porous media of the aquifer immediately surrounding the screens. Under constant pressure operation, clogging can reduce the injection rate until an unacceptably low value. Moreover, when the system is recharged at a constant rate, clogging can cause excessive increase in well and aquifer head losses within days, weeks or months of operation depending on the clogging effect involved. Clogging processes, which can be extremely rapid, have often been shown to be the consequence of deposition of suspended sediments in the injected water within the pores of the aquifer. Other factors, such as bacterial growth, chemical reactions forming precipitates, swelling of clay minerals in the formation, the generation of gases, air entrainment or mobilisation, migration and deposition of fines are also capable of producing clogging. Clogging of recharge bores can be due to one or any combination of these factors. All these details of clogging formation and the effect in ASR systems are extensively described in previous report D22.4(a).

The cost-benefit analysis of managed aquifer recharge demands maximum infiltration rates along the duration of the injection phase. Therefore, as a matter of practical concern it is desirable to evaluate the potential apparition and evolution of clogging, and the strategies to counteract this phenomenon. For the given set of physical proprerties of Llobregat aquifer and sand filtered water chosen to be injected, a prediction of the susceptibility of recharge bores to suspended solids causing physical clogging and biofilm formation causing bioclogging is needed for the planning and operation of DESSIN ASR demonstration.

The purpose of this work is to investigate the nature of well clogging and biofilm formation using sand filtered water as injection water, representing similar conditions of operation than real aquifer injection system, which is going to be tested in demonstration phase. To this end, it was designed a pilot aquifer recharge experiment using: real sand filtered water produced in the DWTP in SJD (Barcelona), real aquifer material, well screen reproduction with same material and screen dimensions, same water injection velocity and dark conditions. Therefore, with this pilot experiment it is intended to obtain conclusions about the clogging fomation and will permit to see visually what it cannot be seen in a well, allowing the well screen evolution within the injection time. Furthermore it has been assessed potential pre-treatments of sand filtered water before injection, considering their advantages and drawbacks in case it would be necessary to have a more disinfected water for the injection.

# 2 Objectives

The objectives of this report are to summarize the findings in the task 22.4 of the DESSIN project regarding the experimental evaluation of pre-potable water requirements for a safe injection in the aquifer and the evaluation of additional pre-treatments requirements. Specifically, the objectives of these tasks are:

- **Evaluation of clogging occurring by the injection of pre-potable water at pilot scale:**
    - Design of the column experiment for aquifer recharge simulation.
    - Assessment of pressure loss along the experiment as indirect measure of clogging.
    - Evaluate the physical clogging formation in the well screen.
    - Biofilm characterisation.

- **Characterisation of potential pre-treatments to sand filtered water:**
    - Design and performance of laboratory experiments.
    - Determination of doses, pathogens removal and by-product formation of different disinfection methods.

- **Provision of recommendations for a successful injection of sand filtered water:**
    - Clogging control.
    - Selection of most suitable pre-treatment before injection if needed.

# 3 Colum experiment for the evaluation if the injection of pre-potable water at pilot scale

## 3.1 Design of column experiment

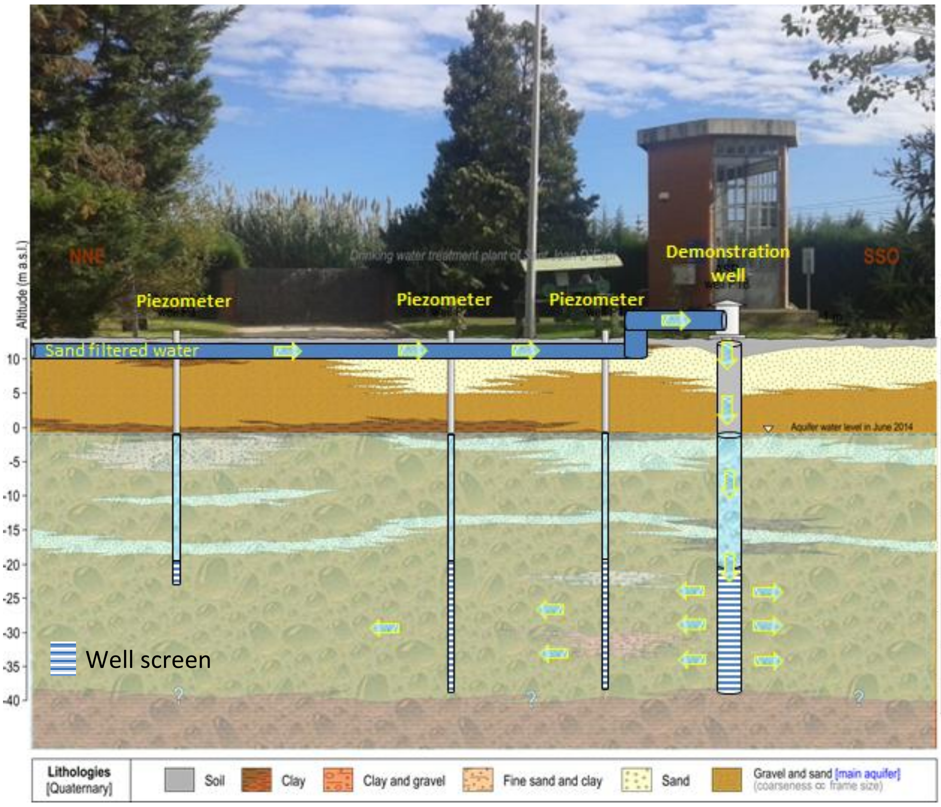
Deliverable D22.4(a) contains the evaluation of pre-potable water for injection using ASR, applied to the case study of the DWTP in SJD (Barcelona). International experiences and recommendations have shown that a wide range of parameters (MFI, AOC, DOC, TDS, Turbidity, pH, etc) are useful parameters for comparing relative clogging potentials of various types of water, but they cannot be used to predict clogging and decay in injection rates. The specific effects occurring in the injection wells also depend on well construction and aquifer characteristics. Thus, full-scale studies on injection wells are still necessary to determine feasibility, design and management criteria for operational ASR wells.

Therefore, in order to evaluate the clogging potential of sand filtered water of SJD DWTP, it has been designed an experiment using the real water to be injected and simulating as close as possible the conditions in the ASR well.

Figure 1 represents the operation of the ASR well where the injecting water (after primary treatment and sand filtration) is injected in the recharging well. Water is introduced by gravity into the borehole. Afterwards by pressure gradient the injected water flows into the aquifer by crossing the metallic well screen, located between 30 m and 50 m depth. Figure 2 represents the clogging formation in the well screen (painted in brown in the figure) or in the gravels of the aquifer surrounding the well (painted in orange in the Figure 2).

Then, in order to simulate injection process, it has been constructed an column-type experiment where injection water enters on the top and by gravity goes down and cross a metallic screen and a gravels pack, and finally goes out from the bottom (Figure 3). In order to monitor the clogging formation within the column it has been placed a pressure sensor at the bottom of gravel pack that allows to measure if there is any decrease in hydraulic conductivity (which would mean that experiment is being clogged).

**Figure 3** represents the scheme of the pilot column design and in the **Figure 4** can be seen a photo of the constructed column during its operation and placed near the sand filtered water canal in the SJD DWTP.



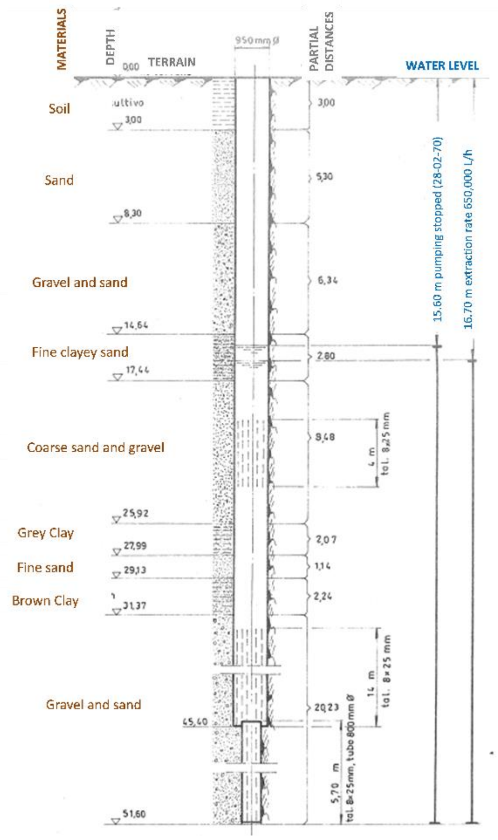




**Figure 1: SJD ASR system scheme and detailed well column**

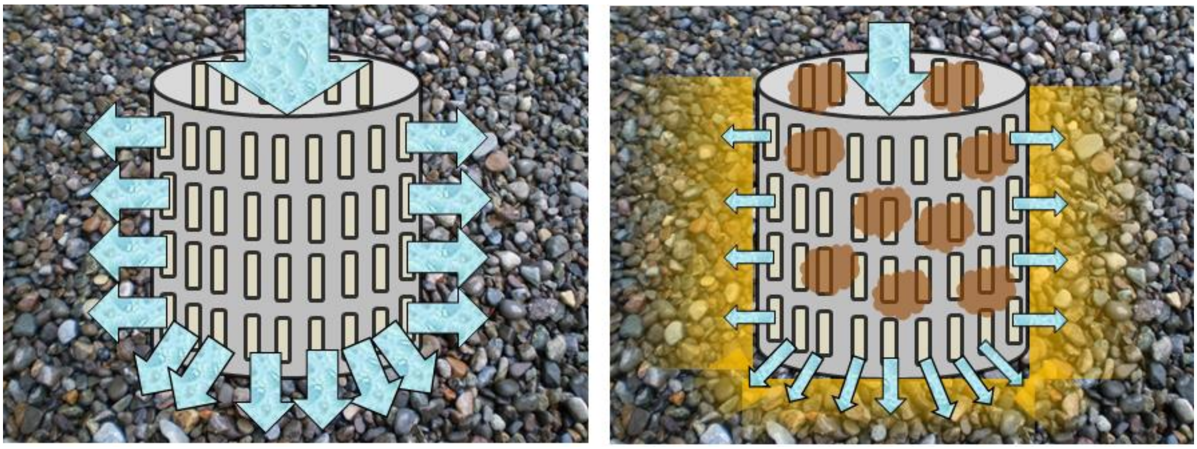


Figure 2: Well screen and aquifer representation without (left) and with clogging (right)

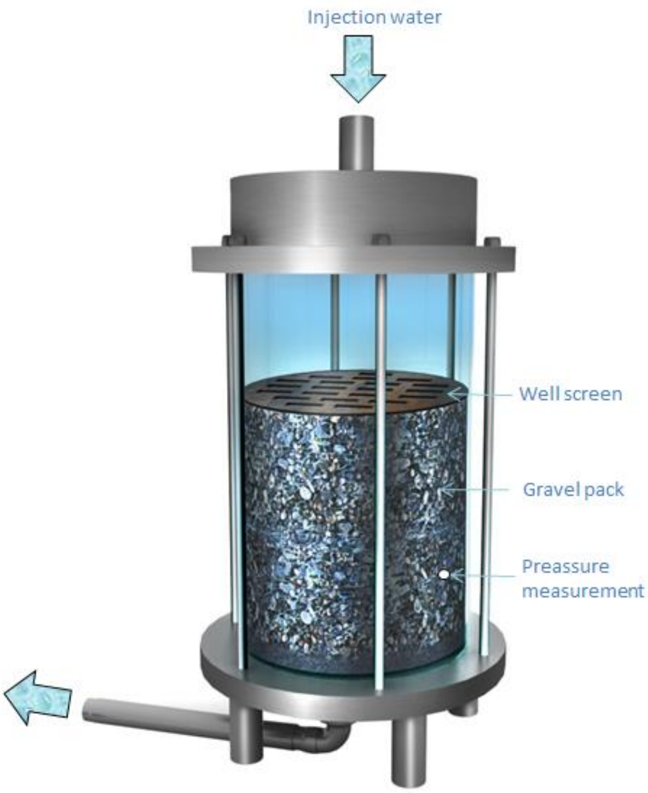



Figure 3: Pilot column simulating ASR

Figure 4: Water injection pilot located next to the sand-filtered water canal of SJD DWTP

As it can be seen in Figure 5 the simulation of metal screen it's done by 16 pieces that have exactly the same filter screen dimensions than the real well screen. This well screen simulation has been designed with independent ST52 steel pieces (4cm x 4cm*0.6cm) in order to be able to remove them separately for sampling purposes (total area covered 16 cm x 16 cm).

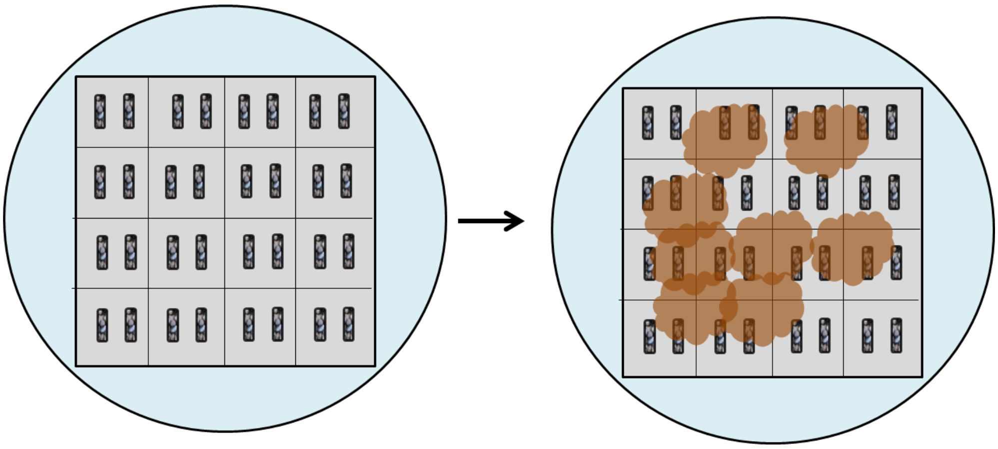


Figure 5: Well screen simulation design. Clean (left) and clogged (right)

**Figure 6: Picture of the well screen simulation in the column experiment**
NOTE: Left picture: before starting. Right picture: at the beginning of the operation

The design of the column also allows the observation the evolution of the clogging in the well screen, so during the experiment it has been taken photos continuously in order to have all the evolution recorded. As it can be seen in **Figure 5** with the passage of the sand filtered water along the weeks, the metal pieces get dirty and develop an orange layer that has been characterised (see sections 3.2.3 -3.2.5).

Below the well screen is placed a gravel pack that simulates the aquifer material surrounding the well. These filling materials came directly from the Llobregat aquifer and were extracted during the drilling of the piezometers of the demonstration phase; therefore the granulometry of the gravels simulates perfectly the hydraulic conditions within the aquifer. This makes possible to simulate the interaction between the sand filtered water and the aquifer material in order to evaluate as much as possible real conditions in the surroundings of the well screen.

Granulometry of filling material (gravels) has been characterised using Breddin curves method (Custodio & Llamas, 1984). This method establishes granulometry curves defining 12 types of sedimentary aquifers depending on the sediment grain size and gives an orientation of the hydraulic conductivity values. **Figure 7** illustrates that the gravels placed in the bottom in the column could be considered as Class 2 corresponding to coarse gravels and with an average transmissivity of around 600 m/day. This exercise have been also performed with the sand of the sand filters in SJD DWTP, Yellow line in the **Figure 7** represents the curve for this sand, which appears much more homogeneous, than the aquifer material, and with lower values of hydraulic transmissivity (43 m/day)

Comparing both granulometry curves, it can be understood that fine particles that could cross the sand filter bed should not be trapped immediately in the coarse material of the aquifer, meaning that there should be no expected problems regarding immediate physical clogging. However this hypothesis has be confirmed along the pilot experiment after months of sand filtered water passage through the column (see Figure 10 in section ).

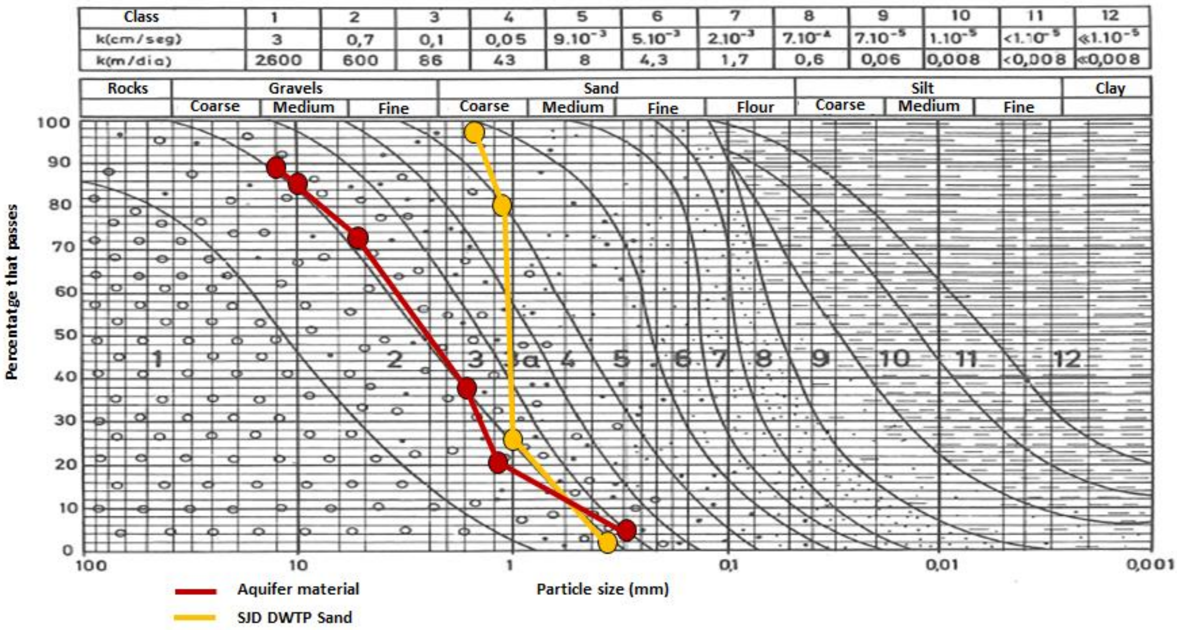



**Figure 7: Breddin curves of granulometry and hydraulic conductivity of porous media**

NOTE: Comparison between aquifer material granulometry and sand from SJD DWTP sand filters

## 3.2 Results: clogging evolution in the column experiment

For the clogging evolution assessment, several multidisciplinary tools were evaluated and have been applied. During the testa visual evaluation, a head loss measurement and a characterization of the material that cause the clogging through the analysis of the organic fraction, the analysis of the biofouling and the analysis by electronic microscope were conducted.

### 3.2.1 General trends and visual aspects

As it can be seen in Figure 8 with the advance of the injection time the well screen placed in the column was slowly covered by orange-brown fine viscous sediments forming the above mentioned clogging. Ths sediment layer increased with the time and after more than 70 days of continuous injection, it reached the status seen in Figure 9. A consequence of the clogging could be the full blockage of the screen grid openings, making impossible the passage of flow of the injection water into the aquifer. In the column experiment, after more than 70 days of continuous injection without any backwash, the grid openings were still open but some of them have reduced their open area (see images in Figure 8).

After several days of continuous injection, sediments deposited in the well screen and in the aquifer gravels below could be observed . However, the deposition of fine particles within the coarse gravels did not seem to be very problematic in terms of full porous gravels clogging.

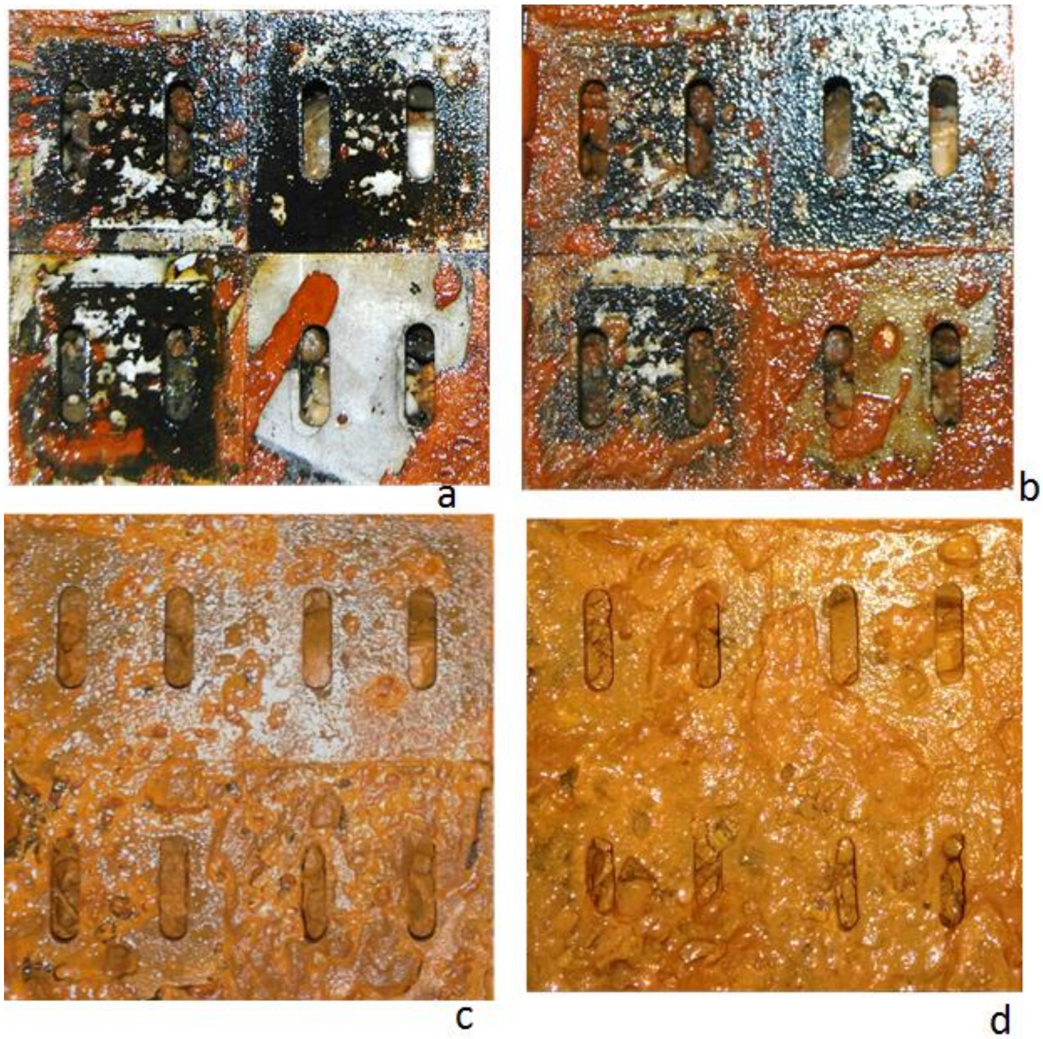



**Figure 8: Evolution of the column well screen clogging after 1(a), 20(b), 29(c) and 73(d) days**


**Figure 9: Sediments layer in well screen after 73 days**

### 3.2.2 Pressure measurements: head loss over time

A pressure sensor was placed in the column in order to measure the head loss within the well screen and the gravel pack caused by the clogging formed during the experiment. As the flow rate within the column was maintained equal during all the experiment through the valve control, the pressure measurement can be considered a direct indicator of the clogging developed within the injection time. Thus, the more clogged the column is the more head loss the column have.

In Figure 10 the evolution of the head loss increase during the pilot column experiment is depicted. During the first 30 days of continuous injection time, the head loss increased constantly until a maximum of 15% of the initial value. After this first 30 days period, it seems to occur an increase in dispersivity of data, having maximum head losses between 10 and 20%.

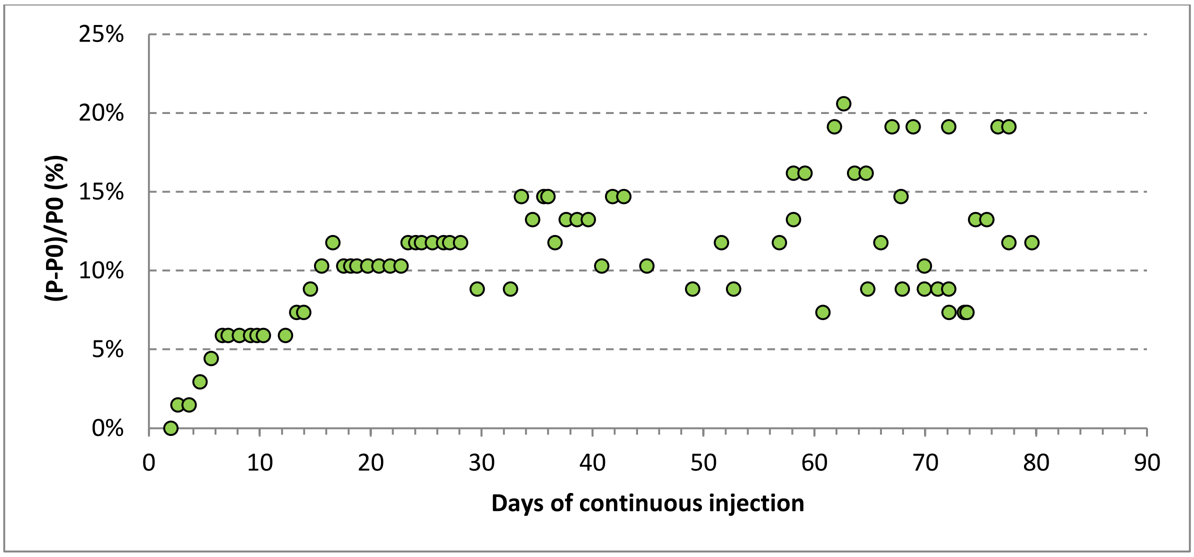



**Figure 10: Normalized head loss in the column experiment within 70 days of injection**

The results obtained show that during the first days of the injection, there is a faster growing of the clogging, but it does not represent a serious problem as it would not reduce the injection rate neither could block the well screen or the porous material, causing an irreversible well damage. Moreover, after the first period, pressure data are more disperse. Taking into account the pressure measurement data and effects in the head loss measured, it could be concluded that there should be no problems in the operation of the SJD ASR system when injecting SJD sand filtered water.

### 3.2.3 Extracellular polymeric substances evolution

Methodology

As explained in Romani et.al (2008) biofilms are structured communities of bacteria, algae, cyanobacteria, fungi, and protozoa embedded in a polymeric matrix. Most microorganisms found in biofilms produce extracellular polymers, which lead to adhesion to the substrate and comprise the polymeric matrix responsible for biofilm integrity, as seen in **Figure 11** Extracellular polymeric substances (EPS) are rich in high molecular weight polysaccharides and other non-sugar compounds such as proteins. The EPS matrix is a crucial structural parameter for biofilm stability and architecture and provides a refuge for the microbial community against shear stress and protection against desiccation.

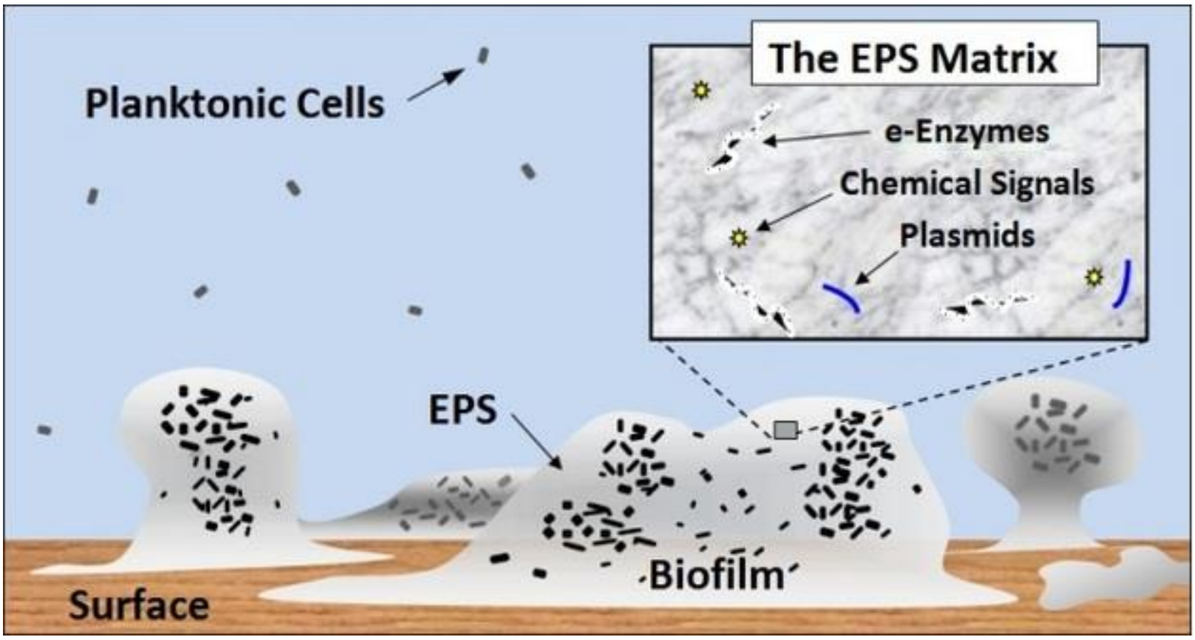



**Figure 11: Biofilm configuration with EPS matrix**
(Extracted from http://www.nature.com/)

Therefore in order to determine the biofilm formation in the well screen and evaluate the bioclogging formation potential of the sand filtered water, successive EPS analysis had been conducted along with column experiment. Following the sampling and analysis method described in Romaní et al. (2008), 8 series of samples with two replicas (16 samples in total) of the biofilm formed in the metal pieces of the simulation of well screen have been taken during the 8 months (April-November 2015) . In order to take the sample,1 cm² of the metallic piece was scrapped with a cell scapper (as shown in Figure 12 ) and the material was removed using a phosphate tampon solution and stored in an eppendorf. The analysis results of the EPS content in the biofilm of the different samples were given in µg of glucose quivalents per $cm^2$.

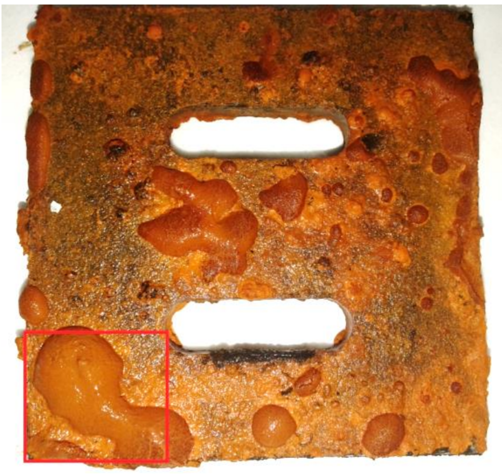


**Figure 12: Metallic piece of the well screen simulation with biofilm material deposition**

<u>Results</u>

Biofilm formation is a complex process depending on the microbial species, its communication and motility skills, the specialties of the surface where the biofilm will be attached, and, of course, the quality as well as chemical and physical properties of the surrounding aquatic environment. On a surface such as metal, biofilms allow for a variety of microorganisms with differing redox potential requirements to reside in close proximity and metal (Else, 2003). In this way, biocorrosion can occur during biofilm formation, as was in the case of metallic pieces of the well screen. Figure 13 depicts EPS content related to the five stages of biofilm formation:

1. Attachment: at first, single bacteria will reach the surface through the liquid phase usually by swimming. Single bacteria will attach to surfaces using flagella and other surface appendages (Kearns, 2010). This attachment is generally reversible and it is largely mediated by “van der Waals forces”. However, early stages of biofilm development will depend on the specific strain. In nature biofilms, other eukaryotic organisms interact with the biofilm, forming part of it, such as fungi, algae, yeasts, protozoa and other microorganisms. Attachment phase took place on the well screen during the beginning of the experiment, in April.

2. Adhesion: during the second stage bacteria will slowly but tightly adhere to the surface via pili, proteins, polysaccharides and fimbriae. Filamentous fungi will carry out deposition of spores or other propagules such as hyphal fragments or sporangia. Diatoms will attach to the substratum by the production of mucilage, which will encapsulate the cells.

3. Proliferation: is a subsequent step characterized mainly by the proliferation and production of extracellular polymeric substances (EPS). During this stage, cells lose their flagella-driven motility and the whole system turns to be immobilized. EPS are not unique to bacteria; some of the most abundant EPS producers are microalgae (in particular, diatoms). Fungi (yeasts and molds) also produce EPS (Flemming, 2010). Second and third stages occurred during the months of May and June.

4. Biofilm maturation: during this step microorganisms continue to proliferate and will excrete larger amounts of hydrated EPS consisting of polysaccharides, proteins, nucleic acids and

lipids, providing stability to the biofilm as a whole and additional shelter to individual microorganisms. Fourth stage corresponds to the period between July and August.

5. Release or detachment: finally motile cells may disperse from the film; by diverse mechanisms. Cells from the biofilm will attach at other places and will promulgate the spreading of the film. Dispersal of fungi involves spore dispersal or release of biofilm fragments. Finally, the last stage took place from September to the end of the experiment, in November.

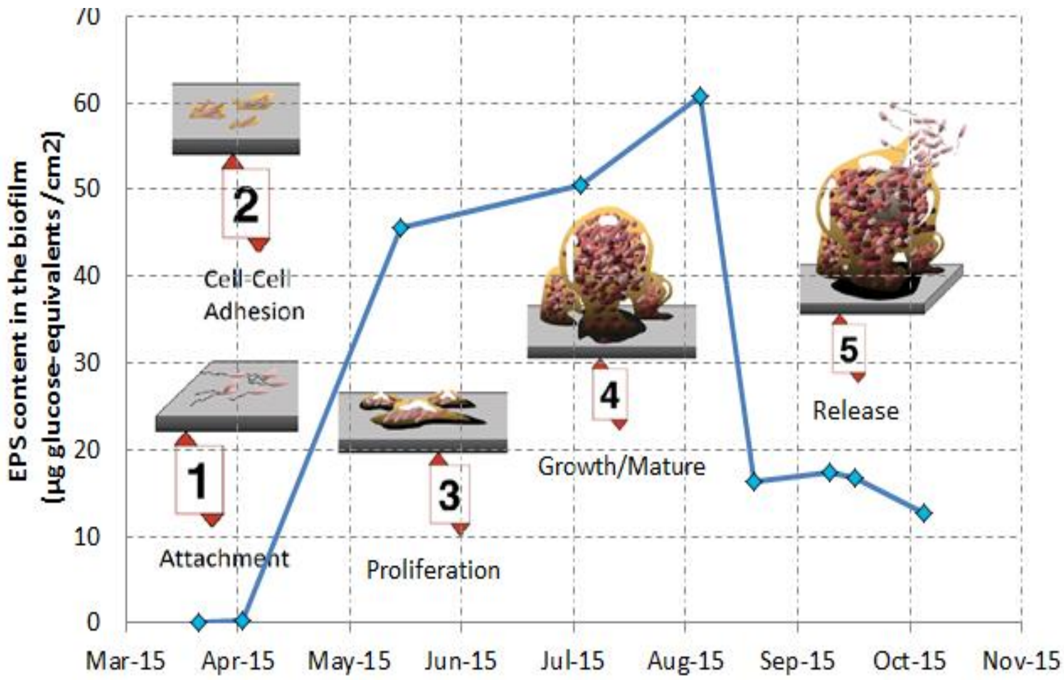



**Figure 13: Biofilm content evolution by the EPS analysis**
NOTE: Draws from Monroe (2007)

### 3.2.4 Estimation of organic and inorganic fraction

Methodology

Loss on ignition determination analysis has been performed to quantify the organic fraction of the material placed in the well screen. This method was described by Dean (1974) and it is based on sequential heating of the samples in a muffle furnace. After oven-drying of the sediment to constant weight (usually 12–24 hours at 105 °C), organic matter is combusted in a first step to ashes and carbon dioxide at a temperature between 500 and 550 °C.
The loss on ignition is then calculated using the following equation:

$$LOI_{550} = ((DW_{105}–DW_{550})/DW_{105})*100$$

where $LOI_{550}$ represents loss on ignition at 550°C (as a percentage), $DW_{105}$ represents the dry weight of the sample before combustion and $DW_{550}$ the dry weight of the sample after heating to 550°C (both in grams). The weight loss should then be proportional to the amount of organic carbon contained in the sample and Dean (1974) showed a strong correlation between LOI at 550°C and organic carbon content determined chromatographically in lake sediments.

Results

It were taken a total of six samples of the fine sediments placed in the well screen of the column (as shown in Figure 9) during the pilot experiment. The results of loss on ignition determination are that the clogging material formed has an average of around 11% of organic fraction. This results would suggest that there is a formation of bioclogging as previously shown, but a bigger fraction of the clogging material are inorganic particles that remained in the sand filtered water and are slowly deposited in the well screen.

**Table 1: Design guidelines to prevent ASR clogging**

| Sample | LOI [%] |
|---|---|
| 1 | 11.2 |
| 2 | 11.6 |
| 3 | 9.9 |
| 4 | 10.3 |
| 5 | 10.8 |
| 6 | 10.9 |

### 3.2.5 Scanning electron microscopy (SEM) observations and energy dispersive X-ray (EDX) microanalysis

In order to have a clearer idea of the morphology of the clogging layer formed in the well screen of the pilot column and to identify biofilm formations it was decided to perform a SEM microscopy observations. As it can be seen in Figure 9, it was placed in the column a 1 $cm^2$ steel piece that within the experiment advance was covered with the clogging material. This 1 $cm^2$ piece was what was placed inside the SEM microscope.

SEM is a very useful technique for the investigation of surface structure of biological samples. Much of the current knowledge about biofilms is due to the advances in imaging studies, especially the SEM (El Abed *et al.*, 2012). The analysis of characteristic X-rays emitted from the sample gives more quantitative elemental information. Furthermore, SEM/EDX is the best known and most widely-used of the surface analytical techniques. High resolution images of surface topography, with excellent depth of field, are produced using a highly-focused, scanning (primary) electron beam.

Methodology

**Sample preparation for SEM/EDX**

Sample preparation was carried out at the Department of TEM-SEM Electron Microscopy of Barcelona University. For scanning electron microscopy (SEM) observations and energy dispersive X-ray (EDX) microanalysis (SEM/EDX) analysis, samples of biofilm were fixed in 2.5% glutaraldehyde in phosphate buffer at 4°C (0.1M and pH7.4) for 2 hours and washed four times (10 minutes each wash) in the same buffer. Samples were immediately postfixed in a mixture of 1% osmium tetroxide and 0.8% potassium ferricyanide in phosphate buffer at 4°C. After washing samples six times in ultrapure water (first wash quick and another 5 washes every 10 minutes, at 4°C), samples were then dehydrated at 4°C in successively increasing gradient concentrations of ethanol (50% ethanol for 10 minutes, 70% ethanol overnight, 80% ethanol for 10 minutes, three changes of 90% ethanol for 10 minutes each, three changes of 96% ethanol for 10 minutes each, three changes of 100% ethanol for 10 minutes each) and dried by critical point, where ethanol was replaced by liquid $CO_2$ and changed to gas without changing its density. Finally, samples were mounted on metal stubs and coated with carbon.

**SEM/EDX microanalysis**

Microscopic observation with SEM/EDX was performed at the Department of Scanning Electron Microscopy of Barcelona University. A JEOL JSM-7100F scanning electron microscope (JEOL LTD, Tokyo, Japan) was used to view the images. An energy dispersive X-ray analyzer (Oxford Instruments, Bucks, UK, INCA-250 model) operated at 20 kV coupled to SEM was used for the elemental analysis of the samples.

<u>Results</u>

SEM/EDX analysis showed a highly developed biofilm with a dense matrix of EPS. The solid-liquid interface between the surface (well screen) and aqueous medium (sand filtered water) provided an ideal environment for the attachment and growth of microorganisms. Different microorganisms were found in the biofilm structure, such as bacteria, algae and fungi. Observations presented in the following images demonstrated the variety of microorganisms in the clogging layer. Besides crystal structures were observed in the biofilm structure. As it can be seen in **Figure 14**, due to dense biofilm matrix, it was only observed bacilli in specific parts of the sample. Regarding algae, it was observed in the sample different types of diatoms. A unique feature of diatom cells is that they are enclosed within a cell wall made of silica (hydrated silicon dioxide) called frustule, hence the presence of silica in the analysis by EDX **Figure 15 (d)**. Diatom shown in **Figure 15 (a)** could belong to the genus Stauroneis or Navicula, but the image did not allow confirm to what genus belongs this diatom **Figure 15 (b)** and **(c)** represent *Cyclotella sp.*, which has been identified by its smooth central zone without any depressions and surrounded by grooves. Presence of filamentous fungi were also observed in **Figure 16 (a)** and **(b)**, as well hyphae, branching filamentous structure of fungi, and a sporangium in **Figure 16 (c)**, enclosure in which spores are formed. Furthermore crystal structures were detected on the sample, where iron, calcium and phosphorus deposits were detected by SEM/EDX analysis (**Figure 17**).

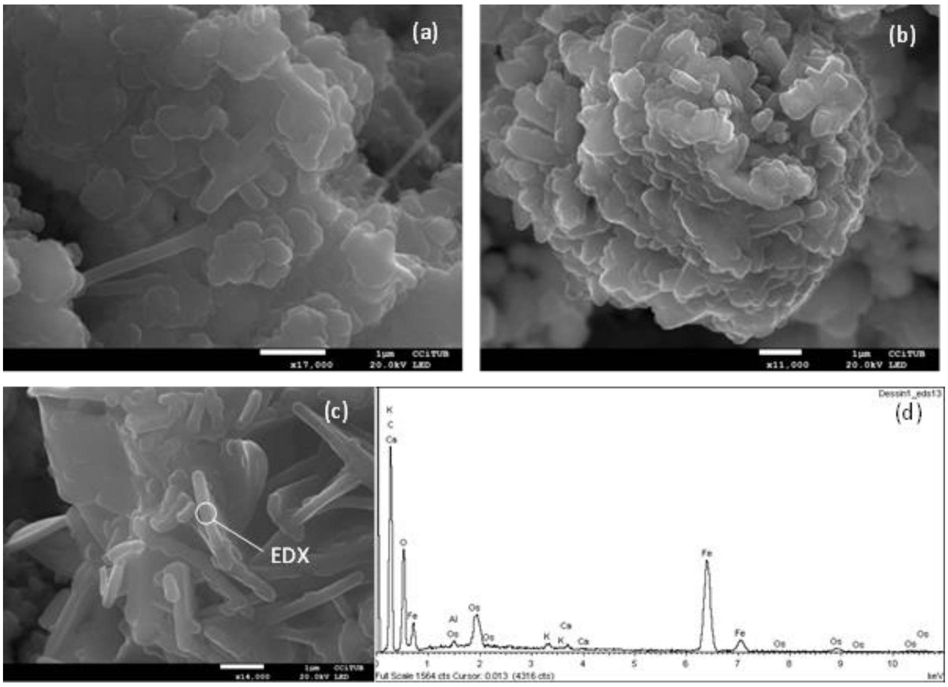



**Figure 14: SEM images (a-c) and EDX analysis (d) of the bacteria in the biofilm**

(a)Bacteria covered by the biofilm matrix. SEM image at 17,000x magnification (b) Bacteria congregate on the biofilm surface (11,000x magnification) (c) Bacilli present on the surface (14,000x magnification) (d) EDX spectrum of the region circled in (c) belonging to a bacillus, showing the presence of carbon, oxygen, potassium, aluminum, calcium, osmium and iron. Osmium and iron occur due to the sample preparation process. (White scale bars = 1µm)

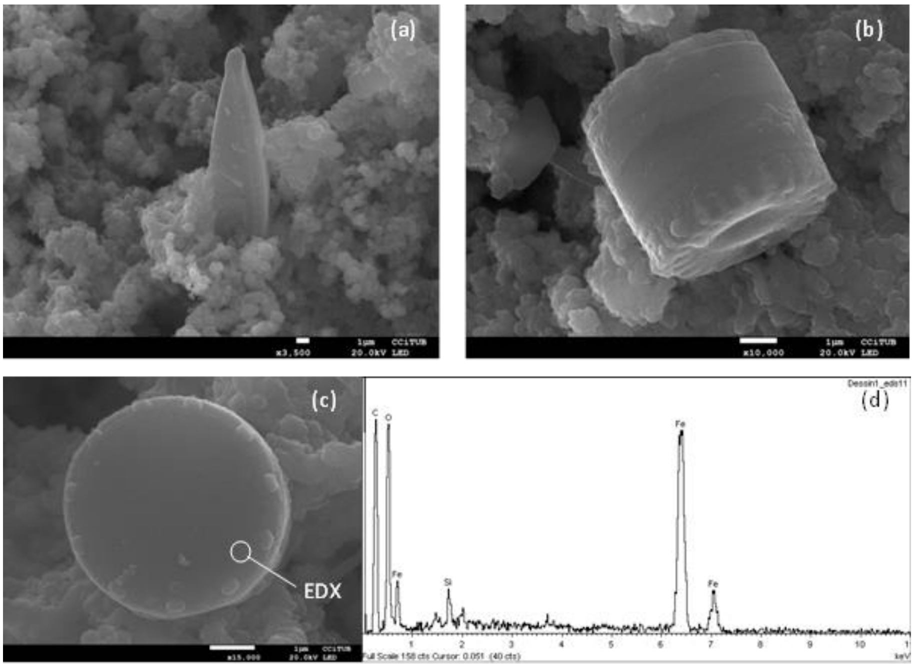



**Figure 15: SEM images (a-c) and EDX analysis (d) of diatoms present in the biofilm**

(a) Front-side view of a diatom (3,500x magnification) (b) Top view of a diatom, *Cyclotella sp.* (10,000x magnification) (c) Front view of a diatom, *Cyclotella sp.* (15,000x magnification) (d) EDX spectrum of the region circled in (c), showing the presence of carbon, oxygen, iron and silica. Iron occurs due to the sample preparation process and self-oxidation of steel plates, where the biofilm is formed. (White scale bars = 1µm)

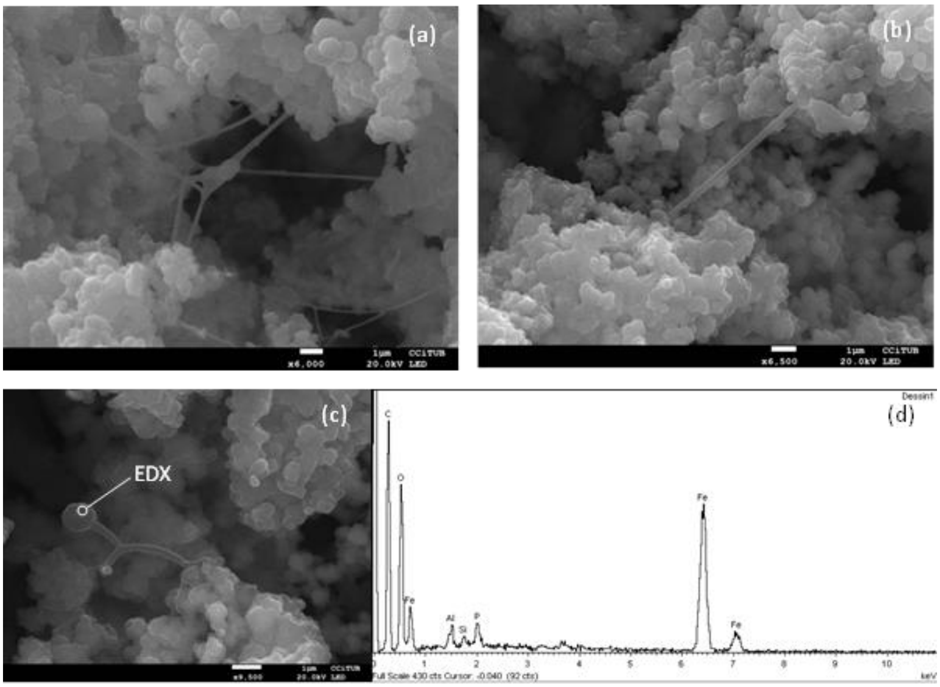



**Figure 16: SEM images (a-c) and EDX analysis (d) of fungi present in the biofilm**
(a) (b) Filamentous fungi present in the biofilm structure at 6,000x and 6,500x magnification respectively (c) Sporangium and hyphae (9,500x magnification) (d) EDX spectrum of the region circled in (c) belonging to sporangium, showing the presence of carbon, oxygen, iron, aluminium, phosphorus and silica. Iron occurs due to the sample preparation process and self-oxidation of steel plates, where the biofilm is formed. (White scale bars = 1µm)

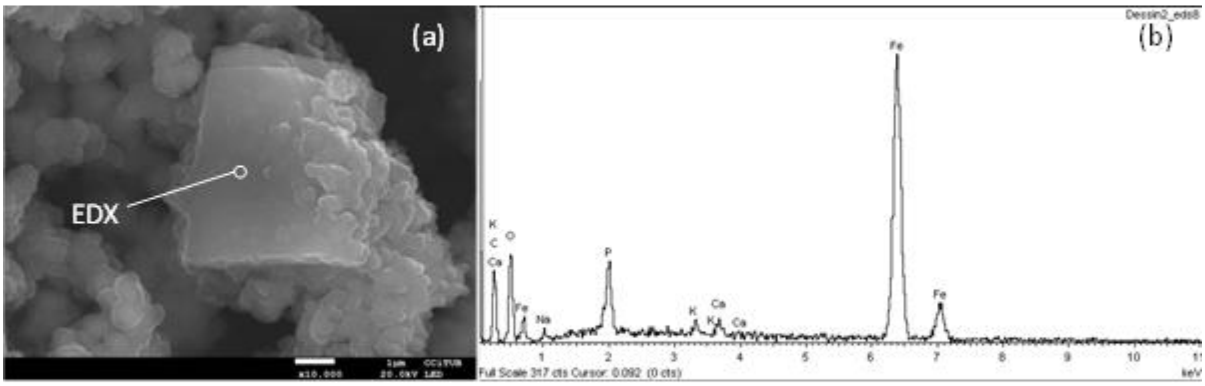



**Figure 17: SEM images (a) and EDX analysis (b) of crystal structures present in the biofilm**
(a) Crystalline deposits on the surface of the biofilm (10,000x magnification) (b) EDX spectrum of the region circled in (a), showing the presence of carbon, oxygen, phosphorus, potassium, sodium, calcium and iron. Iron occurs due to the sample preparation process and self-oxidation of steel plates, where the biofilm is formed. (White scale bars = 1µm)

### 3.2.6 ASR well backwashes

Methodology

In order to avoid the decrease of the injection capacity within the ASR systems, the normal procedure is to perform backwashes in the system by pumping part of the injected water with a higher flow rate than the injection. This procedure cause the detachment of part of the clogging formed and maintains the injection rate within the ASR system. In the SJD ASR system, based on their experience is used a procedure of making a backwash with a four times higher flow than during the injection every 15 days.

In our column pilot experiment it was decided not to follow this procedure in order to evaluate the clogging with a higher limitation, so the pilot was operated with a continuous injection during all the experiment (without backwashes). However, although it would not appear in the present deliverable, as the pilot is still available and the DESSIN project still running, the backwash procedure will be evaluated in the future.

## 3.3 Column experiment conclusions

After the evaluation of the different parameters related to the well clogging by means of experimental methods, it could be concluded that the sand filtered water of SJD DWTP would be suitable for the injection in the Llobregat ASR System .

During the 8 months experiment period, the non disinfected sand filtered water caused a bioclogging formation and its fine particles had been attached to the well screen and gravels forming a physical clogging. However, the configuration of Llobregat ASR System with a well screen with wide grid openings and gravels with high transmissivity allowed that the clogging formed do not caused a blockage or limitation of the injection water flow. Therefore all the clogging related effects detected and quantified seems not problematic for the operation of the Llobregat ASR System.

The limitation in time of the experiment allowed to evaluate the short and mid term effects, so it should be taken into account possible other long term effects.

# 4 Identification of potential pre-treatments

In parallel to the experimental evaluation of sand filtered water with the pilot column experiment, a laboratory study regarding potential additional treatments to be applied to the sand filtered water to reduce the microbiological content (sand filtered water was not disinfected before injection in the past) was conducted.

As it was described in the previous deliverable D22.4(a), one of the threats identified of sand filtered water is the microbial load that could affect the well operation in terms of bioclogging formation and the aquifer water quality in terms of pathogens increase. Therefore, it was decided to evaluate the performance and the operational parameters of different disinfection methods like chlorination, dioxichlorination, ozonation and ultraviolet light.

The performance and comparison between disinfection methods is based on the microbial indicators removal obtained in each treatment, assessing also different doses for treatment. A serial of disinfection treatments were conducted in the laboratory with water from the sand filtered water in the channels of SJD DWTP. Selected indicators and control parameters are listed below:

- Total coliforms (Most probable number – MPN)
- *Clostridium perfringens* (Colony Forming Unit – CFU)
- Trihalomethanes, Chlorates, Chlorites and Bromates as a by-products

As it was presented in previous report D22.4(a), sand filtered water of SJD DWTP presents an average values between 0 to 1000 MPN of total coliforms (1 and 3 logs respectively) and an average of 0 to 100 MPN of *C. perfringens*. (1 and 2 logs respectively). Figure 18 illustrates aggregate results of these parameters in sand filtered water in SJD from 2010 to 2014.

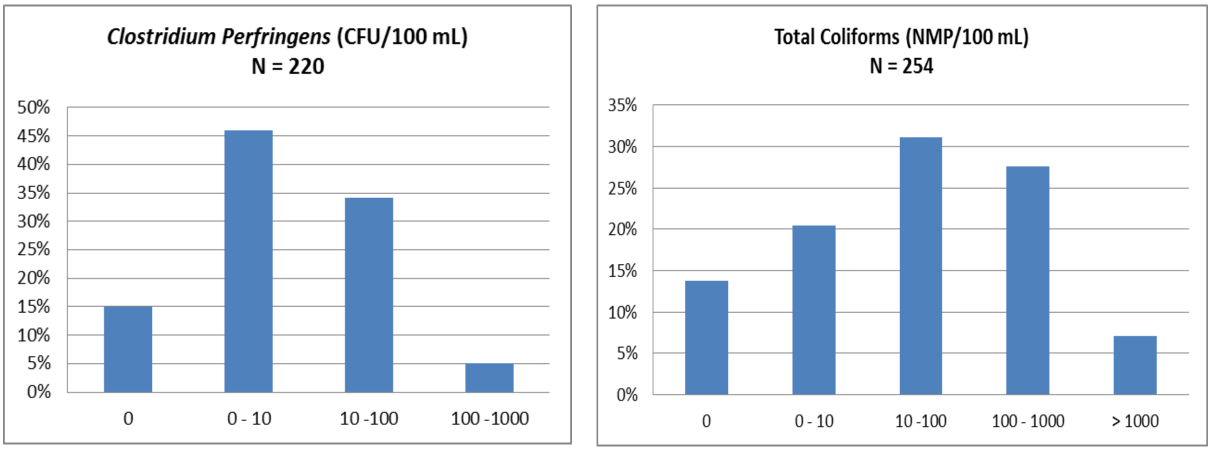



**Figure 18: Total coliforms and *C. perfringens* load of SJD DWTP sand filtered water**

## 4.1 Experimental methodology

Four disinfection methods have been evaluated and compared to eliminate the microbial load of sand filtered water:

- **Chlorination**: Chlorine is the most common solution for disinfection because apart from its use as a primary disinfectant, it provides a residual disinfection concentration that allows bringing safe and disinfected water until the tap of the consumer. Water chlorination is applied adding chlorine ($Cl_2$) or hypochlorite to water.

- **Chlorine dioxide:** is a strong oxidant and it is effective in the inactivation of pathogens. Its oxidizing ability is lower than ozone but much stronger than chlorine and chloramines. The pathogen inactivation efficiency of chlorine dioxide is as great as or greater than that of chlorine but is less than ozone. However different microorganisms have different sensitivity to $ClO_2$, e.g. Cryptosporidium require an order of magnitude higher Ct values compared to Giardia and viruses. Generally, chlorine dioxide is more effective as a disinfectant than chlorine at higher pH but similar or poorer at lower pH; chlorine dioxide performance is generally quoted as not being pH sensitive in the range experienced in water treatment, whereas chlorine is much more effective at lower pH.

- **Ozonation:** Ozone is a powerful oxidizing agent that can disinfect water in concentrations and with contact times lower than those of weaker disinfectants such as chlorine or chlorine dioxide. Despite this, ozone is only used as a primary disinfectant because it cannot remain in sufficient residual concentrations in the distribution system (no residual power). To achieve complete disinfection, it is therefore necessary to combine ozone treatment with secondary disinfection using chlorine or chlorine dioxide.

- **UV disinfection:** Ultraviolet rays are electromagnetic radiation with a wavelength from 100 to 380 nm. This rays act directly on the DNA of pathogens, causing the dimerization of the thymine, blocking the growth, replication and pathogenic power of the microorganisms (CIRSEE, 2009). It has been demonstrated that the effect of UV is maximized at around 254 nm (EPA, 2011).

A total of 10 samples of sand filtered water and were used as target water to be disinfected by the four different methods. The objective of using 10 initial samples was to test different dose of desinfectant. Despite total coliforms and *C.perfringens* varied in time, the assessment of the effectiveness of the disinfection methods was calculated using the decrease in logs. Figure 19 represents the initial microbial load in the experiments (around 2 log for total coliforms and 1-2 logs for *C.perfringens).*

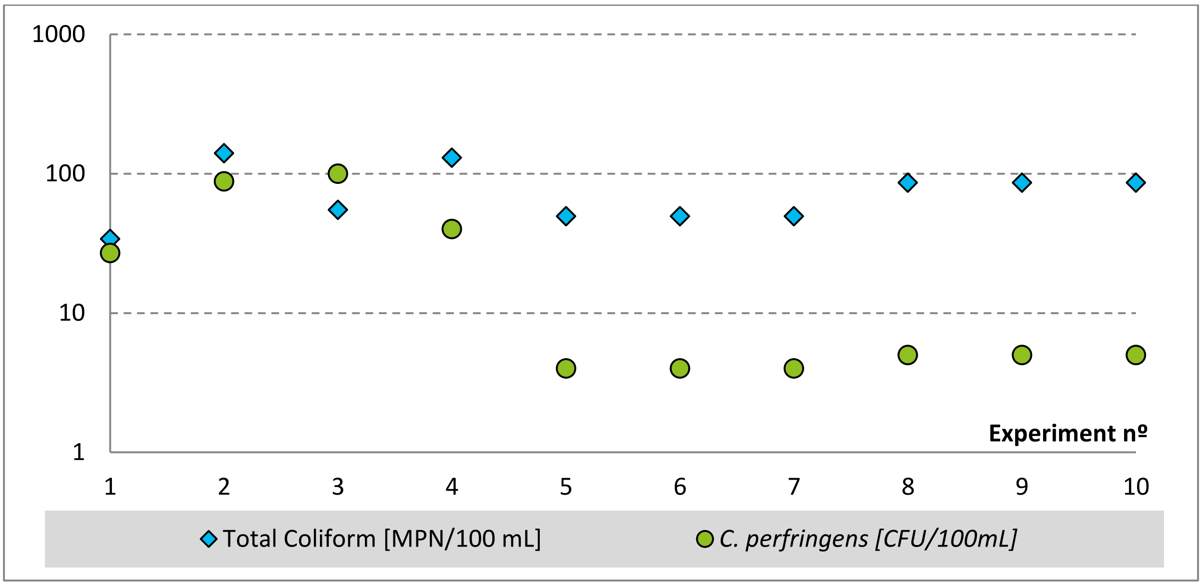



**Figure 19: Microbial loaf of SFW used in the 10 disinfection experiments done**

The dose of the different disinfection methods will be targeted in order to remove this microbial load amount. The disinfectants dose is specified with the parameter Ct. Ct is the product of residual concentration and time of exposition, expressed in mg/L·minute.

From literature review the reference values for the initial dose were taken, and upper and lower doses were applied in the experiment. In Table 2 an example of initial reference values of doses of different disinfection methods for 2 log abatement of different species are shown.

**Table 2: Reference values of Ct needed in mg/L·min for 2 log abatement at pH 7**

| Parameter | Chlorine (15ºC) | Chlorine dioxide (15ºC) | Ozone (15ºC) | UV [J/m2] |
|---|---|---|---|---|
| Giardia | 50 | 6.50 | 1.00 | 520.00 |
| Enteroviruses | 2 | 0.5 | 0.5 | 1450.0 |
| Coliforms | 0.02 | 0.3 | 0.01 | 430 |

### 4.1.1 Chlorination: Methodology and experimental doses

Overview

Chlorine is typically dosed in form of chlorine gas, sodium hypochlorite or calcium hypochlorite. Although chlorine gas has the advantage of being the cheapest option, its dangerousness and the cost of increased health and safety measures makes sodium hypochlorite a good alternative. Table 3 compares the three options of chlorination (CIRSEE, 2009).

Table 3: Most common methods to treat water with chlorine

| Product | Form | % Chlorine | Stability | Safety | pH effect |
|---|---|---|---|---|---|
| Chlorine gas | Pressurized liquid | 99.00 | Very high | Toxic gas | Reduction |
| Sodium hypochlorite | Liquid | 15.0 | Instable (light, heat) | Corrosive | Increase |
| Calcium hypochlorite | Solid | 60-70 | High | Corrosive | Increase |

Chlorine gas is manufactured off site as a gas, liquefied under pressure and stored has a liquid. The main problem with this chlorination option is the high toxicity gas and the potential harm that a leak would cause.

Sodium hypochlorite, the second most commonly used form of chlorine, is supplied as an aqueous solution with a maximum concentration of 15% w/w $Cl_2$. It presents some problems (CIRSEE, 2009):

- Brings inorganic by-products such as bromates and chlorates.
- Increases the pH of water.
- When water is hard, a carbonation phenomenon occurs.
- Commercial solutions are chemically not stable, suffering degradation.
- Due to its alkalinity, it reacts violently with acids and it is very corrosive.

Calcium hypochlorite is mainly used in certain areas such as North Africa due to the high variation of temperatures, which affect the stability of sodium hypochlorite and limiting the use of chlorine gas. This dry chlorine technology is used to produce an onsite solution of hypochlorite. The negative points are mainly the high cost of the reagent and the high maintenance of the equipment for its production.

Methodology and doses

The selected chlorination process for DESSIN has been the sodium hypochlorite solution. It is less dangerous than chlorine gas and a faster and cheaper option than calcium hypochlorite. One of the main problems of chlorination is the generation of trihalomethanes caused by the reaction between residual chlorine and the organic matter present in water. In order to minimize the trihalomethanes production it should be used the minim dose necessary to achieve the requested disinfection. The determination of this concentration is obtained using the Break Point method. Figure 20 shows the evolution of free chlorine when added to water.

While chlorine is added at the beginning of the chlorination there is a growth of the residual chlorine. At this stage, chlorine reacts with the free ammonia present in water, generating chloramines. After the first peak of residual chlorine, the addition of more chlorine creates dichloramines and trichloramines. At one point, the minimum in Figure 20, there is no more free ammonia available and then all the chlorine added remains in water, attacking microorganisms and pathogens. This point is what is called Break Point.

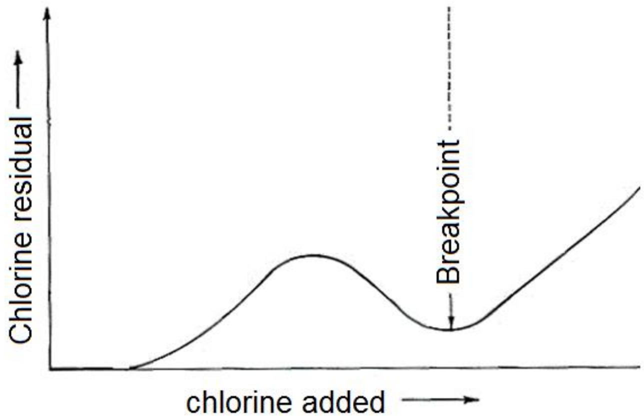



Figure 20: Break Point curve. Added chlorine vs. residual chlorine

To determine which concentration represents the break point, there is a methodology that is commonly utilized. Known ascendant concentrations of free chlorine are added to the process water. After fixing pH with a sulphates buffer solution and adding a reagent that indicates the presence of free chlorine (Syringaldazine) the solution is mixed for a few minutes. Agitation is then stopped and after this the colour of the solution is checked. The first concentration that makes changes the colour of water to pink will be the break point, the minimum concentration that brings free chlorine to water. Figure 21 illustrates this test; the second beaker would be the break point, as it is the first concentration that shows colour:

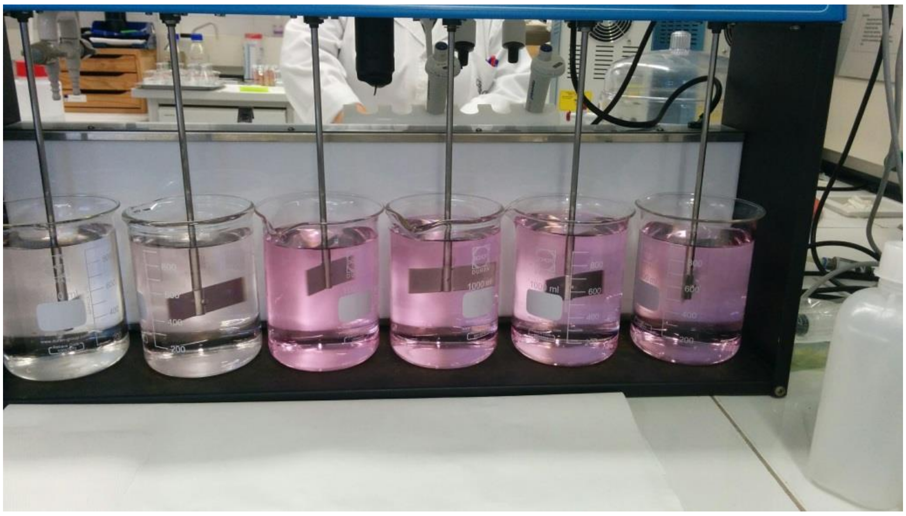


Figure 21: Break Point determination analysis

The break point determination for the SJD DWTP sand filtered water was done in different times with a result of a 1 mg/L of chlorine dose. In Figure 22 (left) can be seen the different doses applied in chlorination experiments and the residual chlorine measured, showing also that from 1mg/L of dose there is a clear increase of residual chlorine confirming this dose as a break point. Figure 17 (right) shows Ct doses around the referenced applied in the experiment.

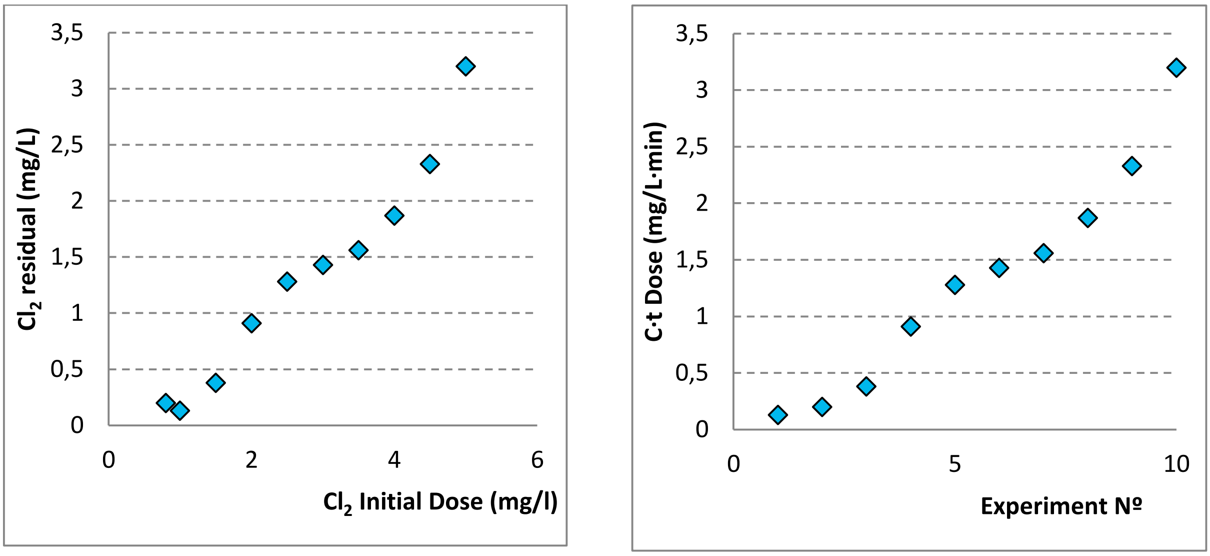



**Figure 22: Initial Cl2 dose vs residual dose (left) and C·t doses in the experiment (right)**

### 4.1.2 Chlorine dioxide: Methodology and experimental doses

Overview

Chlorine dioxide, chemical formula ClO2, is a red-yellow gas (depending on the concentration) with a very high oxidizing power. Because of its high reactivity, it can only be produced in the place where it will be used ("in situ"). In practice, generation by a chlorite-chlorine or chlorite-acid reaction is generally used. As chlorine dioxide is highly soluble in water, the generator allows for its dissolution to produce a concentrated aqueous solution. This solution, characterized by a green-yellow color, is then stored and dosed in the water to be treated.

The theoretical expression of the overall chlorine-chlorite reaction is:

$Cl_2 + 2\ ClO_2 \rightarrow 2\ ClO_2 + 2\ Cl^-$

The theoretical expression of the overall acid-chlorite reaction is:

5 NaClO2 + 4 HCl → 4 ClO2 + 5 NaCl + 2 H2O

Methodology and doses

Due to its high instability and the possible violent decomposition when separated from diluting substances, a sample of this reagent is directly taken from the one utilized in SJD DWTP. The concentration of this product is 1.5 g/L. It cannot be stored, as it decomposes in approximately 24 hours. Based on literature references, it had been done disinfection experiments with doses between 0.3 and 4.5 mg/L that produced residual chlorine between 0.08 and 2.36 mg/L (Figure 23 left). Then, the C·t doses for the different experiments done were between 0.08 and 2.36 mg/L·min Figure 23 right).

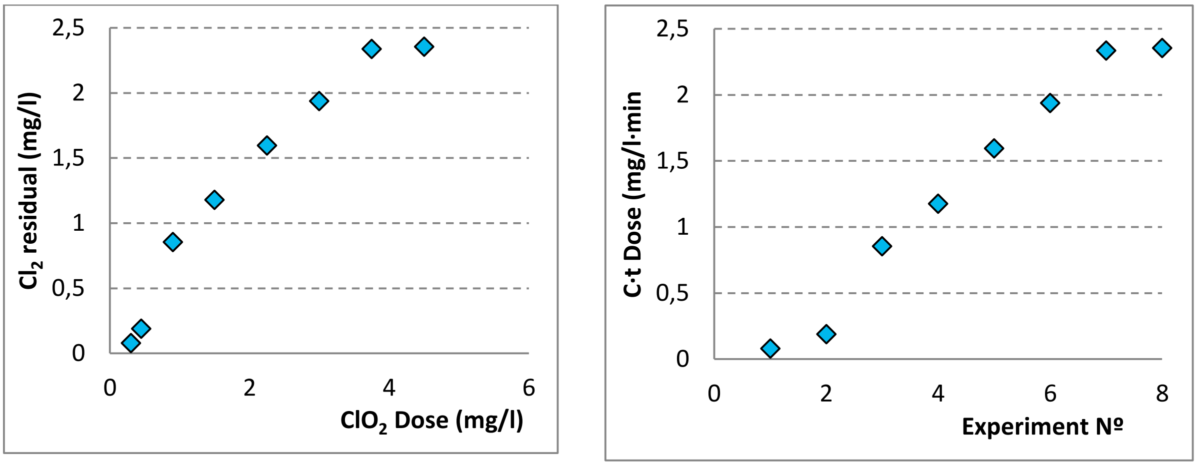



Figure 23: Initial $ClO_2$ dose vs residual dose (left) and C·t doses in the experiment (right)

### 4.1.3 Ozonation

Overview

Ozone oxidizes and thus modifies, but does not completely eliminate, a large number of compounds present in water. The molecules resulting from this oxidation may come from bacterial regrowth in the distribution system. In order to eliminate them and thereby guarantee the biological stability of the water, biological filtration is often necessary after oxidation.

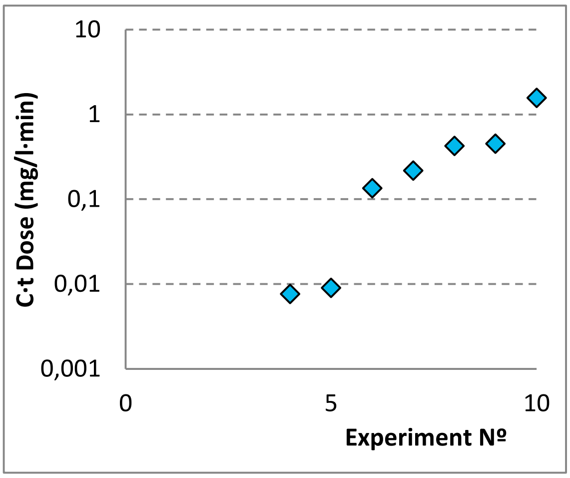
The stability of dissolved ozone decreases with increasing pH and temperature. At 15°C and a pH of 7.6 the lifetime of the residual is reported to be in the order of 40 minutes, but at higher temperatures it can be as low as 10 - 20 minutes. This occurs due to a decrease in the efficiency of transfer of ozone into water as temperature increases. Dissolved ozone can react directly or indirectly with the water into which it is dosed. Direct reactions occur with the ozone molecule. Indirect reactions occur with hydroxyl radicals that are formed when molecular ozone decomposes in water. In practice, reactions by both mechanisms are likely to occur in parallel, with the prevailing water quality influencing the extent to which hydroxyl radicals are formed.

Methodology and doses

The disinfection with ozone has been performed in a pilot plant that has an ozone generator (Figure 24). Its disinfection capacity has been tested combined with a sand filter. Water is pumped from a tank to fill de column. After this, the inlet valve is closed and water recirculates in the system. Meanwhile, ozone is created in a generator from the conversion of oxygen and is bubbled from the bottom of the column. When the contact time is the desired, the ozone generator is stopped and a sample of treated water is taken. The residence time in the column is determined by the C·t parameter desired.

Figure 24: Ozone disinfection pilot

Figure 25 (left) shows the range of doses applied in ozonation experiments and the residual ozone measured, showing that from 5 mg/L of dose there is residual ozone that increases linearly when increased initial dose. Figure 25 (right) shows the C·t doses used in 7 of the 10 different experiments, for the other 3 it was not measured any residual ozone so the C·t dose it is considered 0.



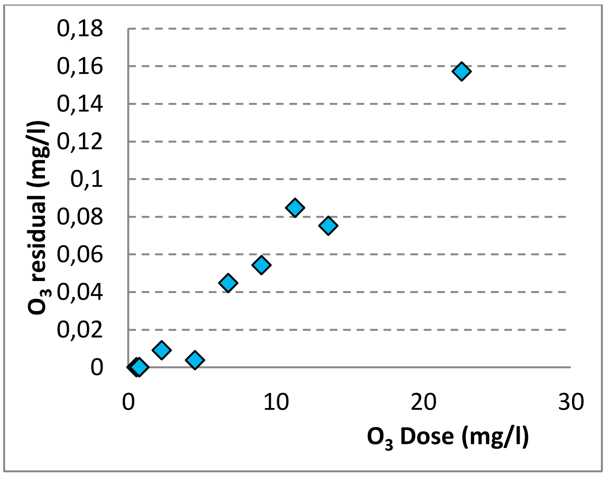




Figure 25: Ozone dose vs ozone residual (Left) and C·t doses in the experiment (Right)

As it was explained in the previous deliverable D22.4 (a), as a part of advance treatment the DWTP of SJD have an ozonation system after the sand filters. So it is very worthy to analyse the historical operation parameters and disinfection efficiency in comparison with experiments done specially for DESSIN project. It has been analysed SJD DWTP historical data of doses during one year period (between 2014-2015) and historical data of microbiology of 4 years (2010 to 2014).

The average ozone dose is around 2,2 mg/L and as it can be seen in Figure 26 the ozone dose is in 85% of samples less than 4 mg/L and around 12% is from 4 to 6 mg/L. The microbial load of sand filtered water and ozonation water can be seen in Figure 27 Sand filtered water has a 14% of samples without detection of total coliforms meanwhile after ozonation this value increase until 47% of the samples. In average the sand filtered water passed from 2.3 logs to 1.5 logs of total coliforms after ozone disinfection. Regarding *Clostridium perfingens*, sand filtered water passed from 15 to 52% of no detection samples after the ozone disinfection, and average log values decreased from 1.6 to 0.6 logs.

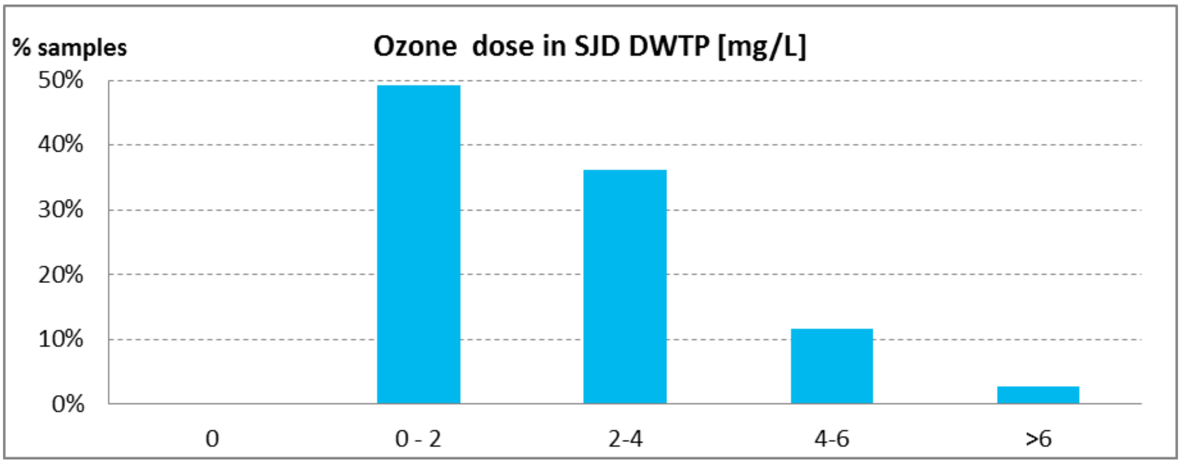



**Figure 26: Ozone treatment data in SJD DWTP. Doses applied**

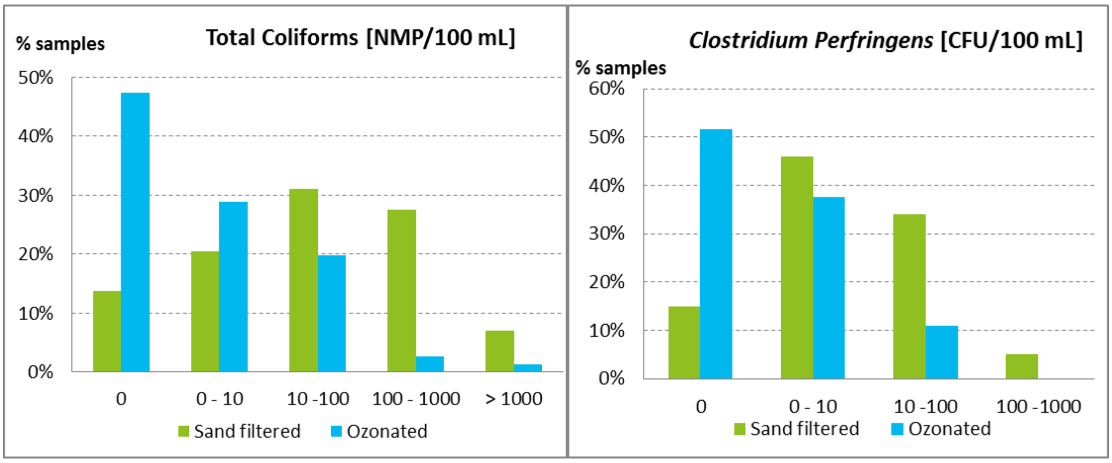



**Figure 27: *C. perfringens* and total coliforms of SJD DWTP sand filtered and ozonated water**

### 4.1.4 UV disinfection

Overview

UV dose is typically expressed in units of $mJ/cm^2$ or $J/m^2$ and is a function of UV intensity (or fluence rate), $mW/cm^2$, and exposure time, s. ($1\ mWs/cm^2 = 1\ mJ/cm^2$). UV disinfection systems market equipment are capable of applying a specified dose over a defined range of operating conditions (i.e. flow rate, water quality) and are validated to inactivate bacteria, protozoan pathogens such as *Cryptosporidium*, and viruses.

Methodology and doses

The disinfection with UV it was performed in a pilot UV reactor (Figure 28). This reactor has a lamp irradiating at 254 nm. Water recirculates in a column and passes through the reactor. Time is counted and when I·t dose is reached the recirculation stops and the lamp is disconnected. Irradiation is constantly measured with a detector so that the calculation of dose can be accurately performed.

**Figure 28: UV Reactor used in the experiment**

If UV is being installed for primary disinfection, the dose must achieve adequate inactivation of a range of pathogens. The Austrian ONORM standards, which apply to UV equipment intended for the disinfection of potable water, justify the stipulated dose of 400 $J/m^2$ on the grounds that it assures “a 6-log-reduction of health-related water transmittable bacteria, and a 4-log-reduction of health-related water transmittable viruses according to the state of the art”. According to this and other literature references, for the disinfection experiments it had been used doses from 100 to 2000 $J/m^2$ as it can be seen in Figure 29.

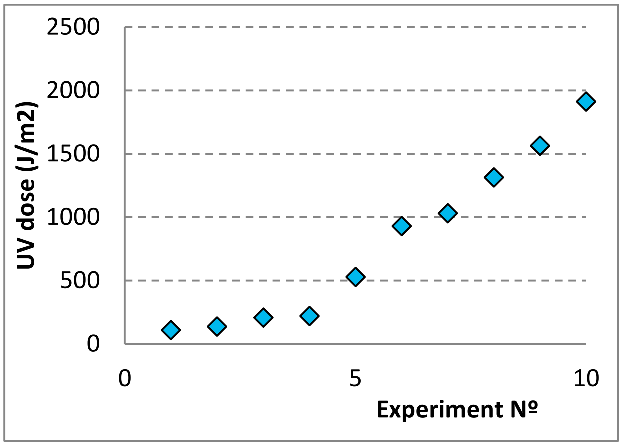



Figure 29: UV doses used in disinfection experiments

## 4.2 Results of disinfection methods

### 4.2.1 Chlorination: Assessment of results

In order to evaluate the disinfection efficiency of different methods, it is taken into account the microbial load present in disinfected sand filtered water and also it is calculated the log removal of the microbial load between before and after the disinfection.

It is important to note that in microbial quantification analysis , the meaning for the analitic result values are:

- Zero means no presence of the microorganism
- From 1 to 3 mean just presence of the microorganism
- From 4 to 20 is considered an estimated value
- More than 20 is considered a real value

Therefore the results should be analysed taking into account these orders of magnitude.

Figure 30 (left) shows chlorine disinfection results in a complete removal of total coliforms from doses of 0.4 mg/L·min (there is only one point out of the trend at 2.33 mg/L·min that does not have a total removal). Regarding *C. perfringens* until doses of around 2 mg/L·min it could not be seen a clear removal efficiency

Figure 30 (right). From that doses the disinfected water it has only presence (between 1 and 2) of *C. perfringens* and at the maximum dose of 3.2 mg/L·min there is a total removal.

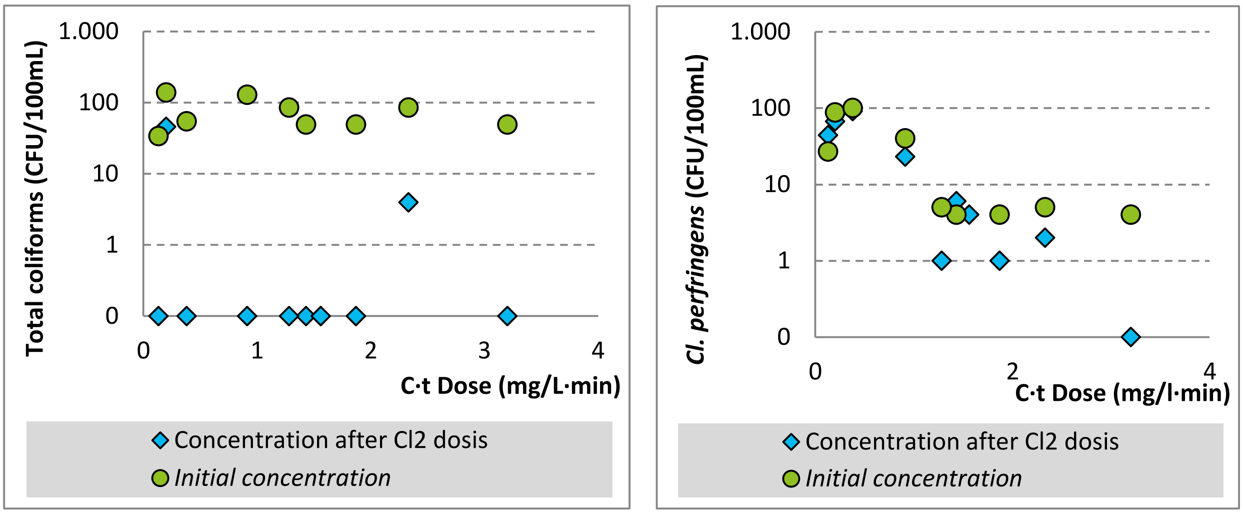



**Figure 30: Total coliform and *C. perfringens* load before and after the $Cl_2$ disinfection for each Ct dose**

Literature reports tables of microbial removal for different doses (CIRSEE, 2009 and EPA, 2011). It is important to note that the log removal of the experiments is restricted by the amount of microbes present in the original water. Therefore if sand filtered water contained 100 CUF/100mL of total coliforms the maximum log removal will be 2, and if it has 10 CUF/100 mL of *C. perfringens,* the maximum reduction assessed would be 1. Figure 31 compares literature and experimental values of log removal and doses. For coliforms, literature gives 2 logs removal from 0.02 mg/L·min and in the experiments it has been found from 0.38 mg/L·min. For bacteria, literature gives values of 2 log removal from 3.3 mg/L·min and at that dose it has been detected total removal in *C. perfringens*.

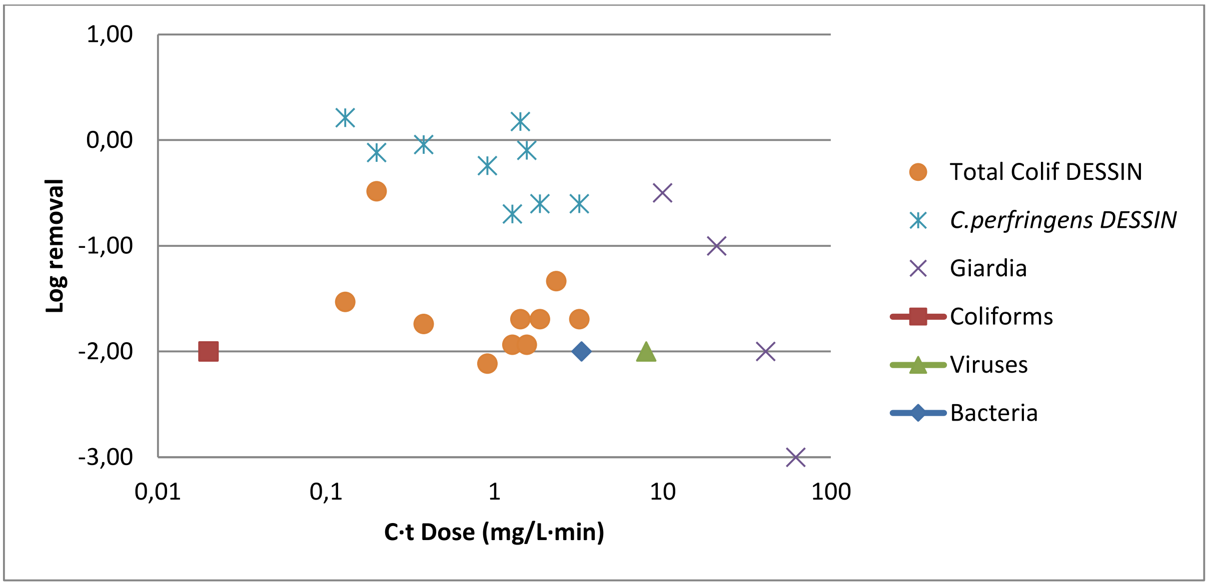



**Figure 31: Literature values of microbial log removal depending on chlorine C·t dose compared to DESSIN chlorine disinfection experiments**

### 4.2.2 Chlorination: Evaluation of by-products formation (THMs)

To date the major organochlorine by-products of concern have been the four chlorinated compounds, known collectively as the trihalomethanes (THMs): bromoform (tribromomethane), dibromochloromethane, bromodichloromethane and chloroform (trichloromethane). European Communities Drinking Water Regulation (SI 278 of 2007) stipulates a maximum of 100 µg/L total THMs at the consumers tap, which is a widespread standard in individual member states. The concentrations of THM compounds produced by chlorination are a function of pH, temperature, free chlorine concentration, contact time, bromide and concentration and nature of oxidisable organic material in the water.

Figure 32 shows the formation of THMs in the different experiments for the different doses. It can be observed a maximum of 120 µg/L of THMs in a dose of 1.4 mg/L of chlorine. These values are in the range of drinking water standards so it should not be any sanitary or environmental problem the injection of this disinfected water. Moreover THMs concentrations in ASR systems could be eliminated over a few weeks, primarily due to anaerobic microbial reactions promoted in the aquifer. Even though, during ASR operation it will be neccesary to plan regular water sampling campaigns (pre/post water injection) in order to analyse the evolution of these organochlorine by-products in the aquifer.

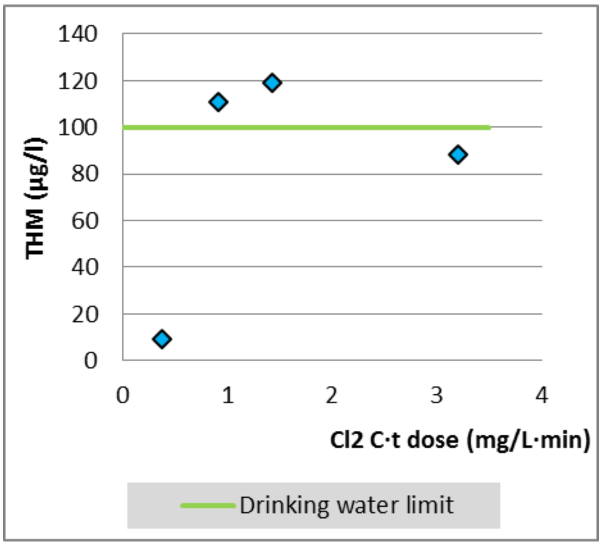



Figure 32: THMs formation for increasing $Cl_2$ doses

### 4.2.3 Chlorine dioxide: Assessment of results

As in chlorine disinfection experiments, it has been plotted the microbial load before and after disinfection of sand filtered water and also the log removal. Figure 33 (left) shows that the chlorine dioxide disinfection results with a removal until 1 CFU/100mL or total elimination of total coliforms from doses of 0.2 mg/L·min. Figure 33 (right) can be seen also in a qualitative perspective, as *C.*

*perfringens* concentration in SF water was extremely low (below 10 CFU/100 mL). This low range of concentration in initial conditions (SF water) does not allow evaluating the disinfectant capacity in terms of logarithmic decay.

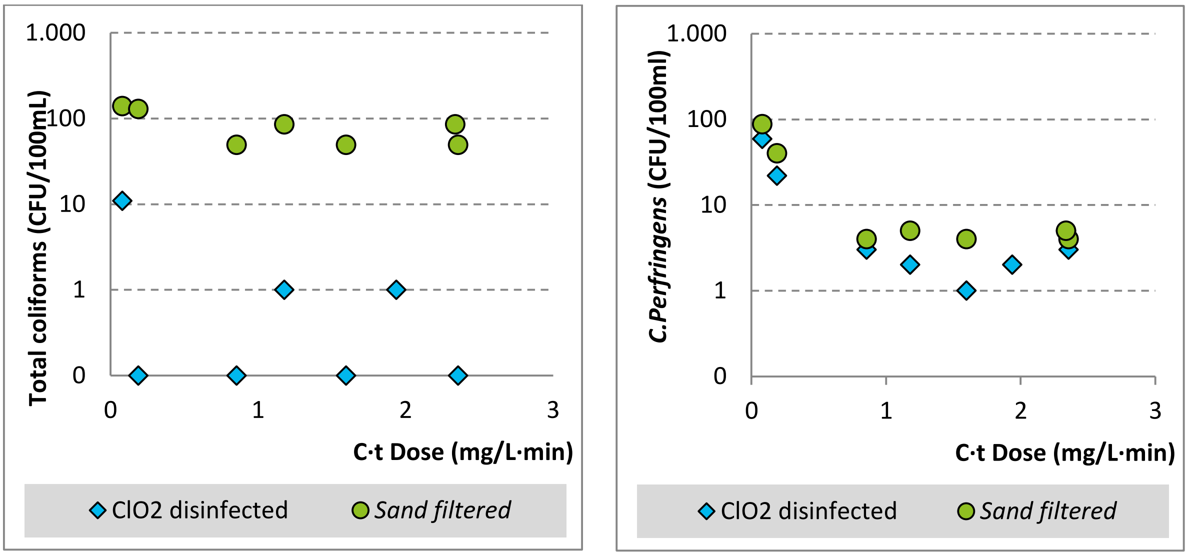



**Figure 33: Total coliforms (left) and *C. perfringens* (right) load before and after chlorine dioxide disinfection for each Ct dose**

In Figure 34 can be seen the comparison of log removal between literature data and disinfection experiments. Data from total coliforms appears with the same tendency in the experiments and literature, having a 2 log removal from doses of 0.3 mg/L·min. *C.perfringens* as the initial load of high doses was around 1 load, it can't be seen a bigger log removal than 0.6.

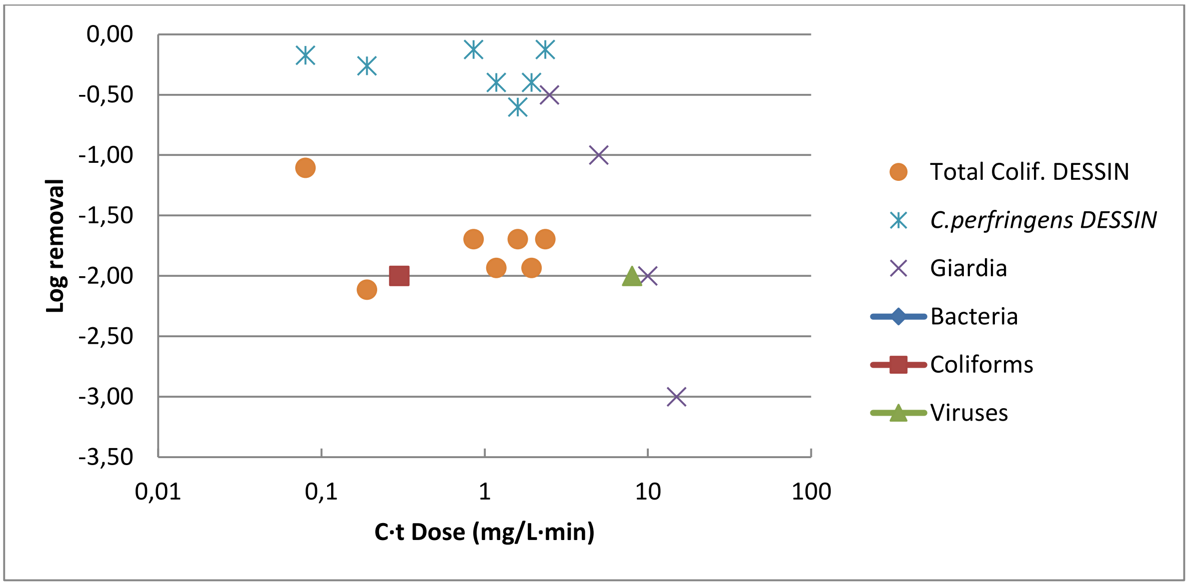
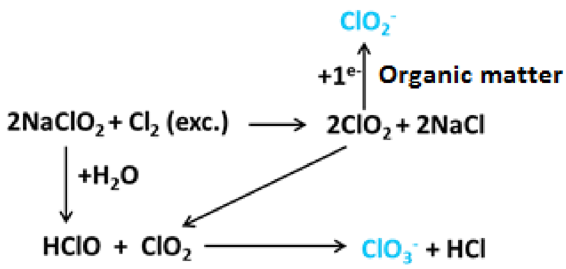



Figure 34: Literature values (EPA 2011, CIRSEE 2009) of microbial log removal depending on chlorine dioxide C·t dose compared with DESSIN chlorine disinfection experiments

### 4.2.4 Chlorine dioxide: By-product formation

The major chlorine dioxide by-products of concern are chlorite and chlorate. Chlorine dioxide reacts generallyas an electron acceptor, and hydrogen atoms present inactivated organic C–H or N–H structures are thereby not substituted by chlorine (Hoigne&Bader, 1994). Moreover, in contrast to chlorine, chlorine dioxide's efficiency for disinfection does not vary with pH or in the presence of ammonia, and it does not oxidize bromide. As opposed to chlorine, which reacts via oxidation and electrophilic substitution, chlorine dioxide reacts only by oxidation; this explains why it does not produce organochlorine compounds.

Chlorine dioxide is generally produced by reacting aqueous (sodium) chlorite with chlorine. However, under conditions of low initial reactant concentrations or in the presence of excess chlorine,the reactant produces chlorate ion. This reaction scenario is common in generators that overchlorinate to achieve high reaction yields based on chlorite ion consumption. Chlorite ion is also produced when chlorine dioxide reacts with organics matter.




Regarding regulatory limits for chlorites and chlorates, WHO have set a provisional guideline value of 0.7 mg/L for both chlorate and chlorite individually, based on health considerations. The US EPA has a maximum contaminant level (MCL) of 1 mg/L for chlorite at plants using $ClO_2$ and a maximum residual disinfection level (MRDL) of 0.8 mg/L for $ClO_2$. They recommend a maximum dose of 1.4 mg/L chlorine dioxide to maintain chlorite below the MCL, on the basis that 70% of the chlorine dioxide could be converted to chlorite. Typical dosages of chlorine dioxide used as a disinfectant in drinking water treatment range from 0.07 to 2.0 mg/L (EPA, 2011).

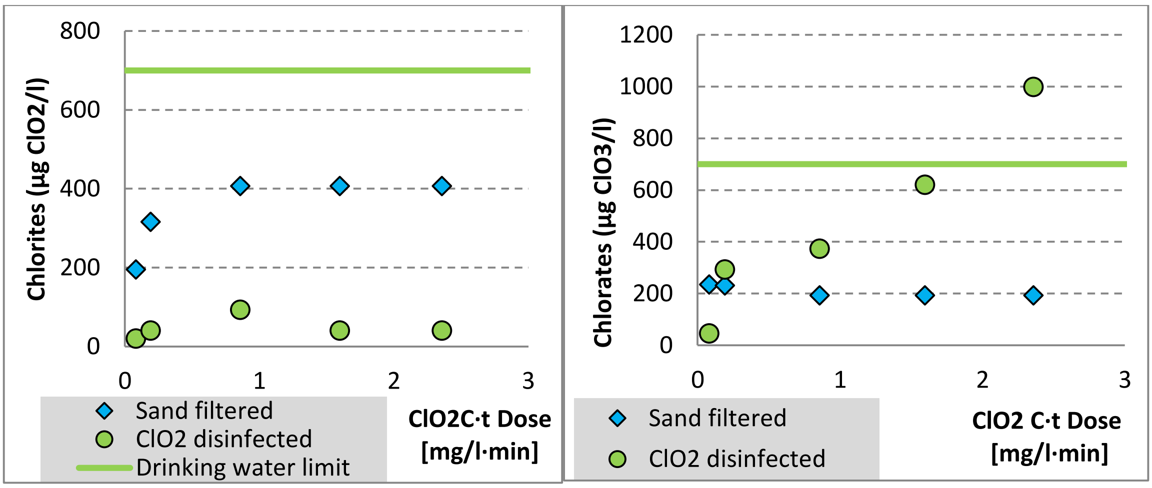



Figure 35: Chlorates and Chlorites formation using $ClO_2$ in increasing doses applied

In the Figure 35 can be seen the chlorates and chlorites formation in the chlorine dioxide disinfection experiments. The chlorates formation increase with the increasing doses and around 1.6 mg/L·min arrive near drinking water limit level. Whereas chlorites seems to decrease when applying chlorine dioxide  to the sand fitered water, probably forming chlorates.

### 4.2.5 Ozonation:  Assessment of results

In order to be able to compare with SJD DWTP data where there is only available values of doses in mg/L, in the following figures it is represented in that unit instead of mg/L·min. In Figure 36 and Figure 37 can be seen the microbial load of ozone experiments. For doses bigger than 4.5 mg/L there is a total elimination of coliforms and *C.perfringens*. Figure 38 represents same data presented as log removal calculation.

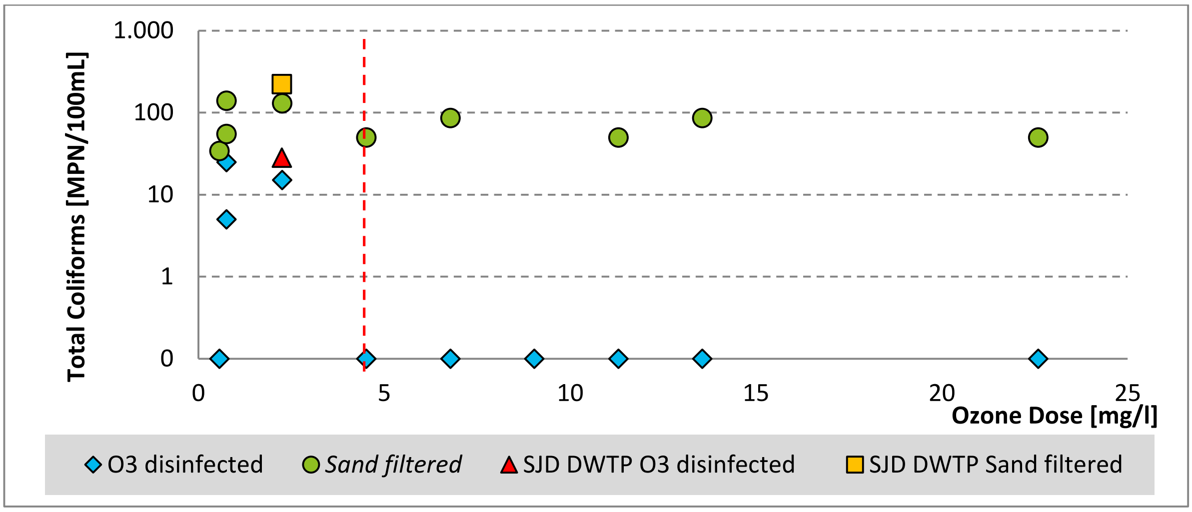



**Figure 36: Total coliforms load before and after ozone disinfection (experimental and full-scale plant)**

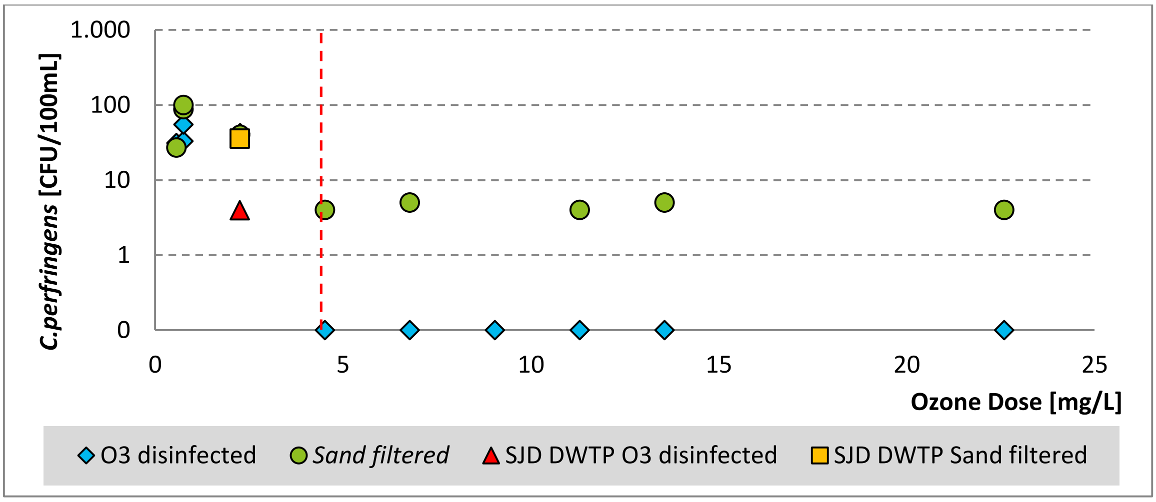



**Figure 37: *C. perfringens* load before and after ozone disinfection (experimental and full-scale plant)**

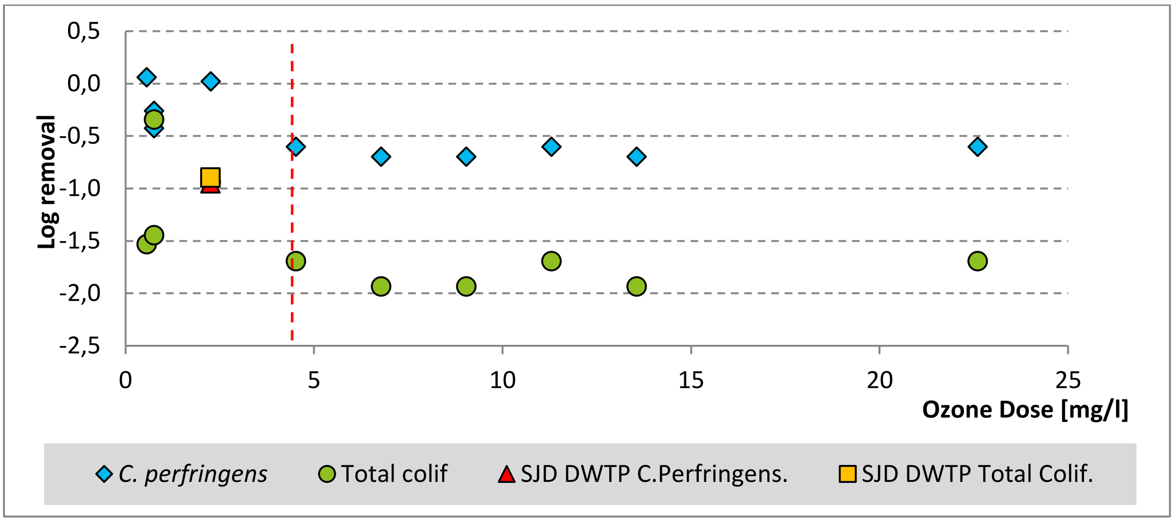



**Figure 38: Microbial log removal for ozone disinfection (experimental and full-scale plant)**

### 4.2.6 Ozonation: By-product formation

In the presence of bromides in water on which ozonation is carried out, there is a risk of bromate formation, which are carcinogens (quality limit according to European Directive 98/83/EC of November 3, 1998: bromates = 10 µg/L). Two reaction mechanisms (molecular and radical) are involved in the formation of bromates. The reaction with the bromide ion is considered to be slow.

To limit the formation of bromates, the following parameters must be monitored: bromide concentration of the water to be ozonated, C·t dose, and design of the reactor (two distinct zones recommended, diffusion zone and contact zone).It is recommended that pH values are less than or equal to 7.5.

As shown in Figure 39 there is a clear increase of the bromates production with the increasing dose. With a dose of 2.3 mg/L of ozone there is bromates formation below the drinking water limit of 10 µg $BrO_3$/L, meanwhile with a dose of 4.5 mg/L of ozone it is exceeded the drinking water limit. It is just at this dose when the removal of microbial load reaches its maximum level (Figs. 36, 37 and 38). In this regard and taking into account the analytical results presented at this chapter, it is not possible to maintain this removal capacity and, simultaneously, to fulfill with the legal threshold value (Fig. 39).

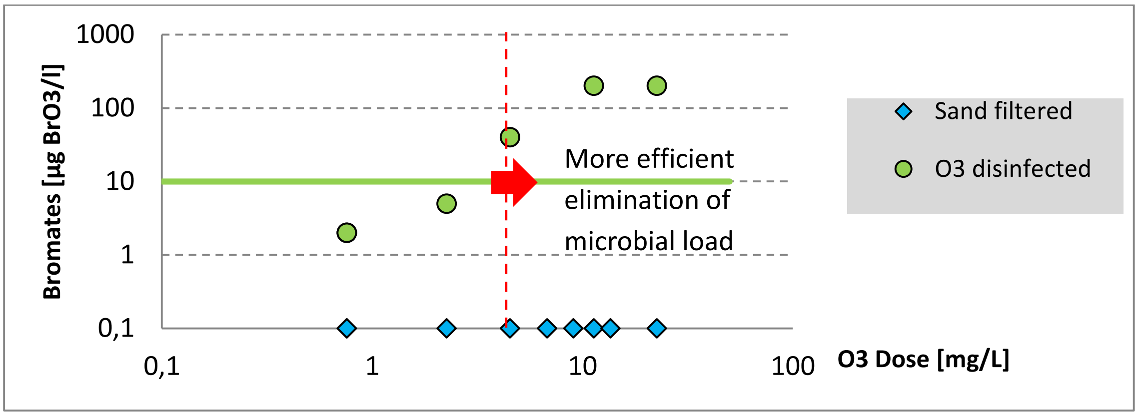



Figure 39: Bromates production in ozone disinfection experiments

### 4.2.7 UV: Assessment of results

Figure 40 shows the results of UV disinfection on total coliforms and *C. perfringens*. It can be observed that from a dose of 526 J/m² there is a total removal of both parameters. Figure 41 shows same data but represented as microbial log removal and also with a total coliforms log removal value from literature, which appears to be on the same tendency than the experiments.

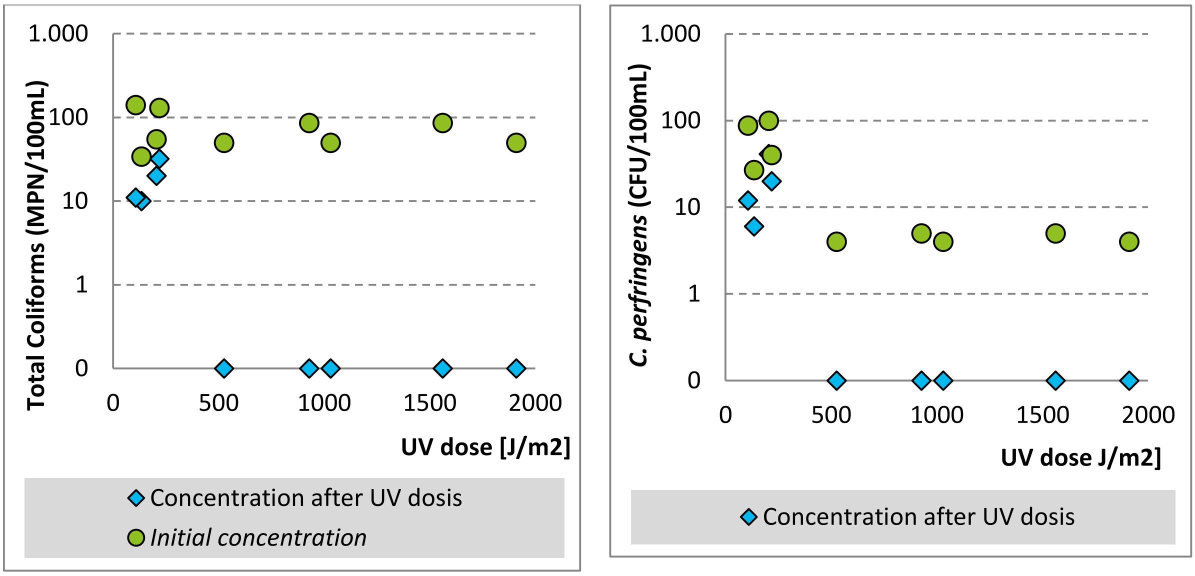



Figure 40: Total coliforms and *C. perfringens* load before and after UV disinfection

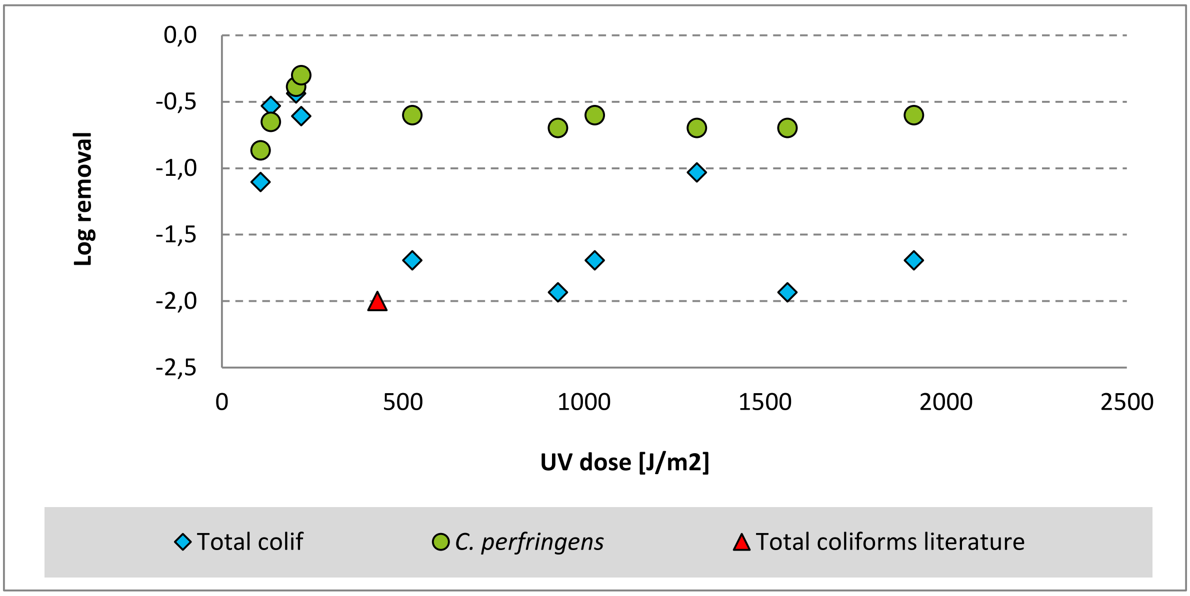



**Figure 41: Microbial log removal for UV disinfection experiments and literature total coliforms reference value (CIRSEE 2009)**

### 4.2.8 By-product formation using UV: none

One advantage for the UV disinfection is that there is not formation of standard by-products as the other methods previously presented.

## 4.3 Comparison and discussion of using disinfection methods for pre-potable water before ASR injection

As explained in section 3.3, the quality of sand filtered water should be enough to use it to inject in the Llobregat ASR System directly. Therefore the evaluation for the additional treatments has been done as a first experimental estimation in order to characterize the doses that could be applied to eliminate the microbial load. As it is not expected to implement this disinfection treatments, it was not done neither any cost estimation analysis, that could be at last the determining factor to choose one treatment or other.

In the

Table 4 it can be seen the advantages and drawbacks of each disinfection treatment method. In one hand, taking into account that injection water goes to a natural environment, one important advantage of UV disinfection is that is not using any chemicals and therefore would not add more species in the original river water. On the other hand for the sustainability of the process would be interesting the processes that does not need additional energy as chlorination and dioxichlorination. Finally, oxidation could be interesting in terms of removal of micropollutants, effect that is not studied in this project but that some other european public founding projects are studying.

**Table 4: Advantages and drawbacks of different disinfection methods**

| Disinfection method | Advantages | Drawbacks |
|---|---|---|
| Chlorination | · Simplicity of implementation<br>· Stable reagent | · Safety constraints<br>· THM's formation<br>· Use of chemicals |
| Dioxichlorination | · Avoids THM's formation<br>· High persistence | · Safety constraints<br>· Chlorates formation<br>· Use of chemicals<br>· Use of two reagents |
| Ozone | · Oxidation of micropollutants | · Bromates formation<br>· Ozone production<br>· Energy consumption |
| UV | · No by-products formation<br>· Low encumbrance<br>· No chemical reagent | · Energy consumption<br>· Reliability of equipment (UV lamps, UV sensors) |

Table 5 summarizes the recommended doses to be implemented for each disinfection treatment for the complete removal of microbial indicators and by-products formation estimates for the maximum dose.

**Table 5: Summary of doses for total elimination of selected indicators**

| Disinfection method | Total coliforms | | C. perfringens | | By products |
|---|---|---|---|---|---|
| Chlorination | Total elimination | 0.38 mg/L•min | Total elimination | 3.2 mg/L•min | 88 µg THM's/L |
| Dioxichlorination | Total elimination or 1 CFU/100 mL | 0.2 mg/L•min | 1 CFU/100 mL | 1.6 mg/L•min | 620 µg ClO3/L |
| Ozone | Total elimination | 4.5 mg/L | Total elimination | 4.5 mg/L | 40 µg BrO3/L |
| UV | Total elimination | 526 J/m² | Total elimination | 526 J/m² | None |

# 5 Conclusions

Although there was a difficulty in finding a parameter as an indicator of well clogging or bioclogging, it was accomplished the objective of validating the sand filtered water as the type of water to inject in the demonstration phase, by experimental evaluation in diferent fields. Some potential indicators reviewed in the deliverable D22.4(a) as the AOC determination were discarded because its analytical complexity and because it does not take into account any ASR system parameters as the well screen configuration or the aquifer transmissivity. Therefore the innovative way of simulating the ASR system in a column allowed a new way of evaluation like seeing directly the well screen evolution or the measurement of bioclogging by the EPS analysis. We think that the selection of EPS as bioclogging indicator might have repercussions in the ASR studies and Managed Aquifer Recharge in general since there are other research groups that are yet using it.

The main results in the experimental evaluation were:

- Maximum head loss of 20% after 75 days of continuous operation in the ARS simulation experiment. This increase in pressure does not mean a limitation in the aquifer capacity of infiltration of injection water( the gravels still are able to accept this injection flow), but an ascent in piezometric level would be expected in the well.

- Bioclogging formation evolution with a rapid increase in the first 140 days but with a stabilization and decrease in the following days, based on the EPS formation. Moreover the composition of the muddy sediment settled in well screen simulation was mainly inorganic but with a 11% organic.

- Bioclogging formation characterization by SEM photography and elemental composition determinations identified some isolated bacillus and hifas, while most of the ubiquitous material observed with the microscope corresponds to biological mass aggregates, presumably extracellular polysaccharides (EPS).

- Assessment of possible commercial disinfection methods as additional treatment has been done with limited conclusions. Microobiological load in injection water is relatively small to compare the effectivity of methods with literature references. Moreover, disinfection by-products appear in some of the tested methods (clorination, dioxichlorination and ozonation). This drawback questions the utility of disinfection methods before ASR, and ranks UV disinfection as the most advisable method if necessary.

Therefore, after working in lab and pilot scale it was finally decided to use the sand filtered water directly without any additional treatment as injection water for the following reasons:

- It is demonstrated that although clogging can appear, it is not hazardous in operational terms for the ASR Llobregat system.

- The sustainability of the ASR system increases if the energy consumption and the chemical reagents quantity for the treatment of the injected water are decreased.

- It is more economically efficient option.

- It minimizes the possible disinfection by-products introduction in the aquifer.

The pilot evaluation was done simulating an ARS well as realistic as possible, but the scale of a real system is very different and it has also lots of natural systems complexities. Then, after this evaluation, the next step of the DESSIN project in Llobregat site where there will be a sand filtered water injection in a real ASR well, will allow to confirm this deliverable conclusions.

Moreover, as some of the studied parameters can have also a longer term effect, the pilot column will still being operated meanwhile the demonstration phase is being performed.

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# D22.4 Evaluation of pre-potable water requirements for a safe injection in the Aquifer through ASR

## CHAPTER C: Regional and local numerical modeling to simulate the flow and conservative transport in the Llobregat demo site

CUADLL - Cetaqua, February (2015)

Revised version, November (2017)

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# D22.4 Evaluation of pre-potable water requirements for a safe injection in the Aquifer through ASR

CHAPTER C: REGIONAL AND LOCAL NUMERICAL MODELING TO SIMULATE THE FLOW AND CONSERVATIVE TRANSPORT IN THE LLOBREGAT DEMO SITE

**SUMMARY**

This report summarises results of the application of numerical modelling to the Llobregat ASR system. The work corresponds to the first phase of the project, focused on the impact assessment of ASR in terms of groundwater volume infiltrated in the aquifer and the improvements and/or impacts in groundwater quality. The work has been divided in two parts: (i) MODFLOW-based numerical model to simulate the impact on injected water in the local piezometric network installed for the project (4 $km^2$) (ii) VISUAL TRANSIN-based numerical model to simulate the impact of ASR and ASTR at regional scale (129 $km^2$).

Results of this report correspond to the simulations carried out of Scenario 1 (Demonstration scale of the project) and Scenario 2 (application of ASR in the full system). Results of Scenario 1 conclude that the demonstration phase of the project will have a local impact in the aquifer, as the mixing ratio between injected water and native groundwater will be below 10% after 1.4 km of aquifer passage. Local model and regional model have been key information for the establishment of local control network (Pz1, Pz2 and Pz3) and the selection of external control points in the aquifer (P10, P13 and P03) to verify the impact in groundwater quality during the demonstration phase.

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| **LEAD BENEFICIARY** | **DELIVERABLE AUTHOR(S)** |
| Cetaqua | Enric Queralt (CUADLL - Subcontractor)<br>Jordi Massana (CUADLL- Subcontractor)<br>Marta Hernández (Cetaqua) |
| **QUALITY ASSURANCE** | |
| Aigües de Barcelona (internal) | Jordi Martín Alonso, Oriol Gibert |
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| 31/08/2015 (M20) | 20/03/2015, revised version 09/11/2017 |
| **DISSEMINATION LEVEL** | X PU = Public<br>□ PP = Restricted to other programme participants<br>□ RE = Restricted to a group specified by the consortium. Please specify: ________________________________<br>□ CO = Confidential, only for members of the consortium |



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# List of Acronyms and Abbreviations

| | |
|---|---|
| AB | Aigües de Barcelona Drinking Water Supply company |
| ASR | Aquifer Storage and Recovery |
| ASTR | Aquifer Storage Transfer and Recovery |
| CUADLL | Association of groundwater users in the Llobregat aquifer |
| DWTP | Drinking Water Treatment Plant |
| ICC | Catalan Cartographic Institute (http://www.icc.cat/) |
| Pz1 | Piezometer number 1 drilled in SJD |
| Pz2 | Piezometer number 2 drilled in SJD |
| Pz3 | Piezometer number 3 already existing in SJD |
| RFD | Real Full Demonstration (simulation of demonstration phase of the project, injecting 50 L/s in P18, corresponding to 1.57 $Mm^3$/year) |
| SJD | Sant Joan Despí facility of Drinking Water Treatment Plant (Barcelona) |



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# Executive summary

This report summarises results of the application of numerical modelling to the Llobregat ASR system. The work corresponds to the first phase of the project, focused on the impact assessment of ASR in terms of groundwater volume infiltrated in the aquifer and the improvements and/or impacts in groundwater quality. To this end, numerical modelling offers an incomparable opportunity of estimating the arrival plume of the demonstration phase, and what is more, to evaluate future scenarios considering the application of ASR technology to the full system.

The work has been carried out by CUADLL (Association of Users of Llobregat Aquifer). This public entity uses the VISUAL TRANSIN model of the Llobregat area as a management tool, and they have been applied it in previous managed aquifer recharge projects.

The work has been divided in two parts: Local model simulates the changes occurring at local scale, using a mesh of 2 km x 2 km to assess the impact of injection in P18 in the local network of piezometers (Pz1, Pz2 and Pz3). Geological information gathered in drilling works and pumping tests allowed to create the conceptual model. The local numerical model has been developed using MODFLOW, and the main results are the estimation of the rise of groundwater level during the injection phase and the estimation of the arrival time of the injected water to Pz1, Pz2 and Pz3.

The second part consists in the modification of the existing regional model build in VISUAL TRANSIN code to simulate ASR under different scenarios. 4 scenarios have been stated, while the deliverable includes the results of two of them. The pending 2 scenarios will be incorporated later on in deliverables of WA3. Scenario 1 simulates the injection of 50L/s in P18 that will be carried out in the demo phase of DESSIN project. Scenario 2 simulates ASR in the existing facilities of Aigües de Barcelona, having an annual recharge in the aquifer of 5, 10 and 15 $Mm^3$, which correspond to historical values injected in the aquifer in the 90s. Scenario 3 and 4 are out of the scope of the demonstration phase of the project, and will simulate the injection and recovery of pre-potable water in different wells. This is the so called ASTR (Aquifer Storage Transfer and Recovery).

Results of Scenario 1 conclude that the demonstration phase of the project will have a local impact in the aquifer, as the mixing ratio between injected water and native groundwater will be below 10% after 1.4 km of aquifer passage.

Local model and regional model have been key information for the establishment of local control network (Pz1, Pz2 and Pz3) and the selection of external control points in the aquifer (P10, P13 and P03) to verify the impact in groundwater quality during the demonstration phase.

# 1. Introduction

This report aims the presentation of the numerical models to simulate and study the impact of managed aquifer recharge in the Llobregat aquifer using different approaches. The framework of this work is the DESSIN project "Demonstrate Ecosystem Services Enabling Innovation in the Water Sector". DESSIN demo site in Barcelona will study and test the flexibilisation of ASR system of Sant Joan Despí. Drinking Water Treatment Plant (SJD, DWTP). For this purpose, the last phase of the project will consist in a demo phase using an existing injection well located in the DWTP, operated by Aigües de Barcelona (AB). The well named P18 has been selected to carry out the demonstration phase of the project, with an injection flow of 50 L/s during at least one year of continuous operation. More information about the hydrogeology and the history of this ASR system can be found in DESSIN deliverables: D22.4 (a) and D35.1(a).

Numerical modelling is a useful tool in hydrogeology to simulate groundwater flow in the aquifer and predict impact of extraction and injection regimes. The main constraint of using numerical models is the uncertainty of the aquifer configuration and hydrogeological parameters governing water transport. The knowledge of the aquifer is limited to the observation points (geological profiles obtained normally for the recuperation of geological sediments in well excavations), historical pumping tests performed in wells and boreholes, or geophysical techniques applied in the surface. In the specific case of the demonstration site in Barcelona, the Llobregat aquifer in the Lower Valley has been constantly studied, due the large tradition of hydrogeologists in the area. Nonetheless, the small scale of the injection in P18 represents a challenge. Along the project, hydrogeologists will deal with local scale for the interpretation of changes occurring in the observation network, and regional scale to simulate the impact of ASR system operating at full-scale (this is out of the scope of the demonstration phase, but should be assessed theoretically as well).

To fulfil all the needs of the project, two different numerical models have been developed:

- **Local model:** has been developed using MODFLOW. The objective is the simulation of groundwater level variations caused by the injection in P18 of 50 L/s. This injection flow corresponds to the demonstration phase that will be carried out along the project, so it is expected to have two phases in the model development: (i) simulation and (ii) calibration with real data.

- **Regional model:** has been developed using VISUAL TRANSIN. The existing numerical model of "Vall Baixa" (Lower Valley) and Llobregat Delta has been adapted to simulate the impact of ASR operation not only at project scale, but also at full scale.

In parallel, the following scenarios have been described:

- **SCENARIO 1: ASR Demonstration scale:** corresponds to the demonstration phase of the DESSIN project, using a single well (named P18) to inject continuously 50 L/s. The surroundings of P18 have been equipped with an observation network consisting in 3 piezometers: Pz1, Pz2 and Pz3 that will notice the impact of the injected water.

- **SCENARIO 2: ASR Full scale:** The full ASR system includes 12 dual wells, with injection – extraction capabilities. The full scale consists in the simulation of all the system working in injection-extraction periods. This is out of the scope of the demonstrative phase of DESSIN project, but this assessment is quite important for the industrialisation of findings obtained in P18 as pilot test.

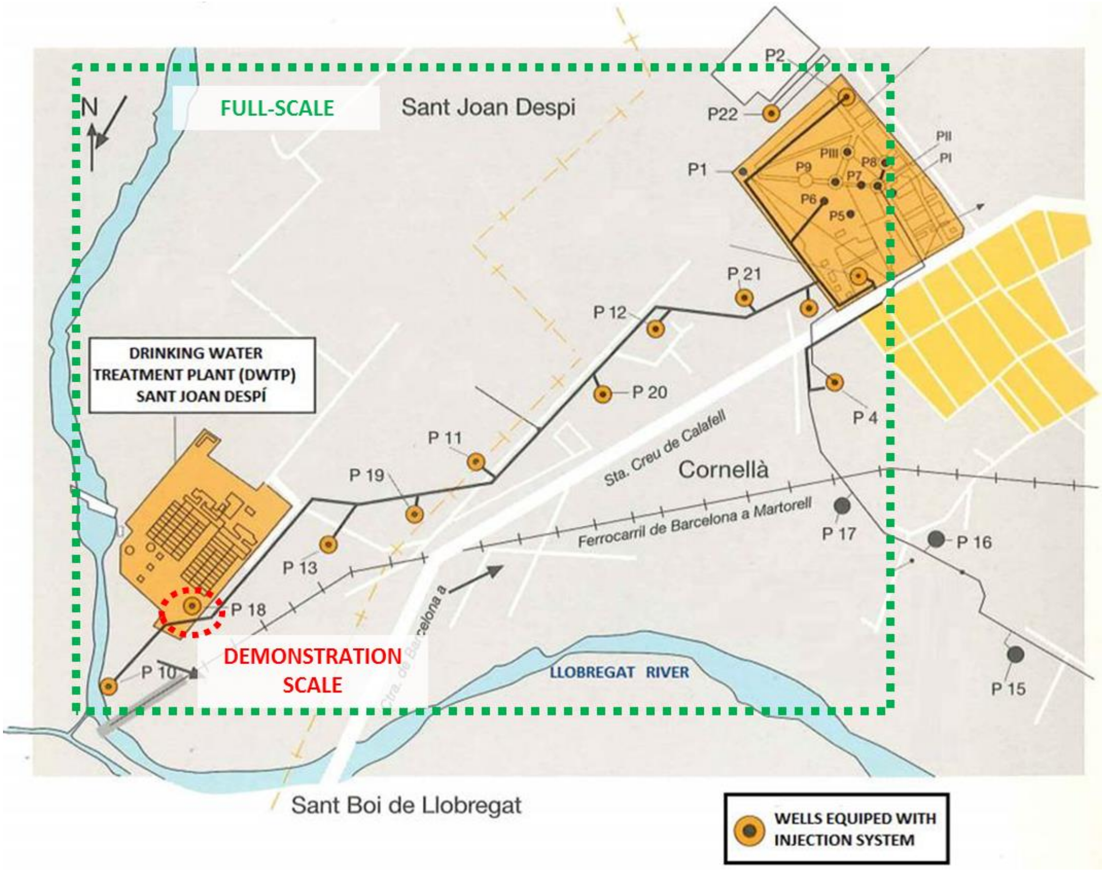


**Figure 1: Graphical representation of the scope of demonstration scale and full scale**

Moreover, taking advantage of the development of the regional model, additional scenarios will be tested as well. Both will consider the possibility of water injection and water recovery in different points, so travel time will play an important role. This variation of the ASR technique is commonly called ASTR "Aquifer Storage Transport and Recovery", and is widely applied is USA (Orange County, California). In fact, some regulations of ASR techniques establish travel times of more than 6 months or 1 kilometer of distance between injection and recovery.

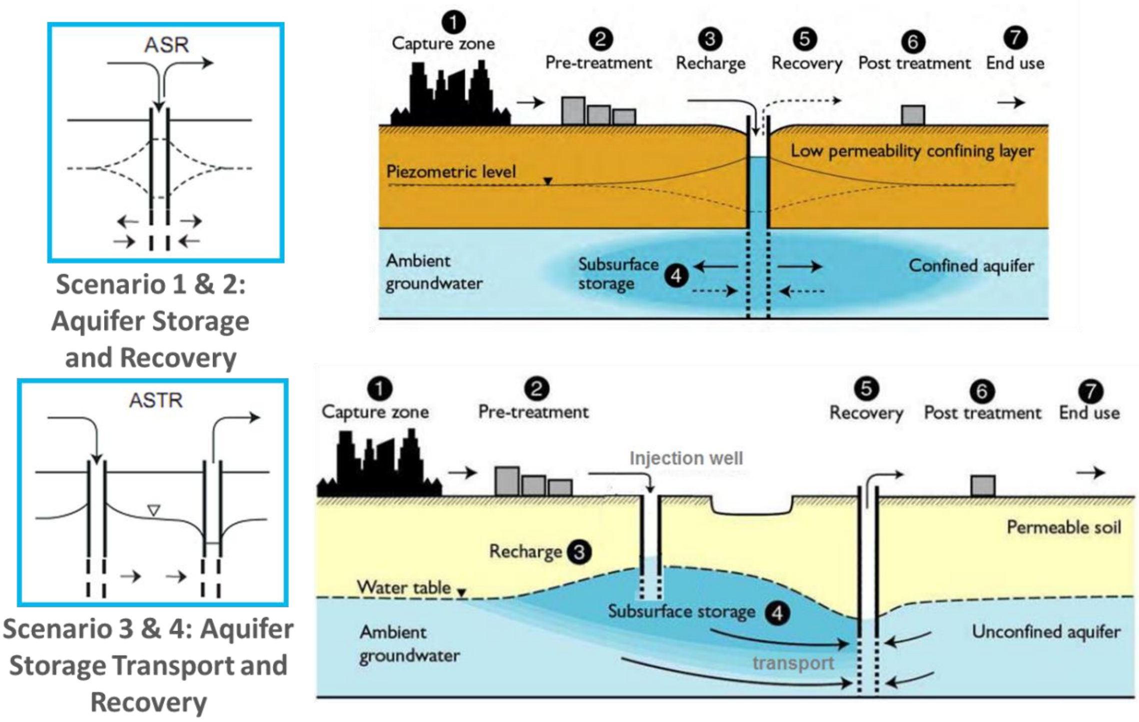


**Figure 2: Conceptual scheme of ASR and ASTR. (Source: )**

In the Llobregat case, AB as operator of ASR facility, showed interest in having theoretical estimations of potential changes introduced in the current system (see aerial view in Figure 3 showing the injection points and recovery points of both scenarios):

- **<u>SCENARIO 3</u>: ASTR Full Scale SJD – Central Cornellà**: this scheme consists in the locate the injection in a group of existing wells near to the DWTP facility, and use the wells of Central Cornellà as recovery point. This situation will allow the use of existing wells, and will improve travel time and residence time in the aquifer. The current limitation for the real application of this scenario is the existence of one single pipe connecting all the wells. Nowadays is not possible to inject and pump water simultaneously. Evidences of water quality improvement could help to propose a separate pipe in future investments.

- **<u>SCENARIO 4</u>: ASTR Full Scale Sant Feliu – Central Cornellà**: water reuse schemes are implemented in Spain, and especially in the Llobregat area. Aquifer recharge is one of the reclaimed water uses that is regulated by the national Degree RD1620/2007, but poorly applied in practise. This last numerical model scenario will simulate the hypothetical ASTR scheme, using reclaimed water produced in the WWTP of Sant Feliu del Llobregat, upstream the recovery point, Central Cornellà. Results will help to assess travel time in the aquifer and study potential changes occurring in water quality (literature based). Results could be a starting point of an additional research projects to deeper study the impacts of reclaimed water in the aquifer.

Scenario 3 and Scenario 4 are out of the scope of this deliverable, and will be prepared later on.

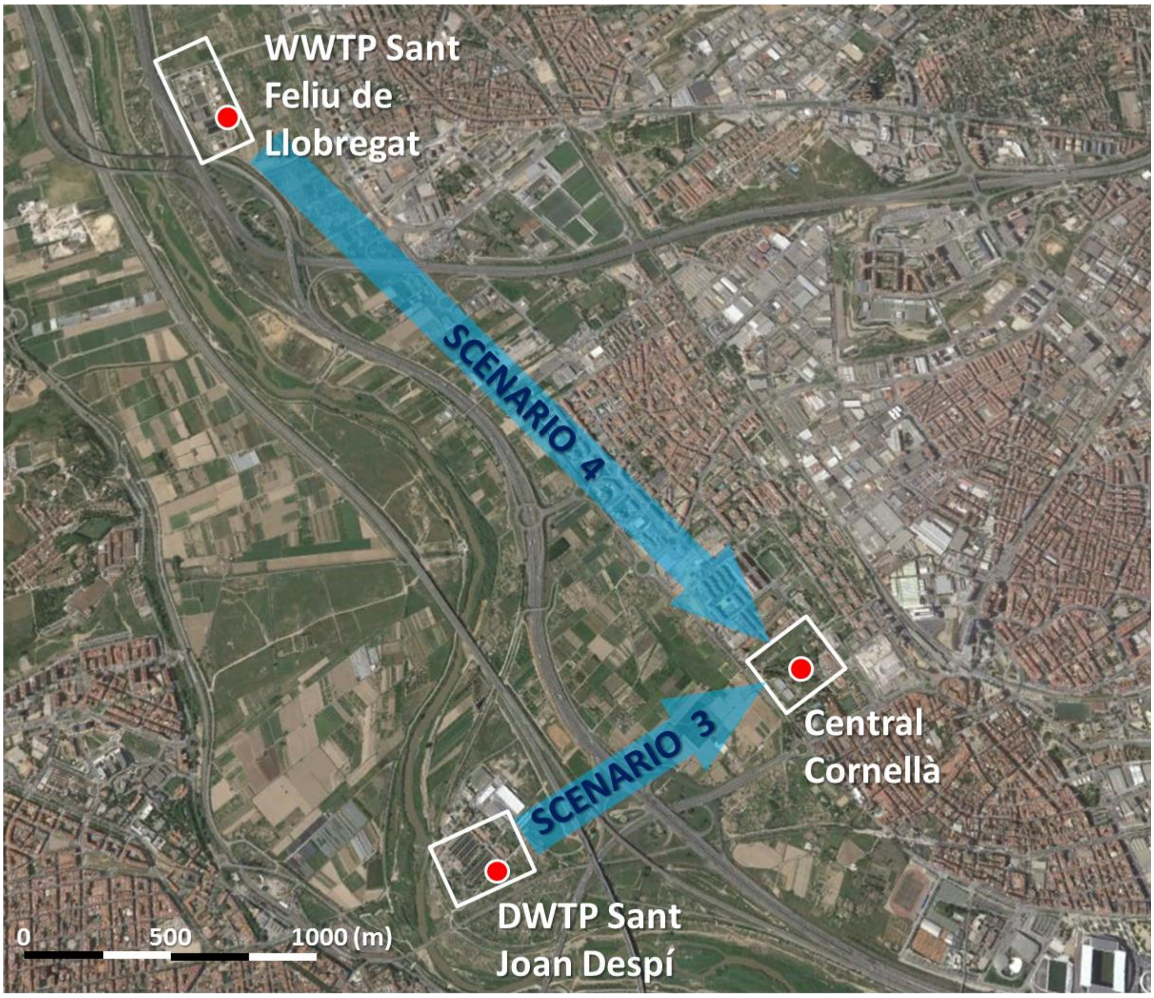


Figure 3: ASTR from Sant Joan Despí and Sant Feliu de Llobregat

Table 1 summarises the objectives of each of the numerical models developed. While local model developed with MODFLOW is specific to assess the impacts of ASR at demonstration scale, regional model is able to represent all the scenarios presented.

Table 1: Combinations of numerical models and scenarios

| | ASR Demo scale | ASR Full Scale | ASTR SJD –Cornellà | ASTR Sant Feliu - Cornellà |
|---|---|---|---|---|
| Local Model (MODFLOW) | YES | NO | NO | NO |
| Regional Model (VISUAL TRANSIN) | YES | YES | YES | YES |

# 2. Objectives

Specific objectives of the numerical modelling applied to the Llobregat demo site are:

- **Evaluate the impact of the application of ASR** at different levels:
  - DESSIN project demonstration scale: evaluate the impact in terms of water quality improvement and piezometric levels by the injection of 50 L/s in a maximum of 2 years of operation.
  - Full ASR capacity: assess the impact in terms of water quality improvement and piezometric levels by the injection of 5, 10 and 15 $Mm^3$/year, corresponding to the historical groundwater injection carried out by the full system.

- **Understand the response of the aquifer at local scale** by using a very detailed numerical model applied in the local monitoring network of the P18. This report presents the results of the simulation of the demonstration phase in the P18, and will be calibrated later on with results obtained in the field.

- **Provide technical support for the selection of the external control points** of the monitoring network of groundwater.

Scenario 3 and scenario 4 are out of the scope of this deliverable, and will be performed later.

# 3. Local model description

## 4.1 Main characteristics

The local model consists in mesh of 2 per 2 kilometers. The plane inclination angle is 325° with respect to the North. The model has the same amplitude than the alluvial aquifer (colored in yellow in Figure 4). The implementation of the studied area has been done by the interpolation and rotation of terrain digital model (originally from ICC with a cells of 5*5 m).

The grid is 25*25 m but the model will work in detail in the zone close to the well named P18 and piezometers of its observation network of Pz1, Pz2, Pz3[1]. Figure 4 indicates the location of the borders of the model, with the location of piezometers of reference in red points. Figure 5 shows the topography of the terrain in the studied area, corresponding to the left margin of the Llobregat River.

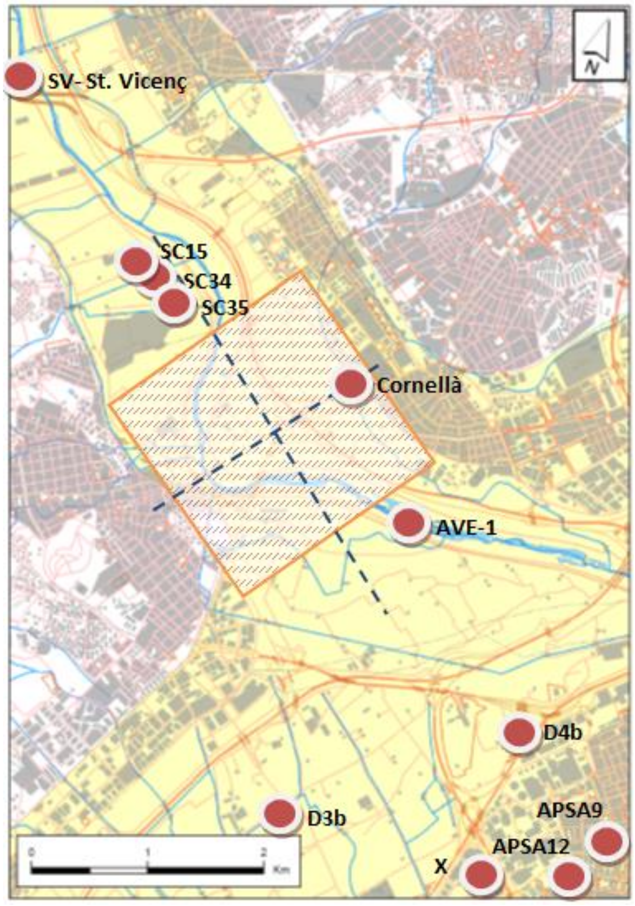


Figure 4: Location of model mesh at local scale

[1] Find more information about well P18 and the piezometer network configuration in Deliverable D35.1(a)

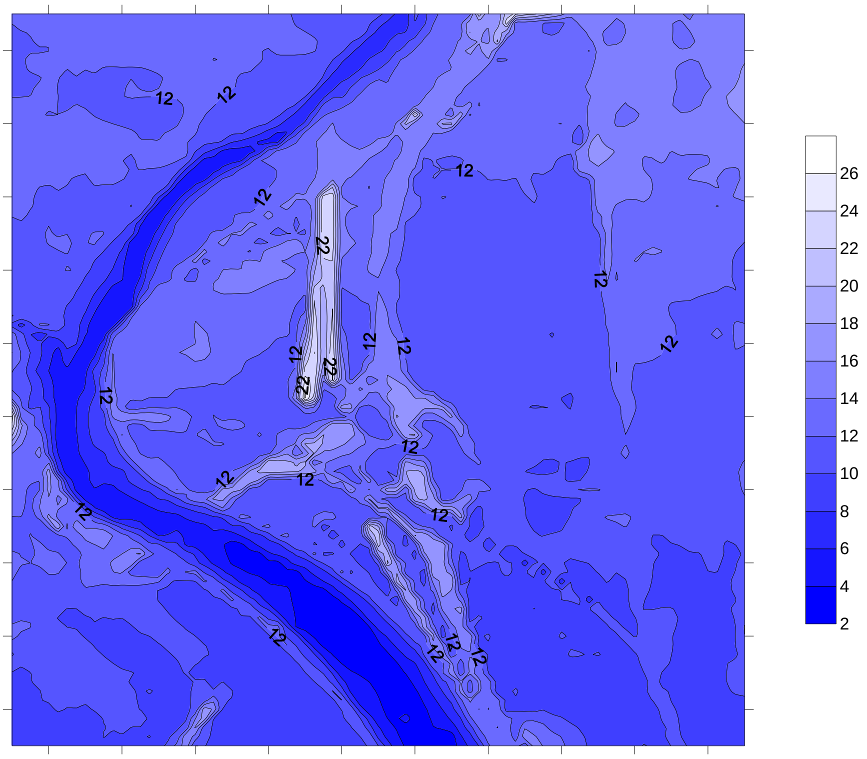

**Figure 5: Topographic digital model in meters**

NOTE: scale represents meters of elevation. Dark blue zone corresponds to Llobregat River bed, while light blue corresponds to terrain elevation.

The model includes three horizontal layers (upper aquifer, aquitard and deep aquifer) and their geometry is like the cross section in Figure 6 (thickness is variable in each cell).

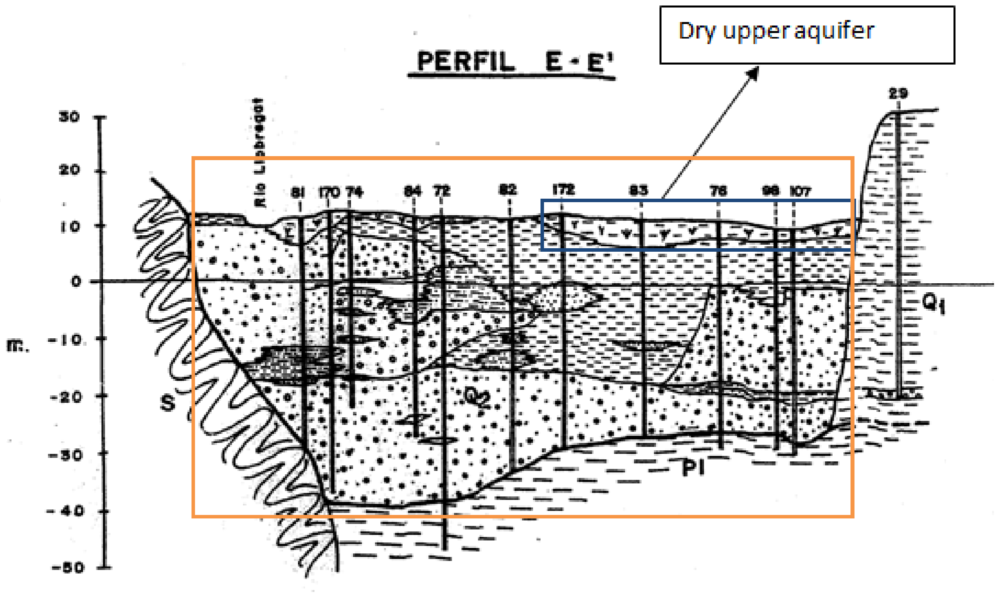


**Figure 6: Transversal geologic cross section. (Source: )**

In orange model extension (Source: Bayó, 1984)

Upper aquifer is partially dry because the aquitard has a high topographic level and in this zone the upper aquifer is very thin. Taking into account the geological profiles obtained in the drilling works of the project, this new information complemented the historical scheme.

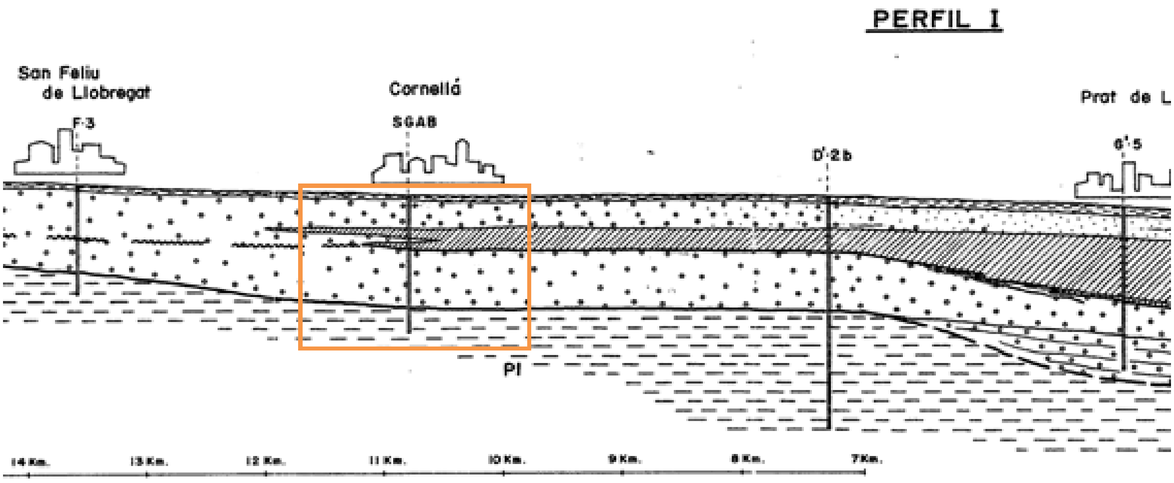


**Figure 7: Longitudinal geological cross section**
In orange model extension (Source: Bayó, 1984)

The hydraulic parameters implemented as a basin in the model are from regional model (UPC, 2004), even though in the demonstration zone of AB (Aigües de Barcelona) more accurate information was obtained after the pumping tests (July 2014).

**Table 2: Hydraulic parameters implemented on the numerical model**

| Parameter | Upper Aquifer | Aquitard | Deep Aquifer |
|---|---|---|---|
| **Permeability** | 37.5 | 0.001 | 747<br>3254<br>51<br>170 |
| **Porosity** | 0.17 | 0.002 | 0.0001 |

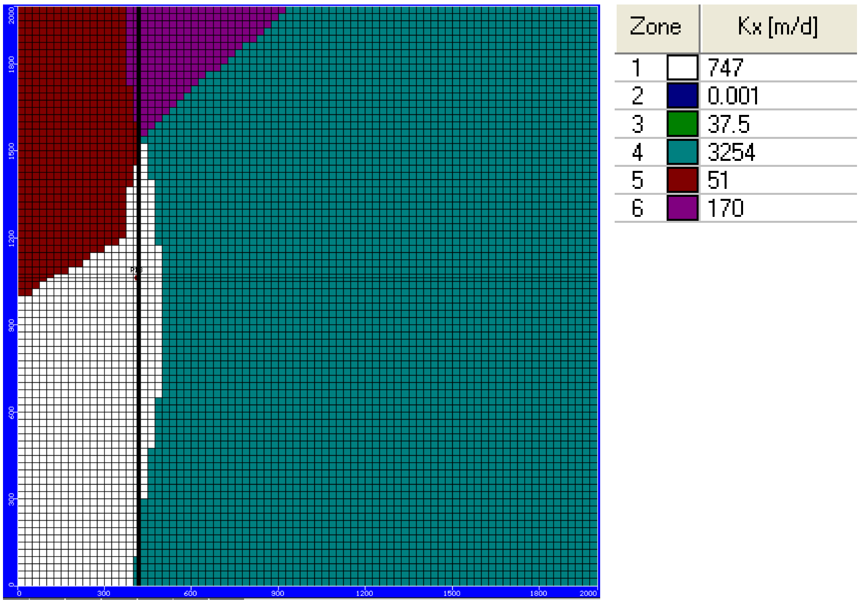


**Figure 8: Distribution of hydraulic conductivity**

Close to the well P18, there is an existing piezometer (Pz3), and two new piezometers have been drilled in July 2014 (Pz1 and Pz2) with different objectives: (i) to use them as a control point during the recharge period (monitoring of groundwater level, temperature and electrical conductivity as well as use them as sampling points) and (ii) improve the knowledge of local geology. These piezometers are at 2, 5 and 10 meters from P18. Cetaqua did the interpretation of the pumping tests, having the results listed in Table 3 [2]:

**Table 3: Hydraulic parameters obtained by Cetaqua in the pumping test. Note: r = radial distance from pumping well to observation well; t = elapsed time since start of pumping**

| Parameter | Pz1 | | Pz2 | | Pz3 | | Average |
|---|---|---|---|---|---|---|---|
| $U = r^2S/(4Tt)$ | 0.002 | 0.010 | 0.016 | 0.013 | 0.017 | 0.011 | 0.012 |
| T ($m^2/d$) | 10984 | 10633 | 9531 | 14642 | 9971 | 11510 | 11212 |
| S | Not reliable results | | Not reliable results | | Not reliable results | | -- |
| K (m/d) | 730 | 709 | 635 | 975 | 665 | 767 | 747 |

[2] See additional information of pumping tests and geological characterisation in Deliverable D35.1(a)

## 4.2 Groundwater level: starting boundary conditions

Groundwater level of an existing piezometer network was evaluated. See location of piezometers in Figure 4. Data from 2010 to 2014 was analysed using statistics. Daily data is available of the following piezometers: SV Sant Feliu, Sondeig A (Cornellà), APSA9, APSA12. The rest of piezometers have monthly data and in some case the series have incomplete data. Average piezometric level was calculated in each piezometer and average values were extrapolated, considering data goodness to fit, on the top and on the bottom model. Finally we introduce a constant head on the top model about 1.8 m and on the bottom model about 0.35 m.

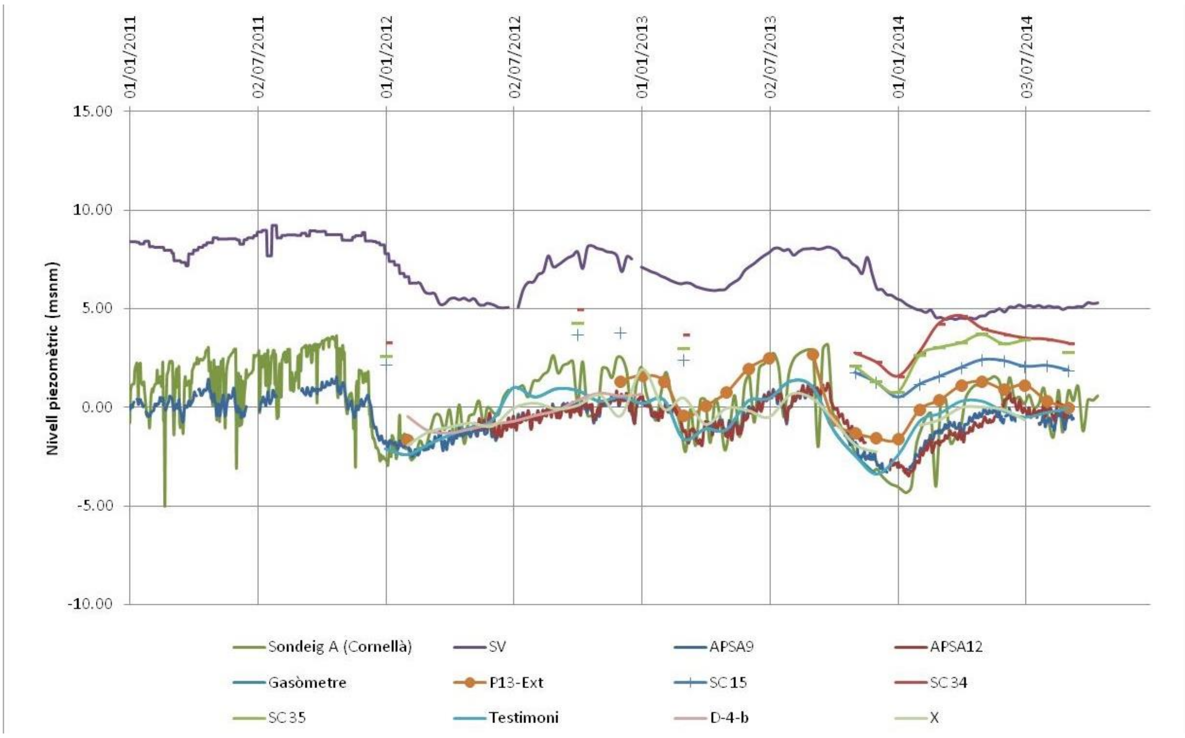


**Figure 9: Evolution of piezometric level at low valley aquifer and deep delta aquifer**

## 4.3 Flow simulations

Water injection flow was simulated in the model using a recharge of 50 L/s P18, corresponding to the demonstration phase of the project that will be carried out since mid-2015. The results are represented in Figure 10 using groundwater level and representing it using a colour legend. Figure 11 represents differences between no-injection scenario and injection scenario of 50 L/s. The radius of influence is close to model extents. Then, this model is limited to simulations according the demonstration phase of the project (50 L/s), while is not prepared to simulate real injection conditions of P18 (250 L/s). To this end, the regional model is more appropriate.

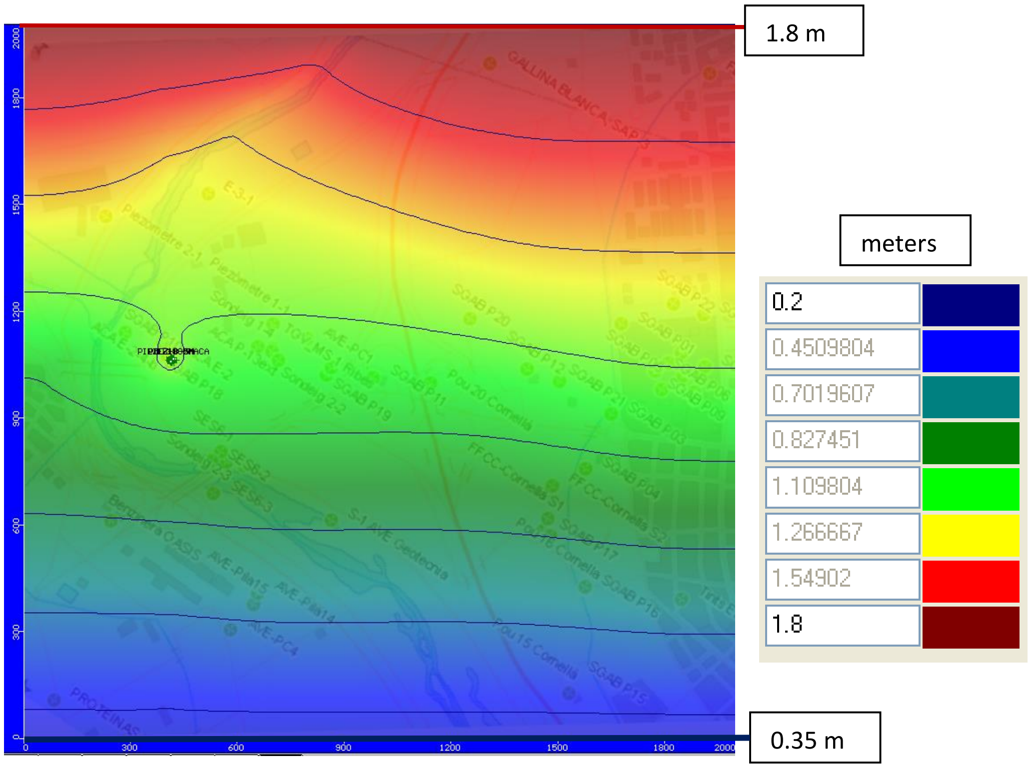


**Figure 10: Isopiezometric level calculated on the local model**

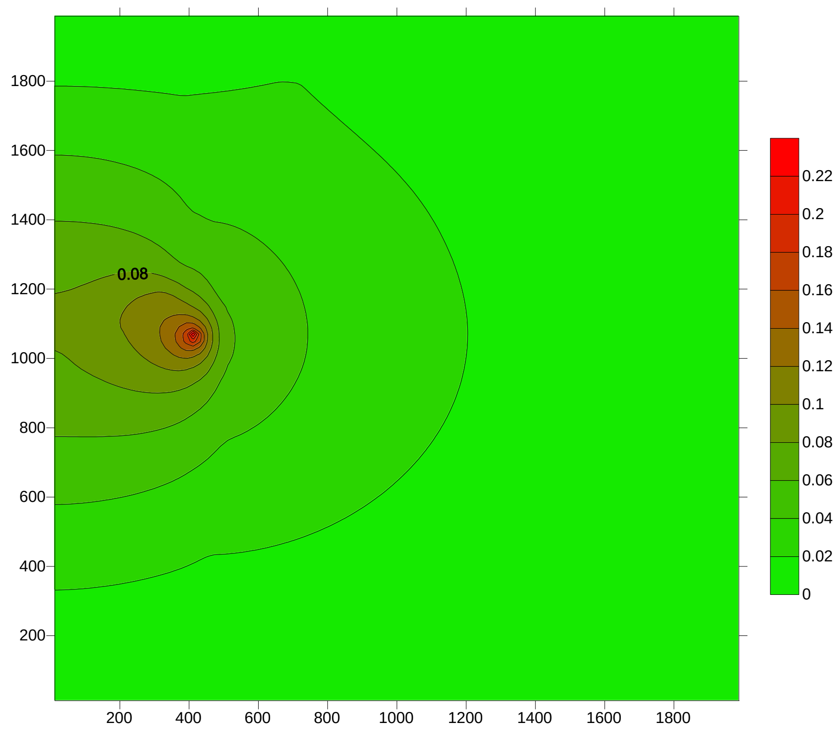


**Figure 11: Differences of piezometric level between P18 in activity and stopped**

NOTE: Flow injected 50 L/s. Data in meters

Figure 12 represents groundwater level variations calculated in Pz1, Pz2 and Pz3 respectively during a constant injection period in P18 of 50 L/s. Injection has been established from the time of 5 hours. Water level rise observed at the beginning of the timeline corresponds to an adaptation of 3 cm of the initial heads. This adaptation occurs during the first 2 hours of simulation, so results observed from 5 hours are not disturbed by these initial non-equilibrium conditions. When P18 starts the injection period at 5 hours, the model is in stationary situation. Figure 13 represents an aerial view of the expected flowlines of the injected water along the deep aquifer.

With this model it is possible to predict the transport time of injected water between the well and piezometers. With the current hydraulics parameters a particle goes from P18 to piezometers Pz1, Pz2 and Pz3 in 0.1, 0.5 and 2.2 hours respectively. These data will be calibrated in demonstration phase, using electrical conductivity, chloride or temperature as a tracer.

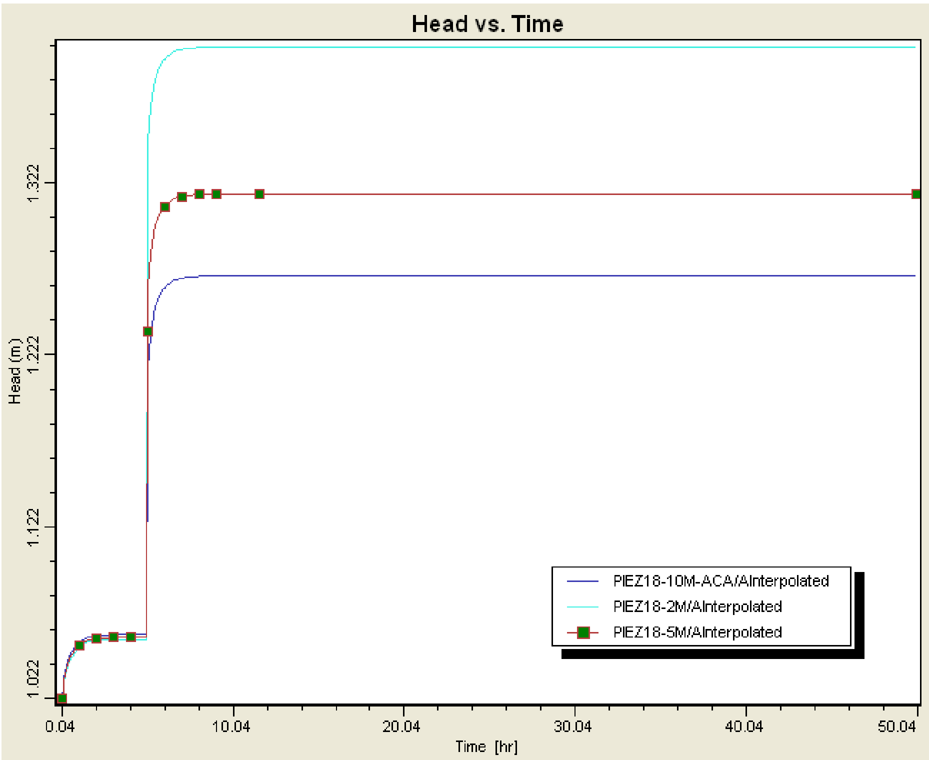


**Figure 12: Model evolution of piezometric level in Pz1, Pz2 and Pz3 in injection period.**
NOTE: P18 injects 50 L/s continuously. Starting time = 5 hours.

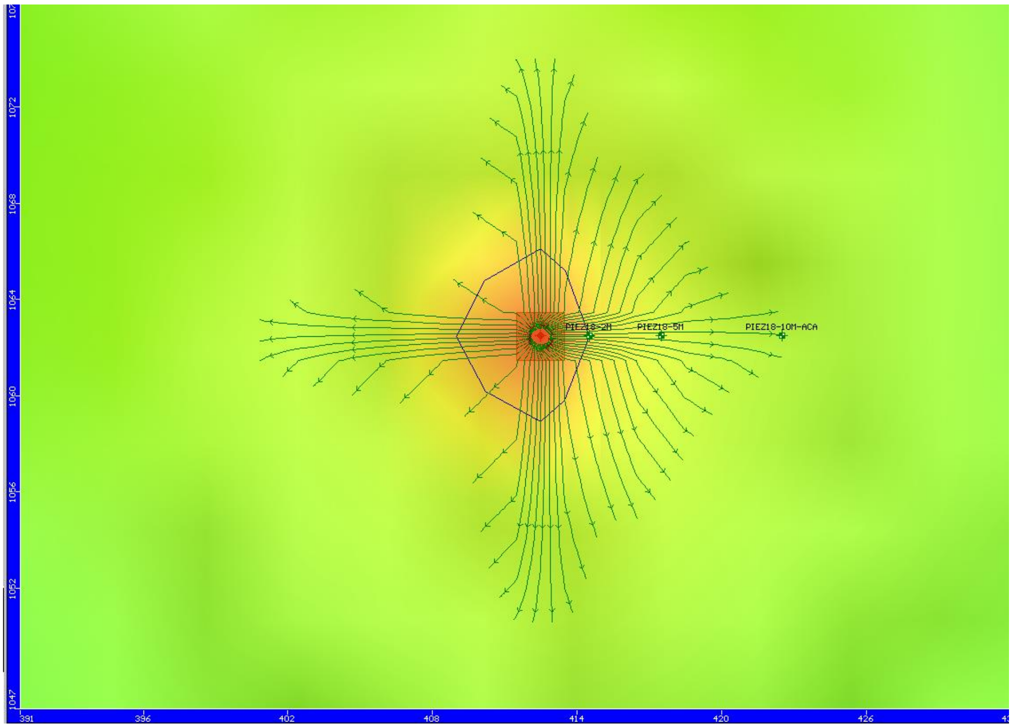


**Figure 13: Model pathlines in injection period**

**Table 4: Summary of results local model**

| Parameter | Pz1 | Pz2 | Pz3 |
|---|---|---|---|
| Groundwater Level rise (cm) | 0.38 | 0.29 | 0.24 |
| Arrival time (hours) | 0.1 | 0.5 | 2.2 |

## 4.4 Model description and considerations

### 4.1.1. The original model

In 2004 the Technical University of Catalonia (UPC) made a groundwater model to simulate the flow and transport in the Llobregat aquifers. This model was built by order of the Catalan Water Agency (ACA). Later the model was transferred to the water Users Association (CUADLL) who has upgraded, calibrated and improved every two years until now.

The software of the model is VISUAL TRANSIN developed by UPC (Galarza et al 1985). The model has 129 km$^2$ of surface. In the delta there are two layers (superficial and deep aquifer) and in the rest one layer (main aquifer). The main aquifer is formed by Cubeta Sant Andreu de la Barca aquifer (9 km$^2$), low Valley aquifer (20 km$^2$) and finally deep delta aquifer (100 km$^2$). The model is formed by finite elements and it has more than 10.000 cells. In the delta zone there are two layers because in this area there are two aquifers separated by an aquitard. This aquitard is thicker in the centre of the delta area (see Figure 15 of geological cross section).

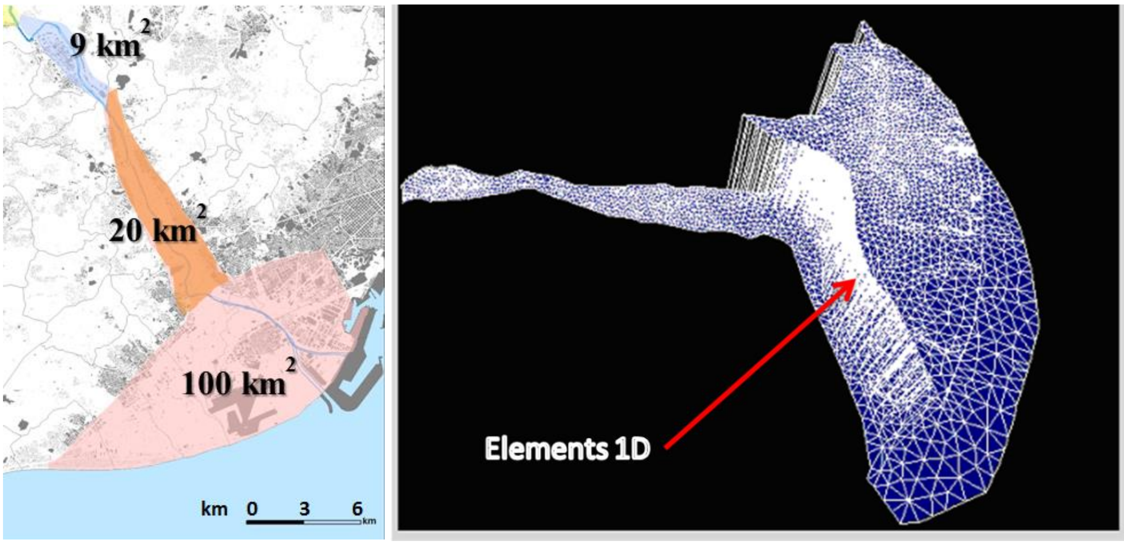


**Figure 14: Extension of regional numerical model and original mesh**
Left: Cubeta Sant Andreu de la Barca aquifer (blue) Low Valley (orange) and Delta (pink). Right: Mesh of the Groundwater Llobregat model

The unit of time used for model simulations is monthly and the working period goes from 1965 to 2013. In consequence there are 576 stress periods. In the model there are 96 wells groups. The average extraction rate in the last twenty years is about 50 hm$^3$/year.

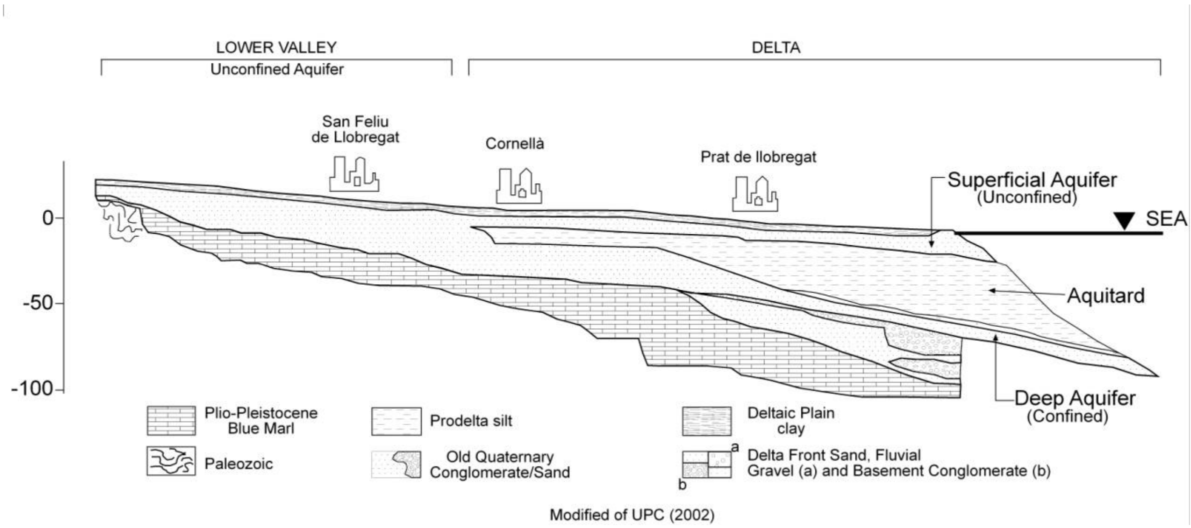


**Figure 15: Cross section of the low valley and delta aquifer**

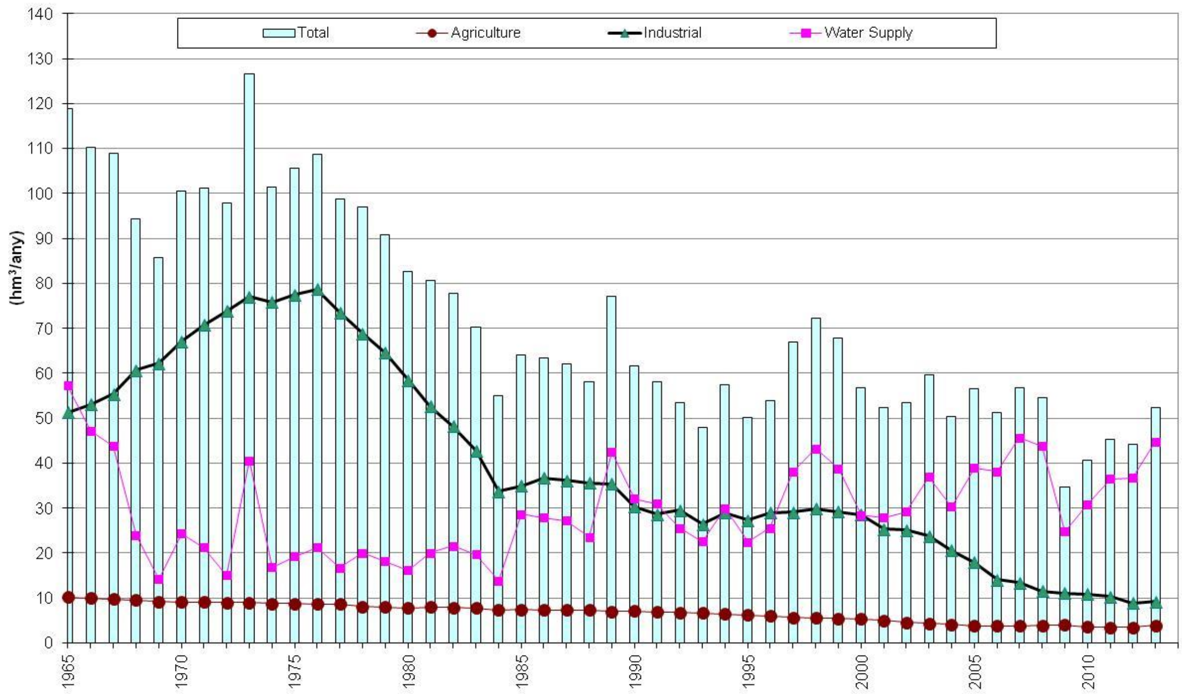


**Figure 16: Historical series of water consumption by uses in the Llobregat area**

Data compiles information of the aquifers: Cubeta Sant Andreu de la Barca, Low Valley and delta aquifer

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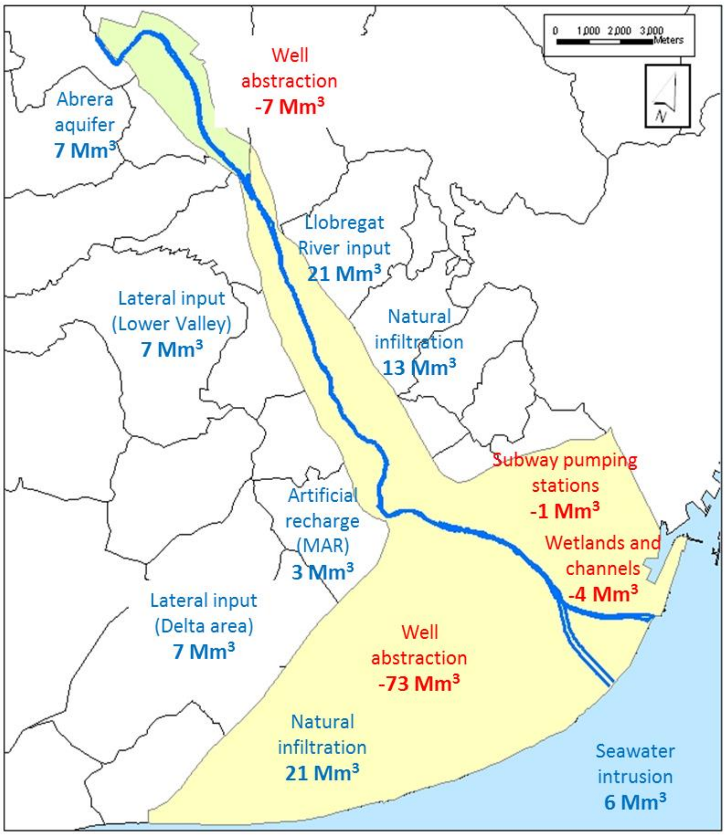


**Figure 17: Hydric balance in the model (1965 – 2013)**

NOTE: Blue colour indicates model inputs, while red colour and negative values indicate model outputs. Total account is 85 $Mm^3$ inputs and 85 $Mm^3$ outputs.

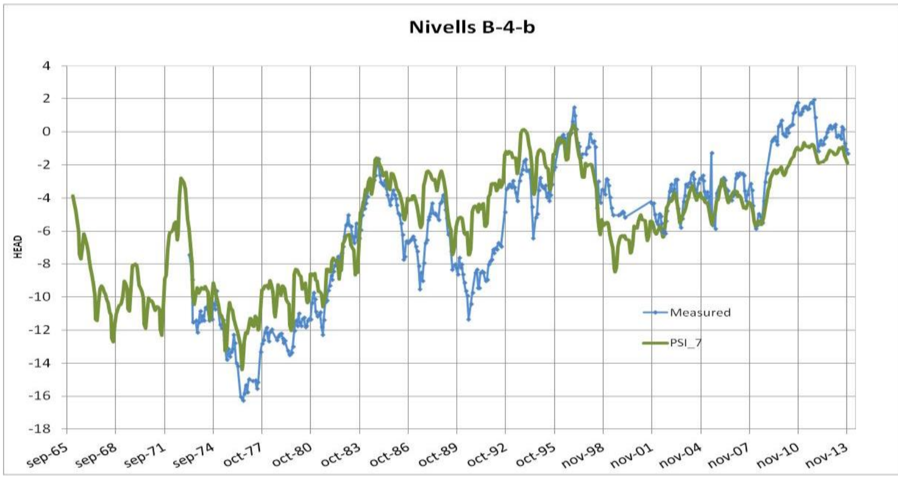


**Figure 18: Calibration of groundwater levels using historical data**

Comparison between measured level (blue line) and calculated levels (green line) in the centre of deep delta aquifer

In this aquifer, the transport model of chloride is very important it has been used historically to simulate the saline intrusion in the Delta. In 2014, the model has been updated until 2013 with a new calibration that correlates satisfactorily (see Figure 19)

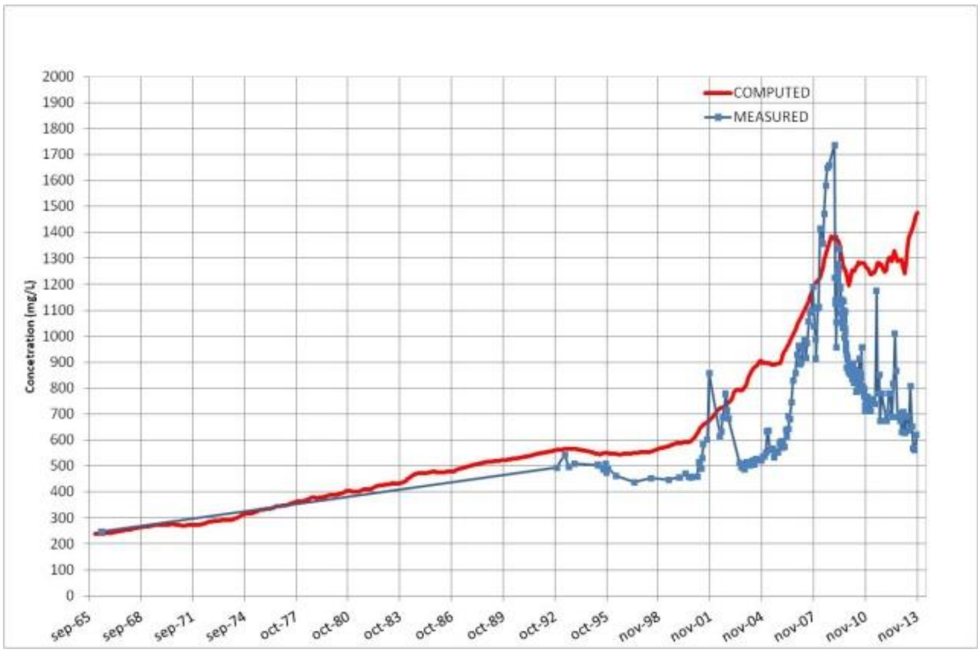



**Figure 19:Example of correlation simulated - historical data using chloride as a tracer**
(Two observation wells have been represented)

### 4.1.2. Adaptation to DESSIN demonstration site

As the original model developed in VISUAL TRANSIN works at regional scale, nodes do not correspond exactly with a pumping well, as wells are too close in some cases. That's why each group is linked to a node, and one group can include 1, 2 or more wells. Table 5 identifies the wells with the group of the model. P18 is part the group 5, which includes P10 and P18. For modelling purposes, the activity linked to group 5 will represent pumping and injection episodes carried out in P18 (considering P10 inactive). Group 11 joints ups the most relevant pumping wells of Central Cornellà. The model will be unable of discretize individual pumping rates in the same group.

Table 5: Nodes of the model representing groups of individual wells

<table>
<tr><th>Group</th><th>Node</th><th>UTM X</th><th>UTM Y</th><th>Well name</th><th>Group</th><th>Node</th><th>UTM X</th><th>UTM Y</th><th>Well name</th></tr>
<tr><td>1</td><td>5038</td><td>22362</td><td>27940</td><td>Pou 14</td><td rowspan="3">10</td><td rowspan="3">5305</td><td rowspan="3">21488</td><td rowspan="3">28942</td><td>Pou 1</td></tr>
<tr><td>2</td><td>5039</td><td>22107</td><td>27982</td><td>Pou 15</td><td>Pou 12</td></tr>
<tr><td>3</td><td>5042</td><td>21866</td><td>28166</td><td>Pou 16</td><td>Pou 21</td></tr>
<tr><td>4</td><td>5079</td><td>21303</td><td>28336</td><td>Pou 11</td><td rowspan="8">11</td><td rowspan="8">5433</td><td rowspan="8">21876</td><td rowspan="8">29060</td><td>Pou 8</td></tr>
<tr><td rowspan="2">5</td><td rowspan="2">5081</td><td rowspan="2">20387</td><td rowspan="2">28229</td><td>Pou 10</td><td>Pou I</td></tr>
<tr><td>Pou 18</td><td>Pou II</td></tr>
<tr><td rowspan="2">6</td><td rowspan="2">5173</td><td rowspan="2">21603</td><td rowspan="2">28517</td><td>Pou 17</td><td>Pou III</td></tr>
<tr><td>Pou 4</td><td>Pou 2</td></tr>
<tr><td rowspan="2">7</td><td rowspan="2">5198</td><td rowspan="2">20766</td><td rowspan="2">28454</td><td>Pou 13</td><td>Pou 6</td></tr>
<tr><td>Pou 19</td><td>Pou 7</td></tr>
<tr><td>8</td><td>5199</td><td>21161</td><td>28697</td><td>Pou 20</td><td>Pou 9</td></tr>
<tr><td rowspan="2">9</td><td rowspan="2">5301</td><td rowspan="2">21806</td><td rowspan="2">28776</td><td>Pou 5</td><td rowspan="2">12</td><td rowspan="2">5440</td><td rowspan="2">21654</td><td rowspan="2">29336</td><td rowspan="2">Pou 22</td></tr>
<tr><td>Pou 3</td></tr>
</table>

The **scenario of reference** is a result of a previous work carried out by the CUADLL with the existing regional model. This scenario has the following characteristics:

- All the temporal functions which are implemented as prescribed flow are constant and its value is an average fruit of the mass balance model.
- The extraction is supposed as a constant value, based on last years' exploitation (50 $Mm^3$/year). 25 $Mm^3$/year corresponds to the extraction located in the groups of Table 5

(Cornellà and Sant Joan Despí Area). The other 25 $Mm^3$ are extracted in other areas out of the scope of this work.

## 4.5 SCENARIO 1: ASR Demonstration scale scenario

As it was explained in the introduction, demonstration scale means the simulation of the demonstration phase of the DESSIN project, consisting in the injection of 50 L/s using a single point (P18). Meanwhile, the rest of the system will continue with the regular regime of extractions of 25 $Hm^3$/year. The extraction rate has been weighted proportionally in the other groups, except in group 5, to simulate the most favorable conditions for ASR impact assessment.

**Injection group**: Group 5 (P18 + P10)
**Pumping rate**: 50 L/s (1.57 $Mm^3$/year)
**Time injection frequency**: Continuous
**Extraction rate**: 25 $Mm^3$/year (divided proportionally in pumping groups except G5)
**Time extraction frequency**: Continuous
**Time step**: Month
**Total time modeled**: 2 years

To evaluate the influence of ASR scheme in the aquifer in terms of mixture, it has been applied a concentration of 100 to the injection water, while the native groundwater in the aquifer has no concentration. The result of this kind of simulation allows evaluate the percentage of recharged water. Three simulations with this condition have built with the three different flow of recharge.

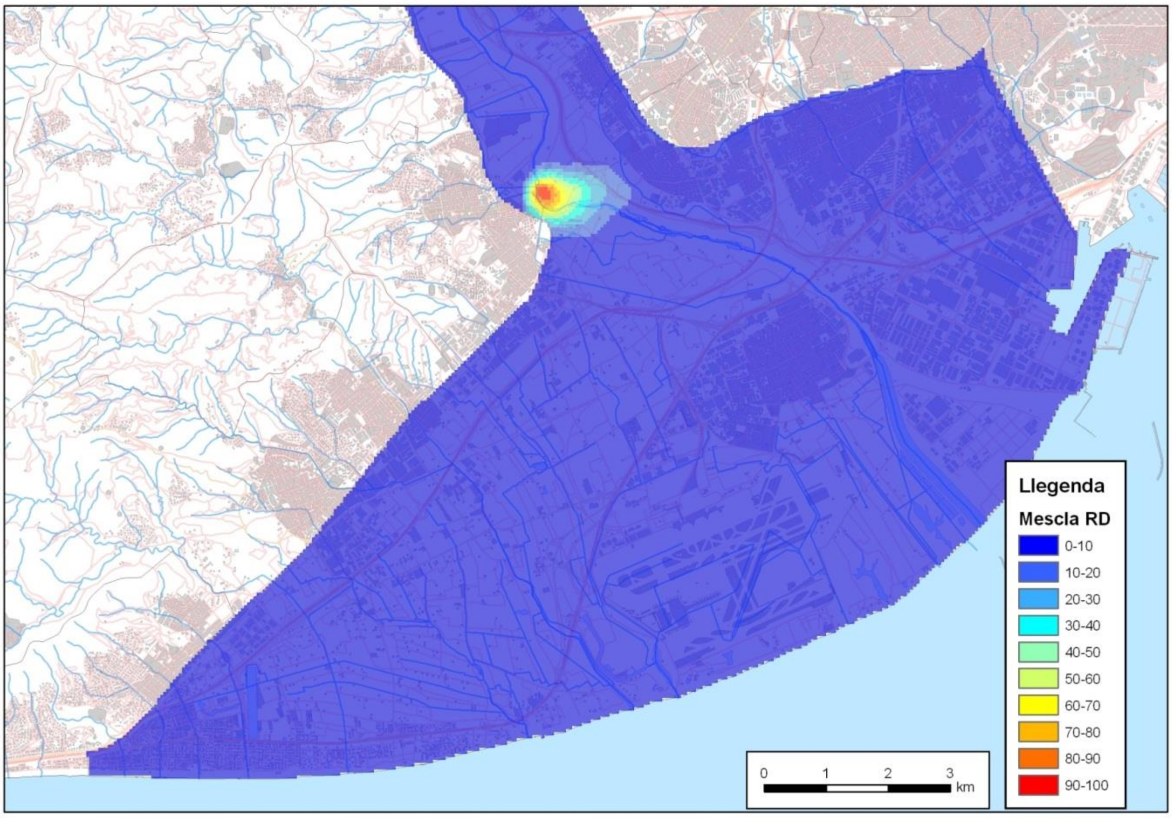


**Figure 20:Arrival plume in the demonstration phase (1.57 $Mm^3$/year) using mixing ratio values**

## 4.6 SCENARIO 2: ASR Full scale scenario

After the assessment of the impact of the injection in P18 that will be effectively done in DESSIN project, the full scale scenario aims at simulate the impact of the operation of all the ASR system, using historical volumes. The maximum injected volume of treated water in the aquifer was in 1992, with an annual volume reported of 15 $hm^3$/year. The objective of these simulations is to study the influence of injected water around the aquifer, that is to say the evolution of the plume's shape and its magnitude.

In this scenario an alternating injection/extraction has been added which is used to build three sub-scenarios with 5, 10 and 15 $Mm^3$/year respectively (these sub-scenarios are called ASR_5, ASR_10, ASR_15). The frequency of the alternation is fourteen days. To evaluate the impact, static conditions have been sought, corresponding to a simulation period of 30 years to assure equilibrium.

These three simulations are used to evaluate the influence of recharge about level heads. As the sub-scenarios with 5 $Mm^3$/year and 10 $Mm^3$/year presents less impact than the sub-scenario with 15 $Mm^3$/year, only results of ASR-15 are presented in the report.

Figure 21 shows the isodifferences' map of level heads. This isodifferences' map is built operating any value in every point of one simulation minus the value of the scenario of reference. That allows visualising if there is some significant difference in the whole aquifer. No relevant difference is between the simulation ASR_15 and the scenario of reference.

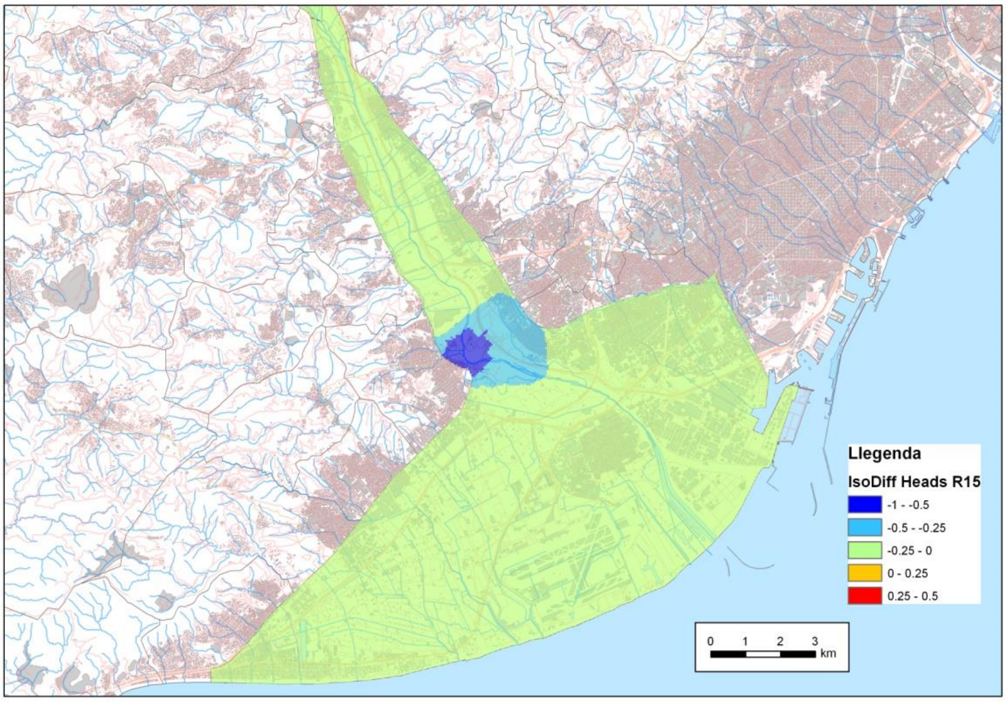


**Figure 21:Map of isodiferences with groundwater level at demonstration scale (15 Mm$^3$/year)**

Obviously, the isodifferences between maps ASR_10 – Reference and ASR_5 – Reference are even less relevant. The isodifferences' map of concentration gives no relevant differences either (see Figure 22).

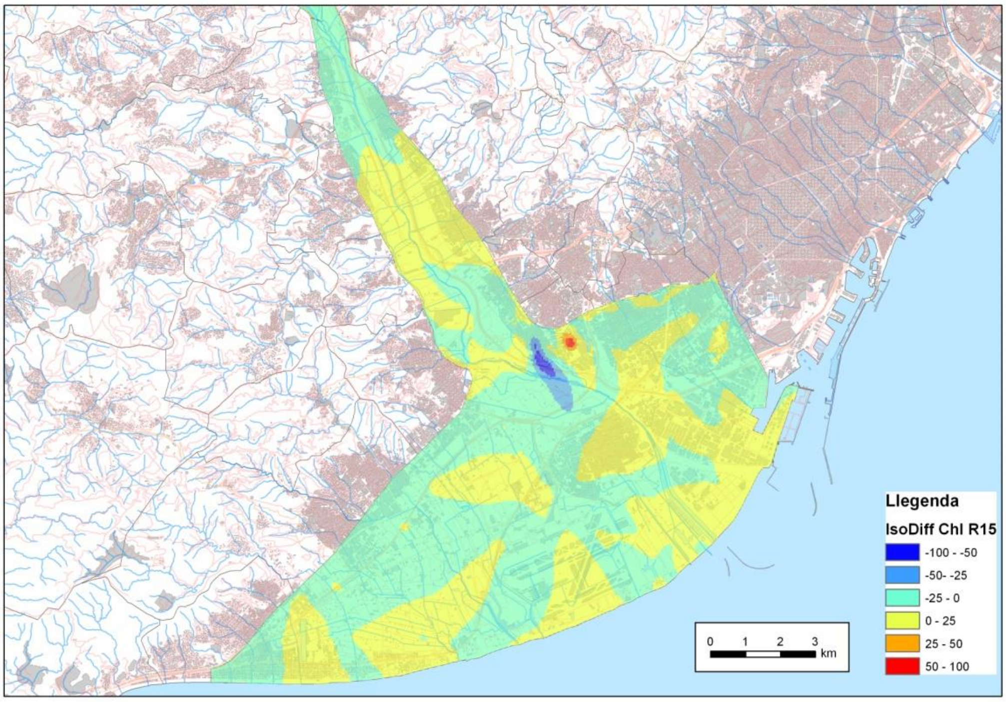


**Figure 22:Map of isodiferences with tracer concentration at demonstration scale (15 Mm$^3$/year)**

Regarding the plume of mixed water using a conservative tracer (starting in the injection water with 100 units) it can be noticed that the shape is larger while the injection/extraction flow increases. The isoline of 10% of mixed water reaches 3.8 km from the injection well in simulation ASR_15. That is on north of Prat de Llobregat village, without getting to Aigües del Prat wells. The same way, mixed water reached at extractor centre of Cornellà in very little percentage.

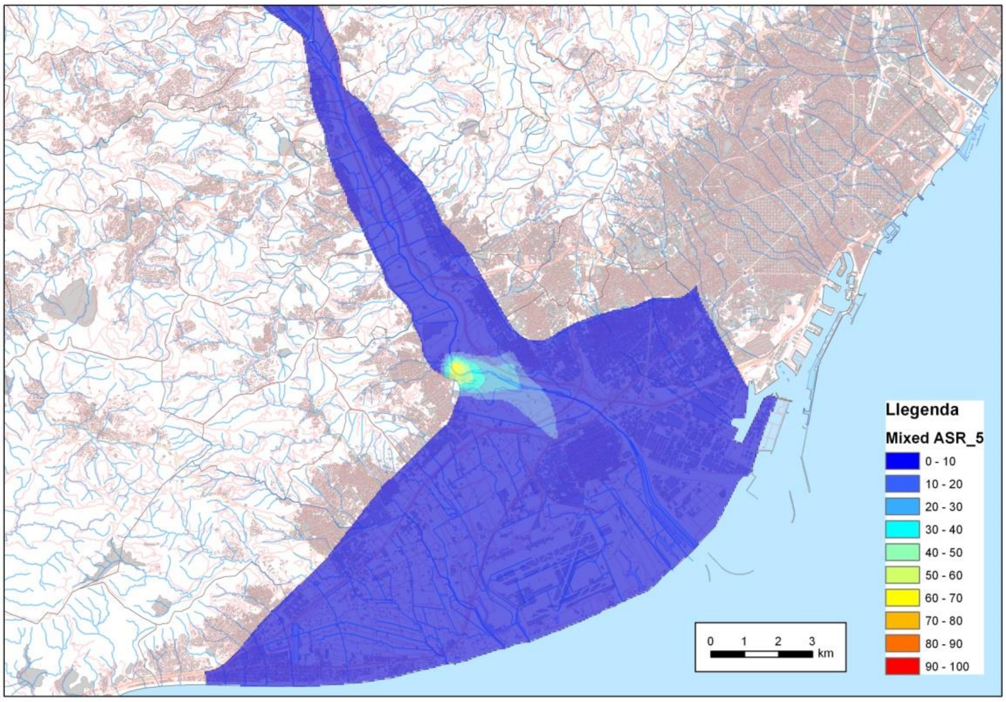


**Figure 23:Arrival plume in the full-scale phase (5 Mm$^3$/year) using conservative tracer**

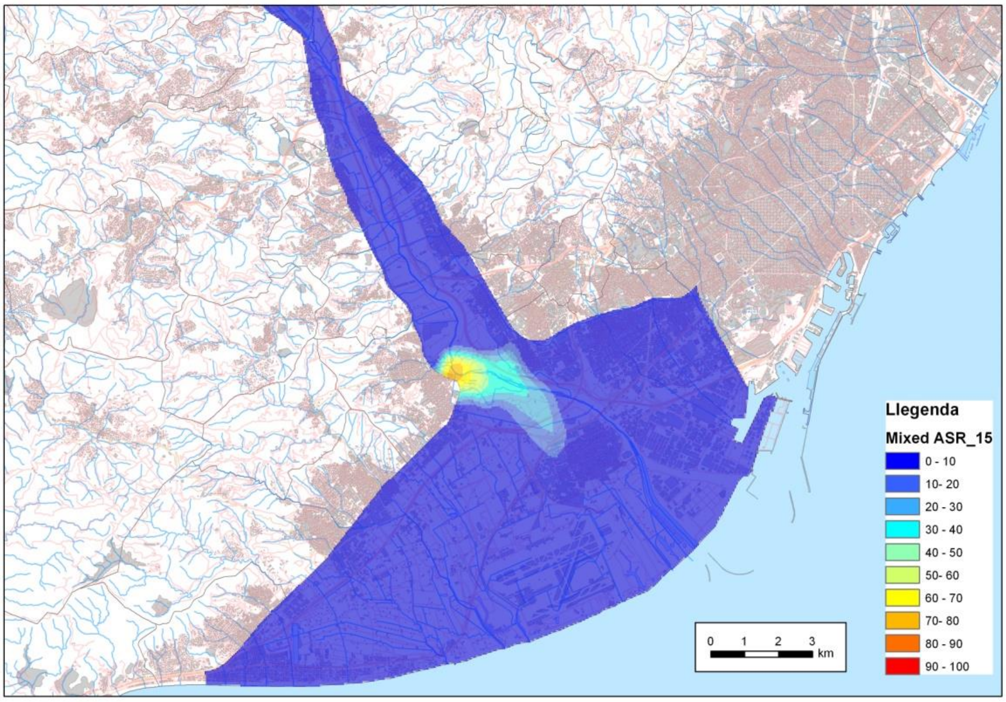


**Figure 24:Arrival plume in the full-scale phase (15 Mm$^3$/year) using conservative tracer**

Table 6: Summary of conditions in the scenarios and sub-scenarios simulated

| Parameter | REFERENCE | SCENARIO 1 RFD | SCENARIO 2 | | | |
|---|---|---|---|---|---|---|
| | | | ASR_5 | ASR_10 | ASR_15 | INJ_5 |
| Total Extraction Rate (Mm³/year) | 25 | 26,57 | 30 | 35 | 40 | 30 |
| Pumping Rate G5 (Mm³/year) | 0 | 1,57 | 5 | 10 | 15 | 5 |
| Time pumping (G5) frequency | continuous | continuous | 15 days | 15 days | 15 days | 15 days |
| Injection rate G5 (Mm³/year) | 0 | 0 | 5 | 10 | 15 | 0 |
| Time injection (G5) frequency | no | no | 15 days | 15 days | 15 days | no |
| Time step | 1 month | 1 month | 15 days | 15 days | 15 days | 15 days |
| Total time modelled | 30 years | 2 years | 30 years | 30 years | 30 years | 30 years |

Table 7: Synthesis of results of the impact of SCENARIO 1 and SCENARIO 2 in the aquifer

| Scenario | Distance of Mixing ratio (Km) | | Area of Mixing ratio (Km²) | |
|---|---|---|---|---|
| | > 10% | > 50% | > 10% | > 50% |
| RFD | 1,4 | 0,5 | 1,6 | 0,3 |
| ASR_5 | 3,6 | 0,3 | 4 | 0,15 |
| ASR_10 | 3,8 | 0,7 | 5,5 | 0,5 |
| ASR_15 | 3,9 | 0,8 | 6,1 | 0,75 |

# 5. Conclusions

Numerical model has been applied in the demonstration site of the Llobregat are as a preliminary work to assess the impact of the application of ASR at different scales. Although VISUAL TRANSIN model allows the simulation of injection and recovery processes with medium resolution (sometimes it integrates several real wells in a single node), its mesh has the adequate geographical dimension to present clearly the results. This means that numerical model is a powerful tool to present results to local stakeholders which are familiar with the use of this tool to evaluate the impact of hydrogeological in the Llobregat area.

Local numerical model MODFLOW-based has been generated using geological information obtained from the drilling works in new piezometers Pz1 and Pz2, and the hydraulic parameters calculated by the interpretation of pumping tests (see D35.1(a)). A groundwater level rise of 0.38 cm is expected in P1 in steady state during the injection phase.  Arrival time to Pz1, Pz2 and Pz3 is estimated in 0.1, 0.5 and 2.2 hours respectively. These results agree with the results obtained in a previous study carried out by Pérez-Paricio (1999), where P13 was used as potential injection well for a similar research project that was finally suspended.

Demonstration phase, in its more optimistic scenario of operation (50 L/s continuously injected along 2 years), will generate a plume of recharged water arriving 1.5 km away and will occupy an area of 1.6 $km^2$. According to the conservative transport model, the mixing ratio will be up to 50% only in the closest area of P18 (500 metres). This result has been useful for the selection of the external monitoring points: P10, P13 (both before 500 metres) and P03 (out of the scope of the plume). The monitoring of water quality of these external monitoring points will validate or vary the initial simulation carried out.

Next steps in the numerical modelling development will be the calibration with field data obtained in the demonstration phase (From July 2016). Moreover, additional simulations out of the scope of the demonstration phase will be developed. The so called “Scenario 3” will evaluate the benefits of the injection and recovery in different wells (ASTR), while “Scenario 4” will simulate also a ASTR scheme from Sant Feliu del Llobregat to Cornellà, to maximise the use of water resources in the Llobregat area. Future results will be presented as part of WA3 (Demonstration in the Llobregat site).

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