CRANFIELD UNIVERSITY

ALBAN ECHCHELH

# THE SUSTAINABILITY OF REUSING OIL AND GAS PRODUCED WATER FOR AGRICULTURAL IRRIGATION IN DRYLANDS

SCHOOL OF WATER, ENERGY AND ENVIRONMENT

Doctor of Philosophy (PhD)
Academic Year: 2016 - 2019

Supervisors: Prof. Tim Hess and Dr Ruben Sakrabani
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This thesis is submitted in partial fulfilment of the requirements for the degree of Doctor of Philosophy

# ABSTRACT

Produced water (PW) is the largest waste stream generated from oil and gas (O&G) extraction. Half of global PW (~8.5 km$^3$/year) is injected into deep disposal wells or discharged on the surface. These practices are controversial due to their environmental impacts causing increased regulation and cost. Meanwhile, water-scarce drylands host significant O&G resources. Reusing PW in irrigation provides an alternative to disposal options and could strengthen agriculture and food security in drylands. However, uncertainties exist regarding the sustainability of this practice. This research addresses these knowledge gaps by evaluating the agro-environmental sustainability and the financial cost of reusing PW in irrigation. First, the existing knowledge about PW irrigation is reviewed to identify the agro-environmental risks posed by this practice and the uncertainties regarding its sustainability. Second, irrigation with PW is simulated using a soil-water model to identify the parameters related to the environment and to the irrigation management which determine the sustainability of irrigation. Finally, a framework combining irrigation modelling and a cost analysis is applied in both regional and industrial case studies to identify agro-environmentally sustainable irrigation strategies with PW and estimate their operating costs.

This research demonstrates that irrigation with PW can be agro-environmentally sustainable if natural conditions are favourable such as on gypsum-rich draining soils in the least arid climates. Furthermore, adapted management combining irrigation at a little over the crop water needs (100–110% of the crop water needs) and PW blending in a 1:1 up to 1:4 ratio with treated sewage effluent or desalinated PW can achieve agro-environmentally sustainable irrigation by preserving soil fertility, crop yield and groundwater quality. The cost of managing PW in irrigation estimated between \$0.19–\$1.09/m$^3$, is higher or within the cost range of surface PW discharge and lower or within the cost range of injecting PW into deep disposal wells. Further research is needed to test and validate the modelling results in field conditions. A case-by-case approach is recommended to assess the broader economic and social impacts of reusing PW in irrigation.


Keywords: Arid climate, modelling, salinity, sodicity, wastewater reuse.

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| | |
|---|---|
| Al | Aluminium |
| Alk | Alkalinity |
| AI | Aridity index |
| ANZECC | Australian & New Zealand Environmental Conservation Council |
| B | Boron |
| Ba | Barium |
| BC | Blending cost |
| CBA | Cost-benefit analysis |
| BDL | Below detection level |
| Ca | Calcium |
| $CaSO_4, 2H_2O$ | Gypsum |
| CBE | Charge balance errors |
| CBM | Coalbed methane |
| CC | Chemical cost |
| CCE | Calcium carbonate equivalent |
| Cd | Cadmium |
| Cl | Chlorine |
| Cr | Chromium |
| $CO_2$ | Carbon dioxide |
| $CO_3^{2-}$ | Carbonate |
| Cu | Copper |
| DC | Desalination cost |
| DW | Drainage water |

| | |
|---|---|
| EC | Electrical conductivity |
| $EC_e$ | Electrical conductivity of the soil saturation extract |
| EPA | Environmental Protection Agency |
| ETc | Crop evapotranspiration |
| ETo | Reference evapotranspiration |
| ESC | Exchangeable sodium content |
| ESP | Exchange sodium percentage |
| FAO | Food and Agricultural Organisation |
| Fe | Iron |
| GDP | Gross domestic product |
| GIS | Geographical information system |
| GSA | Global sensitivity analysis |
| $HCO_3^-$ | Bicarbonate |
| IC | Irrigation system cost |
| K | Potassium |
| $K_{cb}$ | Basal crop coefficient |
| LC | Labour cost |
| MC | Maintenance cost |
| Mg | Magnesium |
| Na | Sodium |
| Ni | Nickel |
| $NO_3^-$ | Nitrate |
| NORM | Naturally occurring radioactive materials |
| O&G | Oil and gas |
| OC | Operating cost |

| | |
|---|---|
| OFATSA | One-factor-at-a-time sensitivity analysis |
| P | Precipitation |
| Pb | Lead |
| PC | Power cost |
| PCA | Principal component analysis |
| $pCO_2$ | Soil carbon dioxide partial pressure |
| PW | Produced water |
| RO | Reverse osmosis |
| ROPW | Reverse osmosis-treated produced water |
| RSG | Reference soil group |
| SAR | Sodium adsorption ratio |
| $SAR_e$ | Sodium adsorption ratio of the soil saturation extract |
| $SO_4^{2-}$ | Sulphate |
| SOM | Soil organic matter |
| SPWG | Stochastic produced water generator |
| Sr | Strontium |
| TDS | Total dissolved solids |
| TSE | Treated sewage effluent |
| ULPRO | Ultra low pressure reverse osmosis |
| UNEP | United Nations Environment Programme |
| USDA | United States Department of Agriculture |
| USGS | United States Geological Survey |
| WC | Water cost |
| WGR | Water-to-gas ratio |
| WOR | Water-to-oil ratio |

| | |
|---|---|
| Zn | Zinc |
| $\rho_{\mathrm{b}}$ | Soil bulk density |
| $\theta_{(m)s}$ | Soil gravimetric water content at saturation |
| $\theta_{fc}$ | Soil volumetric water content at field capacity |
| $\theta_{pwp}$ | Soil volumetric water content at permanent wilting point |

# 1 INTRODUCTION

## 1.1 Oil and gas produced water: origin, management and challenges

Petroleum formations are complex mixtures of oil, gas and water. When valuable oil and gas (O&G) are extracted, they come up to the surface with considerable volumes of water which is naturally present with the hydrocarbons in the rock formation. This water, termed 'produced water' (PW), is considered the main by-product of the O&G industry (Veil, 2011). PW also includes water which originates from surface water bodies or aquifers, that is artificially injected into the rock formation to enhance O&G recovery or to facilitate O&G extraction through hydraulic fracturing of the rock formation (Engle et al., 2014). Whereas nearly half of global PW volume is beneficially reused, mainly by reinjecting it into rock formations, the other half is mostly injected into deep disposal wells or discharged on the surface after de-oiling without being beneficially reused (Global Water Intelligence, 2014). These two management practices have considerable negative externalities. First, deep-well injection is energy-intensive, carbon-intensive, and thus is expensive (Arthur et al., 2011; Al-Rawahi et al., 2014). Besides, it is environmentally risky, as it can contaminate aquifers (Hagström et al., 2016) and induce earthquakes damaging infrastructures (Walsh and Zoback, 2015). Surface discharge is also controversial because of the risks of soil and water pollution (Christie, 2012; Konkel, 2016). Consequently, stricter environmental regulations are being developed limiting deep-well disposal permits and requiring extensive PW treatment prior to injection or discharging (Al-Sofi, 2014; Fakhru'l-Razi et al., 2009) or simply banning it completely (Igunnu and Chen, 2014). On top of that, the volume of PW is projected to increase (Dal Ferro and Smith, 2007), which will reinforce the constraints on current PW management practices and increase their costs for O&G companies (Stanic, 2014).

## 1.2 Reusing oil and gas produced water in irrigation

The beneficial reuse of PW in irrigation could be an alternative to current disposal practices and reduce PW management cost for the O&G industry. Moreover, reusing PW in irrigation could potentially provide a considerable amount of water to farmlands situated within O&G basins (Nijhawan and Myers, 2006) and create economic value (e.g. agricultural products and jobs). This option is of the utmost interest in drylands which host a significant part of the world’s O&G production and reserves (EIA, 2016), where water is scarce, and scarcity is likely to be exacerbated as a result of climate change (Feng and Fu, 2013) and a growing population (Safriel et al., 2006). Therefore, to respond to water scarcity as well as the environmental and economic limits of traditional PW disposal practices, the reuse of PW for irrigation in dry areas must be considered.

Despite the large volume available and the potential benefits of reusing PW to irrigate crops in dry zones (Chapter 3), PW quality remains a challenge as it is contaminated by hydrocarbons, dissolved minerals (e.g. mainly salts, heavy metals and radioelements) and production chemical compounds (e.g. corrosion inhibitors, biocides, emulsion breakers and antifoam) (Fakhru’l-Razi et al., 2009). These PW constituents can contaminate the soil, crop and groundwater (Table 1-1). Salts and sodium dissolved in PW are particularly concerning as their detrimental effects on the soil structural stability, crop yield and groundwater salinity were observed in a large number of experiments (Chapter 2). Although heavy metals and organic pollutants were also observed to contaminate plant tissues and limit crop development, their negative impacts were less often reported compared to salts and sodium. Heavy metals mainly constitute a food safety issue while organic pollutants constitute an emerging and local issue for unconventional O&G productions that are mainly developing in North America.

**Table 1-1. Agro-environmental risks of reusing PW in irrigation**

| Agro-environmental risk and associated issues | Relative probability | Relative severity | Relative significance |
|---|---|---|---|
| Soil sodification (i.e. soil structure destabilisation and erosion, reduced hydraulic conductivity and water logging) | Frequent | Critical | Very high |
| Soil salinisation (i.e. crop yield decline as a result of osmotic stress and groundwater salinisation) | Frequent | Moderate | High |
| Heavy metal build-up into the soil (i.e. reduced crop yield and quality due to heavy metals' phytotoxicity, food chain contamination and groundwater pollution) | Likely | Moderate | Medium |
| Soil contamination by production chemical compounds (reduced crop yield and quality due to the phytotoxicity of production chemical compounds, food chain contamination and groundwater pollution) | Occasional | Moderate | Low |
| Soil contamination by hydrocarbons (reduced crop yield and quality due to hydrocarbons' phytotoxicity, food chain contamination and groundwater pollution) | Seldom | Negligible | Very low |

Although water treatment technologies and irrigation management practices exist to mitigate the negative agro-environmental impacts of PW salinity and sodicity, their costs in the context of reusing PW in irrigation are not precisely known (Chapter 2).

## 1.3 Sustainability of irrigation with produced water

Sustainability is generally defined as meeting current human needs without compromising the ability of future generations to meet their own needs (Held, 2001). The temporality of the concept of sustainability depends on the longevity of mankind which is obviously unknown. Thus, other definitions have ‘simplified’ the temporality of this concept by assuming that an activity is sustainable if it can be maintained indefinitely. For example, the concept of sustainable agriculture was defined by Struik and Kuyper (2017, p.3–4) as “(…) *the ability of farmers to continue harvesting crop and animal products without degrading the environment or the resource base while maintaining economic profitability and social stability. As such, the term sustainability describes the result of processes that achieve that purpose and the ability to permanently and indefinitely maintain the required quantity and quality of the resources*.”

The reviewed short-term agro-environmental impacts and cost estimates call the sustainability of reusing PW in irrigation into question (Chapter 2).

## 1.4 Knowledge gaps

First, most research addressing the impacts of irrigation with PW has comprised short-term field experiments (1–3 years) whereas, O&G fields longevity generally varies from 5 to more than 100 years (Encana, 2011; The Oil & Gas Year, 2019; Total, 2015) and O&G fields generate PW as long as they are operating. Although the reviewed field experiments have shown that the high salinity and sodicity of PW are the main agro-environmental concerns when it is reused in irrigation (Chapter 2), these field experiments do not inform about the extent of these agro-environmental impacts throughout the O&G fields’ longevity. Thus, the agro-environmental sustainability of irrigation with PW remains uncertain.

Second, field experiments were carried out under specific climates and on particular soils so their results cannot be easily extrapolated to other types of drylands which include a large diversity of climates and soil types (Koohafkan and Stewart, 2008). Similarly, the qualities of the PWs used in field experiments are very specific and do not necessarily represent the range of PW qualities observed at a global scale.

Third, the few cost estimates of irrigation with PW have assumed that PW needs to be treated to very high standards (i.e. drinking water quality or water suitable for unrestricted irrigation) using costly desalination technologies to avoid damaging the soil, crop and water bodies. However, lower water quality might be acceptable if halotolerant crops are selected and grown on soils that have a relatively low vulnerability to salinisation and sodification. In addition, blending PW with fresh water to reduce its salinity, and facilitating salt leaching through over-irrigation might be cheaper than desalinating PW to control soil salinisation and sodification.

For these reasons, there is a need to estimate the agro-environmental sustainability of irrigation with different PW qualities that are representative of global PWs, under different dry climates, and on different soil types representative of drylands. Also, there is a need for estimating the cost of irrigation with PW using over-irrigation, PW blending and PW desalination to mitigate the negative impacts of PW on soil, crop and groundwater. This information is crucial to better establish where in drylands and under which conditions PW reuse projects could be successfully implemented without compromising their agro-environmental sustainability. In addition, this information would provide the evidence needed by farmers and environmental regulators to design guidelines and frameworks for the beneficial reuse of PW to irrigate crops in dry areas. Cost estimates would also provide the O&G industry and irrigators better financial visibility and encourage investments in this alternative to current PW disposal practices.

## 1.5 Research aim and objectives

The purpose of this thesis is to contribute significantly to the body of knowledge on sustainable irrigation with saline-sodic water and particularly O&G PW in drylands. This is achieved by identifying key environmental parameters, irrigation practices and costs for ensuring agro-environmentally sustainable irrigation with

PW in dry areas. Beyond its scientific contribution, this work has the ambition to participate in strengthening the O&G industry's environmental and social responsibility as well as reinforcing food security in dry zones.

This research aims to develop a framework to assess the agro-environmental sustainability of reusing O&G PW for agricultural irrigation in drylands.

The specific objectives are to:

- assess the potential of reusing PW for agricultural irrigation in drylands by estimating the volume of PW generated in drylands, the volume of PW that can be potentially reused in irrigation, and by reviewing the experiences of irrigation with PW to identify the agro-environmental risks of this practice and the solutions to adapt PW to irrigation in drylands.
- estimate the long-term impacts of irrigation with PW on soil salinity and sodicity at the field level and its effect on soil structural stability, crop yield and groundwater quality in different climates and on different soils representative of drylands.
- identify agro-environmentally sustainable irrigation strategies with PW using techniques to prevent soil and aquifer degradation.
- estimate the costs of these agro-environmentally sustainable strategies in a regional and in an industrial context.

The hypothesis underlying this research is that with the right management, irrigation with PW in drylands could be both agro-environmentally sustainable and financially competitive compared to traditional PW disposal practices.

## 1.6 Research approach

### 1.6.1 Modelling the long-term impacts of irrigation with produced water in drylands

The field, greenhouse and laboratory experiments have shown that the soil, crop and groundwater can be negatively impacted by irrigation with PW, mainly because of its high salinity and sodicity (Chapter 2). Nonetheless, these practical experiments remain limited by their duration, specific environment (i.e. soil and climate), quality of the PW used, and specific irrigation management practices. A new approach needs to be adopted in order to go beyond these limitations to address the impacts of PW salinity and sodicity in irrigation, in the long-term, in different climates, on different soils, with different PW qualities and using different irrigation managements.

The three main approaches that have been used by researchers to study soil salinisation and sodification caused by irrigation are remote sensing, field experiments, and modelling (Modupe, Alonge and Ojo, 2016).

Remote sensing is a powerful technique to produce geographical information for studying soil salinisation. It is mostly used to survey the extent of soil salinisation in a spatial dimension at a particular time (Scudiero et al., 2016). Although it can compare past and present data to produce trends, remote sensing is generally not adapted to forecast or predict soil salinisation resulting from irrigation.

Field experiments provide natural environmental conditions, however, these conditions cannot be controlled or only marginally (e.g. climate in a greenhouse experiment). Different irrigation managements can only be tested using different trial plots, this is challenging when different climates and soils need to be studied. Consequently, the results obtained from field experiments are only valid in the environmental conditions where they have been carried out and with the irrigation practices and water quality that have been used. Lastly, consistent and reliable results can be obtained after several years due to the duration of crop growth and soil transformation processes. In the case of irrigation with PW, field experiments would be informative but decades of monitoring would be required before discussing its long-term agro-environmental sustainability. All these drawbacks make field experiments time-consuming, resource-intensive, labour-intensive, costly, and consequently, not adapted to the current research.

An alternative approach is to use models, as they do not have the same disadvantages. Models can perform complex simulations quickly, which dramatically reduce the time needed for obtaining results. Models are suitable for long-term predictions and can, therefore, be used to predict the sustainability of a system. Moreover, models can be run with 'what-if' scenarios describing different situations (e.g. different soils, climates and irrigation water qualities) without the need to run many field experiments. Models also allow to simulate irrigation without any consequences on the environment whereas extreme irrigation scenarios could contaminate soil and water if carried out in a field plot. In short, compared to field experiments, models are generally time-saving, cheap to run, flexible, and safe for the environment (Graves et al., 2002). These features make models relevant for studying the agro-environmental sustainability of irrigation with PW in drylands (Mallants, Šimůnek and Torkzaban, 2017).

### 1.6.2 Integrating the irrigation model to the cost analysis

Modelling the impact of irrigation with PW in drylands would enable the identification of potential agro-environmentally sustainable scenarios which depend on soil, climate, PW quality and irrigation management. The model outputs could then be used in a cost analysis to estimate the costs of specific PW treatments and irrigation management to achieve agro-environmental sustainability. This integrated approach combining irrigation modelling and a cost analysis could be applied in different contexts (see 1.8 Thesis outline).

## 1.7 Scope of the sustainability assessment

Experimental results have shown that the soil, crop and groundwater can be negatively affected by salts, sodium, heavy metals and organic compounds present in PW (Chapter 2). Ideally, all the identified risks that these chemical species represent for the soil, crop and groundwater would need to be integrated into a sustainability assessment of PW reuse in irrigation. To date, no single model exists which can simulate the impacts of irrigation on soil salinity, sodicity, heavy metals and organic compounds. This is due to the different nature of processes governing the behaviour of (1) organic (i.e. biodegradable) and inorganic (i.e. non-biodegradable) chemicals, and (2) the mobility of salts (i.e. dissolution and crystallisation processes) and heavy metals (i.e. sorption and desorption processes) in soil (Vereecken et al., 2016). Consequently, the agro-environmental risks represented by salts, sodium, heavy metals and organic compounds need to be classified to address the most significant risks in priority.

The negative impacts of salts and sodium on soil and crop were mentioned in all the experiments discussed in Chapter 2 and appear to be the predominant agro-environmental risks when PW is used in irrigation. Salts dissolved in PW can critically degrade soil fertility because of soil salinisation, sodification and alkalinisation which are linked to a larger phenomenon termed 'secondary salinity', that is, the salinisation of soil due to human activities, particularly irrigation that brings significant salt load dissolved in the water to the soil surface (McFarlane et al., 2016). Secondary salinity can reduce the yield of irrigated crops directly through its negative effects on plant development (e.g. osmotic stress, ion toxicity and nutrition deficiency due to ionic imbalance) and indirectly through soil degradation as sodification eventually leads to topsoil erosion and desertification (Ezlit, Smith and Raine, 2010; Rhoades, Kandiah and Mashali, 1992). Beyond its dramatic consequences for agriculture, secondary salinity has a broader impact on the environment, the economy and communities. For example, secondary salinity alters the quality of downstream and underground water resources receiving the saline effluents which run-off or drain. It also indirectly affects the infrastructures built for the agricultural sector and for supplying water to communities by reducing their efficiency or their utility (Pitman and Läuchli, 2002).

The experiments that addressed the agro-environmental risks related to the heavy metals present in PW observed that these may accumulate in soil and plants but the heavy metals content in PW were not found to impact crop yield nor soil fertility (Burgos and Lebas, 2015; Martel-Valles, Benavides-Mendoza and Valdez-Aguilar, 2017). Actually, unless phytotoxic levels are reached, heavy metals in PW are more a health issue for humans and animals consuming contaminated plants rather than a concern for soil fertility and crop yield (Hass, Mingelgrin and Fine, 2010; Wuana and Okieimen, 2014). Moreover, the contents in heavy metals of PW are much less often reported compared to organic compounds and salts contents (Blondes et al., 2017) making heavy metals risk quantification harder.

In the practical experiments, the agro-environmental risks represented by hydrocarbons were very limited. Nevertheless, the agro-environmental risks represented by other types of organics have emerged through the use of production chemical compounds which are mainly used as hydraulic fracturing additives particularly in unconventional O&G productions (Hagström et al., 2016). These types of O&G productions (e.g. shale O&G, tight O&G, bituminous sands, etc.) have been developing recently and (although growing) represent only 5% of the world oil production and are mostly located in North America (Williams and Simmons, 2013). Thus, for now, the agro-environmental risks related to the reuse of unconventional O&G PW contaminated by production chemical compounds concern a minor proportion of the global PW volume in a very specific geographical area.

In conclusion, taking into account the predominance of the agro-environmental risks represented by salts and sodium; this research focuses on the impacts of PW salinity and sodicity on the soil, crop and groundwater when PW is reused in irrigation in drylands.

## 1.8 Thesis outline

The thesis structure is outlined in Figure 1-1. It is presented in the form of four stand-alone papers (chapters 3, 5, 6 and 7) completed by five 'classic' chapters (1, 2, 4, 8 and 9).

Chapter 1–***Introduction*** provides a description of PW, its current management practices and their environmental, regulatory and financial issues. It presents the potential benefits of reusing PW in drylands as well as the agronomic, environmental and financial challenges. This leads to the identification of knowledge gaps related to the agro-environmental sustainability and the financial cost of reusing PW in irrigation. Then, the research aim and objectives addressing the knowledge gaps are presented. The justifications for the research approach integrating irrigation modelling and a cost analysis are detailed. Finally, the scope of the research is presented, justifying its particular focus on the risks related to PW salinity and sodicity.

Chapter 2–***Challenges of reusing oil and gas produced water in irrigation*** reviews the experiments that were carried out to study the agro-environmental impacts of irrigation with PW on the soil, crop and groundwater. This chapter also reviews the different cost estimates of reusing PW in irrigation. This chapter is used to justify the focus on the main agro-environmental risks related to the reuse of PW in irrigation which are caused by PW salinity and sodicity. This chapter provides further evidence regarding the knowledge gaps related to the agro-environmental sustainability and cost of reusing PW in irrigation.

Chapter 3–***Reusing oil and gas produced water for irrigation of food crops in drylands*** aims to quantify the potential volume of PW available for irrigation in drylands by estimating the PW volumes generated in dry countries and regions and by estimating the proportion of global PW which is not currently beneficially reused. The quality of different types of PW worldwide, as well as the results of field experiments conducted in dry areas, are reviewed to discuss the agro-environmental risks associated with PW quality and how to adapt PW to make its reuse in irrigation more sustainable.

Chapter 4–***Selection and testing of a mathematical model for long-term prediction of soil salinity, sodicity and alkalinity as a result of irrigation*** explains the purpose of modelling, the agro-environmental sustainability indicators that must be estimated with the model, and the criteria for selecting a model. It compares different candidate models and describes the abilities, conception and limitations of the selected model, SALTRISOIL_M. Lastly, the model is tested through a sensitivity analysis to identify the main model input parameters determining the output parameters of interest (i.e. the agro-environmental sustainability indicators).

Chapter 5–***Assessing the agro-environmental environmental sustainability of irrigation with oil and gas produced water in drylands*** presents the results of the application of the SALTIRSOIL_M model in a global context and discusses the agro-environmental sustainability of reusing 15 different qualities of PW to irrigate sugar beet under four climates and on four soil types.

Chapter 6– ***Agro-environmental sustainability and financial cost of reusing produced water for agricultural irrigation in Qatar*** presents the results of the application of the SALTIRSOIL_M model in a regional context (i.e. in Qatar). Different irrigation scenarios of sugar beet with PW are simulated using over-irrigation and PW blending with treated sewage effluents and with desalinated PW to mitigate the negative impacts of PW salinity and sodicity on the soil, crop and groundwater. Besides, the operating costs of agro-environmental sustainable irrigation strategies are estimated and compared to the cost of traditional PW management options (i.e. deep injection of PW and surface discharge).

Chapter 7–***Towards agro-environmentally sustainable irrigation with produced water in hyper-arid environment: the case of Nimr oil field in Oman*** presents the results of the application of the SALTIRSOIL_M model in an industrial context (i.e. in Nimr oil field, Oman) where field experiments are currently conducted. Different scenarios of irrigation of jojoba with PW are simulated using over-irrigation and PW blending with desalinated PW to mitigate the negative impacts of PW salinity and sodicity on the soil and crop. Besides, the operating costs of agro-environmentally sustainable irrigation strategies are estimated and compared to the operating cost of disposing of PW into deep disposal wells.

Chapter 8–***Discussion*** provides the synthesis of the work by integrating the various relevant part of the research project. The knowledge gaps filled by this research are reviewed. The research findings, their practical implications and their limitations are discussed.

Chapter 9–***Conclusions*** summarises the research findings, answers the research hypothesis and suggests further directions for future research.

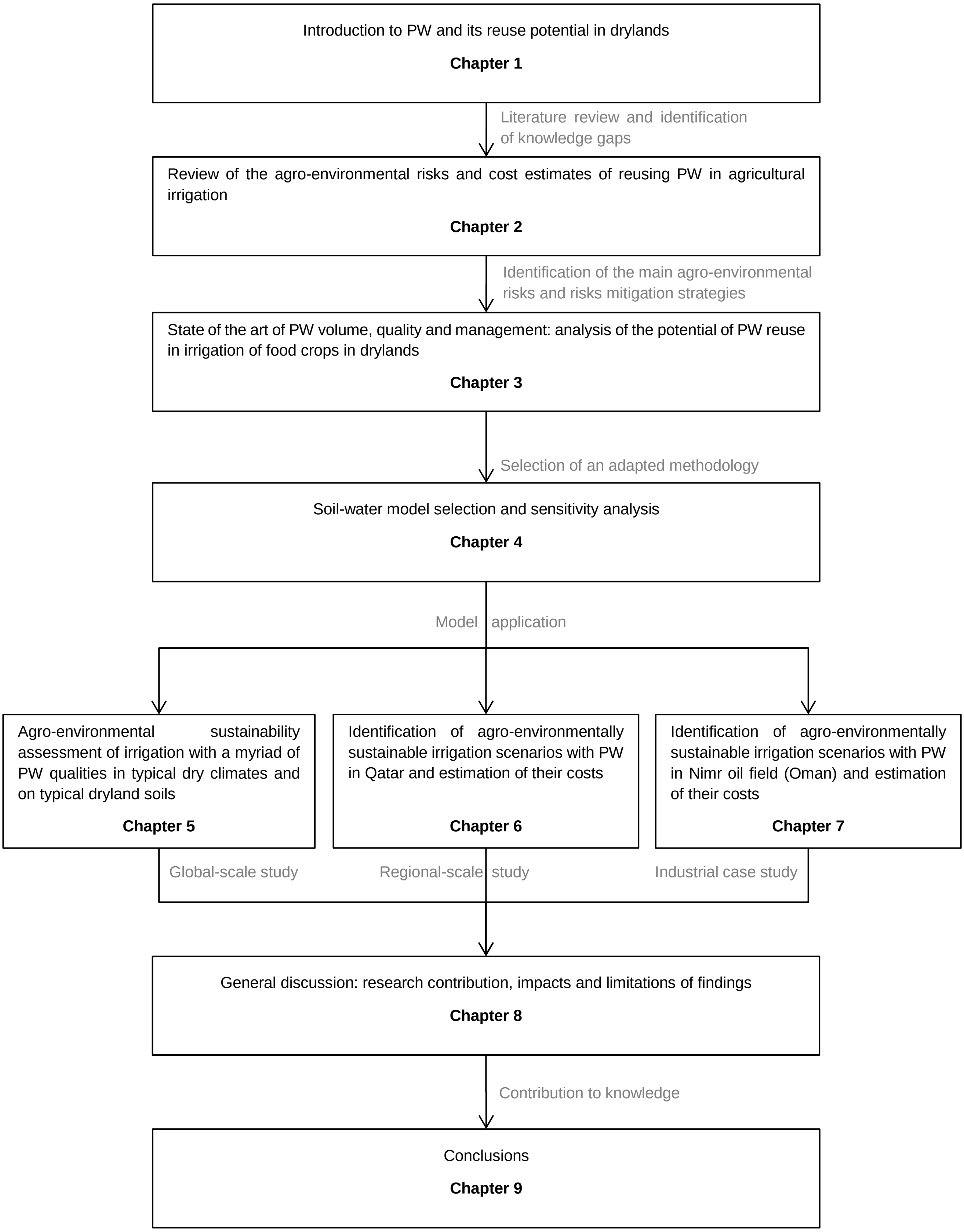


**Figure 1-1 Flow diagram showing the research rationale and the thesis structure**

## 1.9 Submitted and published papers

The following chapters for the thesis have been submitted and/or published in peer-reviewed journals. The supervisory contribution of Prof. Tim Hess and Dr Ruben Sakrabani to all papers included guidance on structure and methods as well as comments and suggestions to improve the manuscripts. Dr Fernando Visconti and Dr Jose Miguel de Paz (Valencian Institute of Agricultural Research, Valencia, Spain) have contributed to Chapter 5 by characterising PW quality through a statistical analysis and by developing a stochastic PW generator (SPWG) used to create theoretical PW qualities for the irrigation simulations.

All original work was carried out by the author.

- Chapter 3: Echchelh, A., Hess, T., Sakrabani, R., 2018. 'Reusing oil and gas produced water for irrigation of food crops in drylands', *Agricultural Water Management,* 206, pp.124–134.
- Chapter 5: Echchelh, A., Hess, T., Sakrabani, R., de Paz, J.M., Visconti, F., 2018. 'Assessing the sustainability of irrigation with oil and gas produced water in drylands', Submitted to Agricultural Water Management.

It is intended that Chapters 6 and 7 will also be submitted to journals in due course.

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# 2 CHALLENGES OF REUSING OIL AND GAS PRODUCED WATER IN IRRIGATION

## 2.1 Introduction

The beneficial reuse of PW in irrigation has raised the interest of the O&G industry and the farming sector in dry areas. However, PW quality remains a substantial barrier making PW reuse in irrigation challenging. Researchers have tried to specify and quantify the environmental risks of reusing PW in irrigation. This chapter reviews the negative impacts of irrigation with PW observed in experimental conditions in order to prioritise the risks posed by PW on the soil, crop and groundwater (see 1.7 Scope). Cost estimates of treating and reusing PW in irrigation are also reviewed to identify the gaps in the estimation of the financial cost of PW reuse in irrigation.

## 2.2 Agronomic and environmental challenges

### 2.2.1 Impacts of salt and sodium dissolved in produced water on soil and crop

Several field experiments conducted in the Powder River Basin, in semi-arid Wyoming and Montana (USA), have demonstrated the negative impacts of PW on soil fertility and on crop yield. For example, a 2-year irrigation experiment conducted by Burkhardt et al (2015) on a clay-loam soil showed that the electrical conductivity of the soil saturation extract ($EC_e$) reached 4.9 dS/m after one year of irrigation with untreated PW, whereas it reached 3.6 dS/m for the control field plot irrigated with municipal water. However, in the second year of the experiment, the $EC_e$ of the field plot irrigated with PW fell to 2.3 dS/m, likewise, the $EC_e$ of the field plot irrigated with municipal water decreased to 2.4 dS/m. This was probably caused by slight over-irrigation that increased salt leaching. As the PW used was moderately saline with an electrical conductivity (EC) of 2 dS/m but very sodic with a sodium adsorption ratio (SAR) of 33, the sodium adsorption ratio of the soil saturation extract ($SAR_e$) of the plot irrigated with PW drastically increased up to 6.7 in the first year, and up to 7.7 in the second year of the study. In contrast, the $SAR_e$ of the soil irrigated with municipal water reached 0.5 in year

1, and 0.6 in year 2. It is interesting to note that the rise of the $SAR_e$ of the soil irrigated with PW was due to both an increase of the soluble $Na^+$ content (from 2 to 21 mmol/L in year 1 and from 2 to 15 mmol/L in year 2) and a decrease of the soluble $Ca^{2+}$ (from 8 to 6 mmol/L in year 1 and from 6 to 3 mmol/L in year 2) and $Mg^{2+}$ (from 6 to 5 mmol/L in year 1 and from 5 to 2 mmol/L in year 2) in the soil saturation extract. This observation was not discussed by the authors, however, it could be explained by the replacement of $Ca^{2+}$ and $Mg^{2+}$ by $Na^+$ on clay micelles leaving $Ca^{2+}$ and $Mg^{2+}$ free in the soil solution, thus, these ions were probably leached out of the root zone by the excessive amount of water applied to the soil in the second year of the experiment. Furthermore, the essential oil yield of spearmint (the cultivated crop) was affected by irrigation with PW and declined from 0.42 to 0.19 g per 100 g of biomass in the first and in the second year of the experiment respectively. Similarly, Johnston et al (2008) used saline-sodic PW (EC = 1.4 dS/m, SAR = 24) to irrigate alfalfa during one crop season on a loamy soil and observed that the $EC_e$ nearly doubled (from 0.8 to 1.5 dS/m) while the $SAR_e$ was multiplied by 17 (from 0.45 to 7.74) compared to pre-irrigation levels. No investigation was conducted on the crop development although the authors noticed that the $EC_e$ remained below alfalfa's salinity threshold level (i.e. ≤ 2 dS/m). Ganjegunte et al (2005) studied the soil of field plots planted with various annual and perennial crops that have been irrigated for four years with PW (average EC = 3.2 dS/m, average SAR = 31). Increasing trends of salt and sodium concentrations were measured in the topsoil of field plots irrigated by PW. This suggests that salt accumulated in upper soil layers under the experimental conditions.

In semi-arid Northeast Brazil, Sousa et al (2017) used PW (EC = 2.7 dS/m, SAR = 44) filtered by sand and by cation-resin to irrigate sunflower during four months on a sandy soil (i.e. Arenosol). At the end of the irrigation period, the $EC_e$ in the 0–20 cm soil layer reached 3.4 dS/m compared to 1.8 dS/m when groundwater was used instead of PW for irrigation. The $SAR_e$ reached 18 when PW was used compared to 10 for the soil irrigated with groundwater. Apart from $Na^+$, other ions such as $Cl^-$ and $HCO_3^-$ were found in significantly larger concentration in the soil irrigated with PW compared to the soil irrigated with groundwater. Additionally, the authors detected lower carbon and fulvic acid contents in the soil irrigated with PW compared to the soil irrigated with groundwater. The authors supposed that this was due to the increasing mobility of humic substances (which contain fulvic acid) in deeper soil layers. Secondly, the organic substances degradation processes may have been affected because of the salt stress on microorganisms. In fact, it is possible that microorganisms have preferably used the fulvic acid fractions as a more labile source of organic matter, and thus the fulvic acid concentration in the soil irrigated with PW was reduced compared to the soil irrigated with groundwater. Finally, the plant biomass was negatively affected by PW as the average crop yield in the plot irrigated with PW was reduced by 15% compared to the control plot irrigated with groundwater.

In semi-arid north-east South Africa, Beletse et al (2008) used PW (EC = 7.5 dS/m, SAR = 85) to irrigate barley, Italian ryegrass, stooling rye, Bermuda grass and cotton on a loamy sand soil. The $EC_e$ ranged between 8–18 dS/m and was most of the season higher than the threshold $EC_e$ value of barley, Italian ryegrass, Bermuda grass and cotton. The crops' threshold $EC_e$ values indicate the maximum soil salinity ($EC_e$) that a crop can tolerate to reach its full yield potential. The soil was found to be enriched in sodium at about 60 cm deep. Waterlogging and drip emitters clogging were also noticed after irrigation events. Waterlogging was possibly linked to a lower soil hydraulic conductivity caused by the high PW sodicity (Mallants, Šimůnek and Torkzaban, 2017) while drip emitters clogging was likely caused by salt crystallisation (Hopkins et al., 2007).

Irrigation with PW has also been tested in extremely dry climates. For instance, in Yemen, Rambeau et al (2004) carried out a field experiment consisting of irrigating cotton and hemp during 193 days with highly saline PW (EC = 23 dS/m). This resulted in an $EC_e$ as high as 7.5 dS/m causing osmotic stress and poor growth to hemp but not to cotton which produced industrial-grade fibre quality. In hyper-arid Oman, PW (EC = 8 dS/m) was used to irrigate barley, alfalfa and Rhodes grass on a sandy soil during 102 days. During that period, the $EC_e$ increased from 1.6 to 7.1 dS/m although fresh water was used at a regular frequency (for 28 days totally) to leach excessive salt out of the root zone. Within the same period of time, the $SAR_e$ increased dramatically from 2 to 68. Lower shoot and root weights were noticed for the alfalfa irrigated with PW compared to the alfalfa irrigated with tap water, while no significant differences were found for barley and Rhodes grass. This observation was explained by the crops' specific $EC_e$ threshold values with alfalfa being less tolerant to salinity than barley and Rhodes grass (Hirayama et al., 2002).

Furthermore, greenhouse studies revealed similar observations as open-field experiments. A study conducted by Sintim et al (2017) in Wyoming (USA) showed that the soil $EC_e$ increased from 1.4 to 1.9 dS/m while the soil $SAR_e$ rose from 0.2 to 2.0 when PW (EC = 2.2 dS/m, SAR = 60) was used to irrigate camelina on a clay-loam soil. The biofuel yield of the crop was severely affected, being 23% lower for the crop irrigated by PW compared to the crop irrigated with tap water.

Similarly, in tropical-humid Alabama (USA), PW (EC = 10.6 dS/m, SAR = 73) was used continuously for 30 days in a greenhouse experiment to irrigate sorghum-sudangrass in pots filled with loamy soil. The $EC_e$ reached 8 dS/m and the $SAR_e$ rose to 91 for the soil irrigated with PW while the $EC_e$ was below 1 dS/m and the $SAR_e$ only reached 0.23 for the soil irrigated with deionised water. Consequently, the crop dry matter yield was 7.4 g/pot while it reached 11.8 g/pot when the crop was irrigated with deionised water. The crop dry matter analysis also revealed that the crop irrigated with PW contained 56% less Ca, 12% less Mg and 22% less K compared to the crop irrigated with deionised water (Mullins and Hajek, 1998).

In hyper-arid Qatar, Atia (2017) used PW diluted with tap water in a 1:9 ratio (EC = 33.6 dS/m, SAR = 14) to grow several crops in a greenhouse on a sandy soil amended with peat moss. The $Na^+$, $K^+$, $Ca^{2+}$, $Mg^{2+}$, $Cl^-$, $NO_3^-$ and $SO_4^{2-}$ contents were much higher in the soil irrigated with diluted PW compared to the concentrations measured in the soil irrigated with tap water. The tested crops were dramatically affected by irrigation with PW. Indeed, the average number of plant survival days was 35 days for saltwort, 30 days for alfalfa, 20 days for sorghum and less than 5 days for maize and sunflower.

In another greenhouse experiment carried out in Qatar (Ibrahim, Marroff and Wafi, 2009), PW (EC = 2.7 dS/m, SAR = 0.34) was used to grow alfalfa in the same soil type as in the previous experiment. Although the soil salinity and ionic composition were not significantly different between the soil irrigated with PW and the soil irrigated with tap water, the plants which were watered with PW were negatively affected compared to those irrigated with tap water. In fact, the average fresh weight, leaf area, number of leaves per plant and plant height were significantly lower when PW was used instead of tap water. What affected plant development was not discussed, but as soil salinity was similar in both treatments, the difference in plant development might be caused by other factors than the irrigation water salinity and sodicity. Another possible explanation is that the soil sampling method was biased. In fact, the whole soil was mixed before sampling, yet, the accumulation of salt in the topsoil might have remained unnoticed as the topsoil layer was not specifically sampled and analysed. Thus, salt concentration in the root zone could be the cause of the underdevelopment of the crop irrigated with PW compared to the same crop irrigated with tap water.

A clay dispersion test was carried out in laboratory conditions by Burgos and Lebas (2015) with two synthetic PWs; the first, $PW_a$ contained organic elements and chemical additives commonly used in the O&G industry (i.e. corrosion inhibitors and low dosage hydrate inhibitors) while the second, $PW_b$ was as saline as $PW_a$ (total dissolved solids = 7140 mg/L) but did not contain organic elements nor chemical additives. The test highlighted the higher vulnerability of the silty clay soil (27% clay) compared to the sandy soil (8% clay) to clay dispersion and soil structure degradation. However, the kinetics of soil structure degradation was slower for the silty clay soil compared to the sandy soil. Interestingly, it was noticed that the soil structure degradation was also slower when $PW_a$ was used instead of $PW_b$. The authors hypothesised that the organic compounds contained in PW could delay soil structure degradation. This hypothesis is relevant knowing the role of organic compounds in the maintenance of soil structural stability (Roldán, Albaladejo and Thornes, 1996).

### 2.2.2 Impacts of salt and sodium dissolved in produced water on groundwater

In addition to negatively affect soil properties and crop yield, applying PW to the soil is also suspected of altering groundwater quality. Engle et al (2011) have shown in the Powder River Basin that the groundwater salinity has significantly increased in a well monitored over a one-year period as a result of irrigation with PW. Despite the lack of experimental evidence, the risk of altering groundwater has been discussed in an environmental impact assessment of PW reuse for irrigation in Australia (Biggs et al., 2013). This issue was also integrated into a framework aiming at preventing environmental degradations resulting from irrigation with PW (Newell and Connor, 2006).

### 2.2.3 Impacts of heavy metals dissolved in produced water on soil and crop

In semi-arid north-east Mexico, Martel-Valles et al (2017) used four different qualities of PW (EC = 1.13–2.30 dS/m, SAR = 1.63–2.92) to irrigate tomato in a greenhouse during 117 days. The crop was grown in containers filled with peat moss and perlite. The results of the experiment showed that compared to the use of fresh water, the use of PW significantly reduced fruit, stem and leaf fresh weights. These reductions were attributed to a nutrient imbalance as the uptake and accumulation of minerals in plant tissues increased for Na and Mg and decreased for Mn and Cu when PW was used instead of fresh water. No significant differences were observed in Ni, Cd, Pb and Cr contents between the plants irrigated with PW and the plants irrigated with fresh water.

In the laboratory experiment (temperature = 20–26°C, relative humidity = 73%) carried out by Burgos and Lebas (2015), the two synthetic PWs: $PW_a$ and $PW_b$ were used raw and diluted with tap water at increasing dilution ratios (i.e. 1:2, 1:3, 1:4, 1:5, 1:6, 1:10 and 1:20) on reed canary grass, switch grass and sorghum. Three substrates were used: hydroponics (i.e. soilless), a sandy soil and a silty clay soil. In the hydroponics experiment, the plant content in Na and Ni logically increased as a consequence of irrigation with $PW_a$ and $PW_b$ compared to irrigation with tap water. On the other hand, the plant contents in Ca, Cr and Al were higher for plants irrigated with $PW_a$ compared to the plants irrigated with tap water, whereas no significant differences were found between the plants irrigated with $PW_b$ and those irrigated with tap water. This suggests that chemical additives in PW have a role in heavy metals absorption by plants. However, when plants are grown in soil, the accumulation of Ca, K, Na, Ni, Zn and Mg in the plants increased proportionally to the PW content in the irrigation water, and this independently of the type of PW used. Thus, it is likely that the soil impacted the plant heavy metals uptake probably through the soil retention and release mechanisms of heavy metals (Hass, Mingelgrin and Fine, 2010).

### 2.2.4 Impacts of organic pollutants dissolved in produced water on soil and crop

Pica et al (2017) studied the impacts of PW salinity and total organic carbon content (TOC) on the growth and physiological parameters of rapeseed and switch grass grown in a greenhouse on a clay-loam soil irrigated during 106 days with PW initially used for hydraulic fracturing operations in the Denver-Julesburg Basin (Colorado, USA). Five different PW qualities varying from low TOC-EC to high TOC-EC were tested. The germination test showed that rapeseed and switch grass seed germination percentages decreased as the EC and TOC content of PW increased. Rapeseed was more tolerant to salinity and to TOC than switch grass but the seedling emergence for both crops was below 5% when the PW EC and TOC content reached 34.4 dS/m and 1352 mg/L respectively. No trace of hydrocarbons was detected in the soil after irrigation because of their volatilisation and biodegradation. The volatilisation of organic compounds after irrigating crops with synthetic PW was also noticed by Burgos and Lebas (2015) in a laboratory experiment, although it was not specifically quantified. The same experiment also showed that the soil positively buffers the negative effects of organic substances on the crop, although further evidence is needed to support this assumption. Nonetheless, these observations support the fact that residual hydrocarbons are not a major concern for the soil when de-oiled PW is used in irrigation. However, at high concentration (i.e. 232.3 mg/L TOC) hydrocarbons dissolved in PW can affect the crop physiology through a decrease of crop biomass and an increase of leaf electrolyte leakage which are symptoms of plant stress (Pica et al., 2017).

Interestingly, these experiments indicate that although residual hydrocarbons and production chemical compounds are present in PW, they are not of primary concern when PW is reused in irrigation. This is explained first by the high technicity and efficiency of the O&G industry to extract as much valuable hydrocarbons as possible from the mixture of oil, gas and water, leaving only a minimal amount of hydrocarbon residues in PW (Kokal and Al Ghamdi, 2006). Second, hydrocarbons contamination is a well-known risk in the O&G sector. Thus, O&G firms and environmental authorities have a relatively long experience and know-how in managing the risks of environmental pollution caused by hydrocarbons dispersed or dissolved in PW (e.g. untreated PW leaks and spills). Therefore, very early, PW treatments were designed to remove hydrocarbons from PW to drastically reduce the risk of polluting the environment by this type of contaminant (Fakhru'l-Razi et al., 2009).

## 2.3 Financial cost of reusing produced water in irrigation

Using water resources of compromised quality (e.g. municipal sewage effluents, industrial wastewater, drainage water, brackish groundwater, saline water, etc.) in agricultural irrigation implies additional costs compared to the use of freshwater resources (Urkiaga et al., 2006). These extra costs are related to the eventual water treatments, specific irrigation practices and soil management techniques that need to be implemented to avoid the negative impacts of irrigation on the soil, crop and groundwater. Regarding the reuse of PW in irrigation, techniques such as over-irrigation to increase salt leaching (Norvell et al., 2009), PW blending (Atia, 2017; Martel-Valles, Benavides-Mendoza and Valdez-Aguilar, 2017; Mullins and Hajek, 1998; Sintim et al., 2017), desalinating PW with reverse osmosis (RO) (Sousa et al., 2017; Weber et al., 2017), and amending the soil with gypsum (Bennett et al., 2016; Johnston, Vance and Ganjegunte, 2008) have been used in field experiments to tackle the principal negative effects of PW that are soil salinisation and sodification.

### 2.3.1 Operating cost of reusing produced water in irrigation

The total operating cost of reusing PW in irrigation varies widely and highly depends on PW treatment (i.e. cost of the treatment technology used) and to a lesser extent on irrigation management (i.e. cost of the irrigation system). In fact, the operating cost of PW treatment partly depends on technological choices. For example, the desalination of PW to generate irrigation-grade water quality using different types of membranes in the USA was estimated between \$0.94–\$1.29/m$^3$ for RO membranes, \$0.89–\$1.19/m$^3$ for ultra-low pressure RO membranes and \$0.89–\$1.22/m$^3$ for nanofiltration membranes (Xu, Drewes and Heil, 2008).

The original PW quality and the targeted water quality after treatment are critical parameters determining PW treatment cost. For instance, the operating cost of upgrading PW up to potable level using RO membranes in California (USA) was estimated between \$0.38–\$0.50/m$^3$ (Tao et al., 1993) and more recently up to \$1.15/m$^3$ (Meng, Chen and Sanders, 2016). However, crops do not need to be irrigated with such high water quality, thus these costs could be revised downwards if irrigation-grade water quality is targeted instead of potable water quality.

The water treatment capacity is also a determinant as scale economies impact the unit cost of PW treatment. As an illustration, a small to mid-sized RO treatment unit with a capacity of 50 m$^3$/day can generate water suitable for irrigation at \$1.23/m$^3$ (Muraleedaaran et al., 2009) whereas a 300 m$^3$/day-plant could generate a similar water quality for \$0.88/m$^3$ (Ersahin et al., 2018).

The cost of reusing PW in irrigation also depends on how saline ‘wastewater’ (e.g. RO brine and collected drainage water from irrigation) is managed. While discharging saline effluents to the sea would cost be between \$0.06–\$0.50/m$^3$ (Fakhru’l-Razi et al., 2009), it is only relevant for coastal areas. For inland locations, saline effluents can be either deep-injected or evaporated. The injection of saline effluents into deep disposal wells was estimated at \$0.30/m$^3$ in Oman (Hardisty, 2010) whereas the same technique costs \$1.85/m$^3$ in New Mexico (USA) due to high transportation cost (Muraleedaaran et al., 2009). Pond evaporation cost depends on infrastructure size and climatic conditions. Indeed, the cost of evaporating PW in a constructed evaporation pond was estimated between \$1.57–\$5.09/m$^3$ in arid New Mexico (USA) and between \$3.15–\$15.73/m$^3$ in semi-arid Wyoming (USA).

Additional parameters such as the geographical location of the PW reuse project, the cost of inputs in a given area (i.e. energy, labour, etc.) and the distance between O&G wells and irrigated farmlands influence the total operating cost of reusing PW in irrigation.

### 2.3.2 Total cost of reusing produced water in irrigation

Information about the total cost (i.e. operating and capital costs) of reusing PW in irrigation is extremely limited. Moreover, since a large part of the parameters determining costs depends on locations, the few estimates available are site-specific and cannot be considered as references for estimating the cost of PW reuse in irrigation at the global scale.

For instance, a techno-economic analysis considered the reuse of PW for irrigation in Colorado (USA) and estimated the total annualised cost between \$2.32–\$2.96/m$^3$ depending on the PW salinity. These PW treatment costs were higher than the cost of deep-injecting PW in private wells but lower than deep-injecting PW in commercial wells (Dolan, Cath and Hogue, 2018). However, this study did not consider the possibility of treating PW to different standards depending on the soil type and climate aridity. Indeed, as soil and climates influence the long-term soil salinity and sodicity levels (Chapter 4), some environments might be relatively less vulnerable to salinisation and sodification, and could thus tolerate higher PW salinity. Consequently, these costs might be over-estimated as PW could be treated to lower standards making its reuse in irrigation cheaper than expected.

In Montana (USA), the total annualised cost of treating PW with RO to produce irrigation water for neighbouring farms was estimated at \$4.87/m$^3$ which was in this case, cheaper than injecting PW into local deep disposal wells (Szép and Kohlheb, 2010).

### 2.3.3 Profitability of reusing produced water in irrigation

One of the main benefits of reusing PW in irrigation is the creation of added value through the production of crops. Indeed, disposing of or discharging PW has a cost but no economic benefit. On the contrary, reusing PW in irrigation has a cost but it also generates economic value which needs to be quantified to assess the profitability of this practice.

Information about the profitability of PW reuse in irrigation is very scarce. In Oman, a financial simulation assessed the profitability of reusing PW to irrigate tomatoes with treated PW. The PW treatment cost including de-oiling, desalination and boron removal was estimated at $0.41/m$^3$ and the power cost related to the irrigation system at $0.02/m$^3$. Thus, assuming an annual water use of 5,922 m$^3$/ha, a crop yield of 57.34 t/year and a tomato value of $320.11/t, the theoretical profit of using PW to irrigate tomatoes was estimated at $12,433/ha, that is, $2.67 per m$^3$ of PW used (Kojima et al., 2015). This financial analysis does not include any other costs than those related to the irrigation (e.g. fertilisation, machinery, pesticides, labour, etc.).

## 2.4 Conclusions

The principal risks of reusing PW in irrigation are posed by salts and sodium dissolved in PW which can lead to soil salinisation and sodification. Indeed, the vast majority of the reviewed experiments have shown that irrigation with PW increases soil salinity which reduces crop productivity. Furthermore, the application of PW to the soil increases soil sodicity which degrades soil structure, decreases the soil hydraulic conductivity and eventually leads to an irreversible loss of soil fertility and to soil erosion. Other minor risks include the impacts of heavy metals and organic pollutants which are concerns for food safety as heavy metals accumulate in plants, and for crop development which may be negatively affected by organic pollutants. The cost of reusing PW in irrigation is not precisely known as the reviewed cost estimates assumed very high treatment standards while irrigation quality standards might be less expensive to achieve. Finally, whereas PW blending and desalination were the main soil salinity and sodicity control measures tested in experiments, salt leaching by over-irrigation could be also used and might be more economic compared to enhancing PW quality.

## 2.5 References

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# 3 REUSING OIL AND GAS PRODUCED WATER FOR IRRIGATION OF FOOD CROPS IN DRYLANDS[1]

## 3.1 Introduction

The oil and gas (O&G) industry produces large volumes of water during the extraction, processing and refining of hydrocarbons. The water that is brought to the surface with hydrocarbons during extraction is termed 'produced water' (PW); this often comprises both formation water (which naturally occurs in significant quantities in the reservoir with the hydrocarbons) and water that has been withdrawn from another source, injected into the O&G reservoir, and returns to the surface with the hydrocarbons (e.g. water injected for enhanced oil recovery and for hydraulic fracturing) (Engle, Cozzarelli and Smith, 2014). In terms of volume, PW is by far the largest by-product or waste stream associated with the O&G industry (Veil, 2011). In certain conditions, PW can be reused for beneficial purposes such as agricultural irrigation, but, the volume of PW currently reused this way represents only a small proportion of the total PW generated. Nonetheless, beneficial reuse of PW is growing (Burnett, 2004; Clark and Veil, 2015) and could provide a substantial volume of irrigation water to crops located near O&G facilities in drylands (Guerra, Dahm and Dundorf, 2011).

In this paper, drylands are defined by a precipitation to potential evapotranspiration ratio below 0.05 i.e. hyper-arid climate, up to 0.65 i.e. dry sub-humid climate (Barrow, 1992; FAO, 2016; Safriel et al., 2006). Many drylands contain massive hydrocarbon resources (e.g. the Persian Gulf, the Western USA, the Gulf of Mexico, the Libyan Desert or the Caspian Sea countries). There are also large coal resources from which gas and synthetic fuels are produced in the USA, China, Australia and South Africa (Figure 3-1). The Middle East and North Africa region, which is one of the most populated dry areas (World Bank, 2016); represents about 33% of the oil production and 23% of the gas production in the world (EIA, 2016).

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[1]Echchelh, A., Hess, T., Sakrabani, R., 2018. Reusing oil and gas produced water for irrigation of food crops in drylands. *Agricultural Water Management*, 206, pp.124–134.

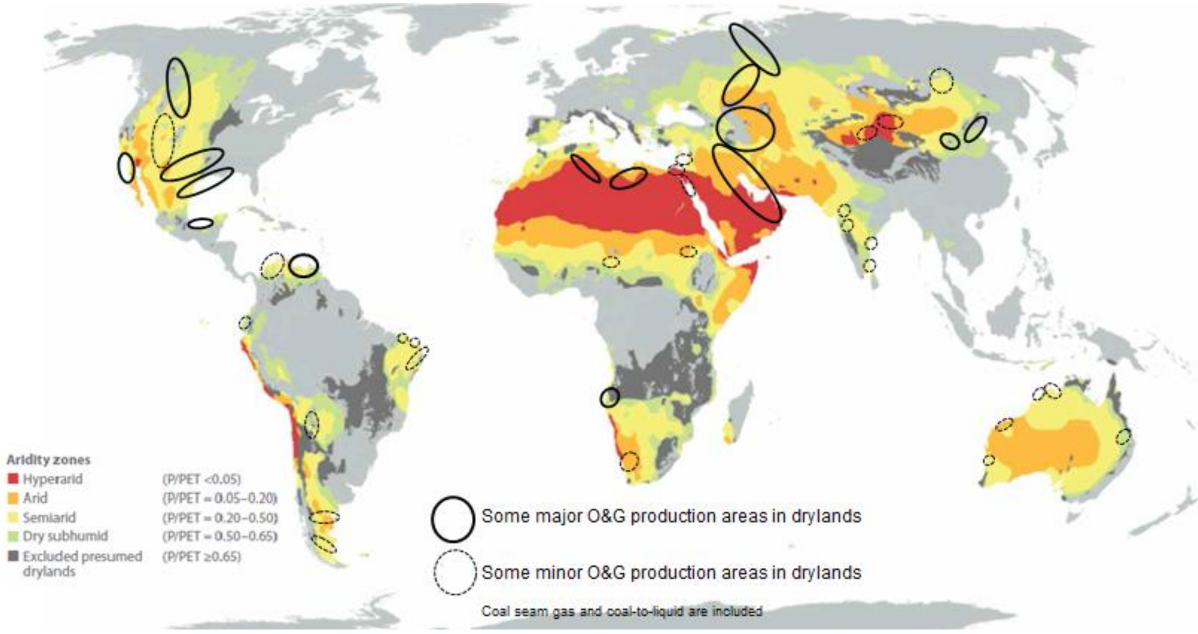


**Figure 3-1 Distribution of drylands and of the main oil and gas production zones located in these areas (adapted from FAO, 2016).**

Drylands occur on all continents (Safriel et al., 2006), cover 41% of the earth's landmass (Adeel et al., 2005) and are projected to expand, partly due to climate change (Feng and Fu, 2013). These regions are inhabited by 2.1 billion people, many of whom live in developing countries and are directly dependent on the land's natural resources (UN, 2010). Projections estimate that half of the global population will live in regions with high water scarcity by 2030 (UN, 2012). Drylands are an important component of the total agricultural land area as well. About 50% of the arid and semi-arid area is used for agriculture (Gratzfeld, 2003), drylands grow 44% of the world's food and support 50% of the world's livestock (Reid, 2014). In drylands, agriculture represents a major economic activity and approximately a third of the population living in these zones depends on agriculture particularly in Africa and in Asia (CGIAR, 2015). Within developed countries, drylands have also significant economic importance. For instance, California represents 13% of the US GDP making this dry state the major contributor to America's national wealth (US Department of Commerce, 2015). California also produces around 70% of the fruit and tree nuts, 55% of the vegetables, 10% of the cotton and about 30% of the rice produced in the USA (US Department of Agriculture, 2015). However, agriculture and populations in drylands are under constant threat of water shortage. In fact, drylands are characterised by physical water scarcity because they are naturally prone to lack of water due to their negative water balance (i.e. low precipitation and high evapotranspiration) (Gassert et al., 2015). In addition, fresh water availability can also be reduced by water pollution (NSW Government, 2011) or seawater intrusion (Qadir and Sato, 2015) which can contaminate the already limited fresh water resources. Climate change is projected to increase water scarcity in most drylands, affecting both rain-fed and irrigated agriculture (Pedrick, 2012). As water resources are diminishing, water users (i.e. industry, agriculture, households and the natural environment) are competing more and more for access to water (El-Zanfaly, 2015; Freyman and Salmon, 2014; Qadir and Sato, 2015).

Therefore, the pressure on water resources from the O&G industry in drylands is expected to intensify and is likely to exacerbate competition and conflicts between water users, and especially between irrigated farming and unconventional O&G firms which use fresh water resources (Galbraith, 2013; Hitaj, Boslett and Weber, 2014). Reusing O&G PW for the irrigation of food crops could contribute considerably to improve the sustainability of irrigated agricultural systems in drylands.

This structured review paper aims to provide a critical review of the potential of O&G PW for the irrigation of food crops in drylands. It starts by providing a review of the volumes and qualities of PW from around the world, followed by a discussion of its treatment and management practices. Finally, the potential for reuse of PW in agriculture is discussed and experiences of irrigation with PW are reviewed in order to identify the main risks associated with using PW in practical conditions. The quality of PW is also discussed from an agricultural viewpoint in order to highlight the agronomic and environmental risks associated with reuse and the perspectives for adapting PW to irrigation.

## 3.2 Volume of produced water

The water-to-oil (WOR) and water-to-gas (WGR) ratios are indicators used to quantify the volume of PW generated compared to the volume of oil or gas produced. Although strictly dimensionless, the O&G industry generally expresses the ratios as barrels (159 L) of water per barrel of oil or million cubic feet of gas. At the world scale, the average WOR was about 3:1 in the 2000s (Khatib and Verbeek, 2002), and is probably nowadays closer to 4:1, but it can locally range from as low as 0.4 to as high as 36 (Table 3-1) depending on the field history, the type of hydrocarbon and the technologies employed (Clark and Veil, 2015). Globally, this ratio has been increasing because conventional O&G fields are ageing so, they produce more and more PW for less hydrocarbons (Healy et al., 2015; Veil et al., 2004). Thus, the highest WOR and WGR are generally related to mature production areas (e.g. California, China and Oman). However, the WOR and WGR of some fields in the Middle East are still low even if they have been operated for several decades due to specific geological and management conditions of these ‘giant fields’ which reach their maturation stage much later than smaller fields (Sorkhabi, 2010; Sorrell et al., 2012).

Significant quantities of PW are generated in dry regions (Table 3-1), although little information is available about volumes of PW in O&G producing countries. Indeed, the only significant O&G producer holding public documented information about PW generation and management is the USA (Clark and Veil, 2009, 2015). Contrary to hydrocarbon production that has a high economic value, PW volume is often not measured and monitored by O&G operators (Clark and Veil, 2009). As a consequence, the data in Table 3-1 are uncertain due to a lack of rigorous reporting and monitoring (Clark and Veil, 2015).

The volume of PW and its evolution over time differ between oil fields and gas fields as oil reservoirs usually contain larger volumes of water than gas reservoirs as gas has a higher compressibility and sorption capacity than oil, and also because gas is stored in less porous reservoirs (Guerra, Dahm and Dundorf, 2011). The volume of PW and wells’ behaviour are also very heterogeneous between the types of production; conventional O&G wells typically show a gradual increase of water production while hydrocarbon production is decreasing (Clark and Veil, 2009; Healy et al., 2015). In contrast, in unconventional O&G

production, the volume of PW tends to be correlated with the volume of hydrocarbons extracted (Healy et al., 2015).

Globally, the estimated quantity of PW has increased by more than 78% between 1990 and 2015 from about 10.6 billion $m^3$ to 18.9 billion $m^3$ compared to 38% growth of the oil production from 3.7 billion $m^3$ to 5.1 billion $m^3$ respectively. This increasing trend is expected to continue as the projected world PW volume is between 29–54 billion $m^3$ in 2020 (Table 3-1).

There is an obvious connection between the increase in WOR and the quantity of PW as illustrated by the situation in North America. Conventional O&G fields in North America are ageing (IEA, 2013); consequently, a significant and continuous increase of PW volume has been observed between 2007 and 2015 from 3.9 to 4.3 billion $m^3$ respectively, it is forecast that 5.6 billion $m^3$ of PW will be generated in 2025 in this part of the world (Shah, 2014). This increase is also partly explained by the rapid development of unconventional hydrocarbons, even if their WOR and WGR are not significantly higher than those of conventional hydrocarbons (Scanlon, Reedy and Nicot, 2014). Most part of the PW is, and will be, generated in relatively dry states and provinces of North America (Guerra, Dahm and Dundorf, 2011).

**Table 3-1 Estimates of water-to-oil ratios (WOR = $m^3$ of produced water/$m^3$ of oil produced), water-to-gas ratios (WGR = $m^3$ of produced water/1000 $m^3$ of gas produced) and total volumes of produced water (PW) by type of production and country or region located in drylands.**

| Country-Region-Field | Type of production | WOR | WGR | PW volume ($m^3$/year) | Year | Reference |
|---|---|---|---|---|---|---|
| World-Total 1990 | All | 2.9 | - | 10 590 541 521 | 1990 | [1; 2; 3] |
| World-Total 2000 | All | 2.9 | - | 12 186 376 545 | 2000 | [1; 2; 3; 4] |
| World-Total 2010 | All | 3.6 | - | 16 886 836 070 | 2010 | [1; 2; 3; 5] |
| World-Total 2015 | All | 3.7 | - | 18 859 868 463 | 2015 | [1; 2; 3] |
| World-Total 2020 (low estimation) | All onshore | 5.6 | - | 29 015 182 250 | 2020 | [1; 6] |
| World-Total 2020 (high estimation) | All | 10.5 | - | 54 020 000 000 | 2020 | [1; 7] |
| USA | All | 10.0 | 0.6 | 3 367 453 720 | 2012 | [8; 9] |
| USA-Texas | All | - | - | 1 182 175 348 | 2012 | [8; 9] |
| USA-Texas-New Mexico-Permian | All | 9 | - | 953 923 800 - | 2014 | [10; 11] |
| USA-California | All | 15.5 | 0.1 | 524 658 090 | 2014 | [9; 12] |
| USA-Wyoming | All | 36.3 | 1.4 | 346 284 674 | 2012 | [9] |
| USA-New Mexico | All | 7.9 | 0.5 | 123 363 016 | 2012 | [9] |
| USA-North Dakota | All | 1.2 | 0.1 | 46 288 675 | 2012 | [9] |
| USA-Montana | All | 6.8 | 0.3 | 29 068 125 | 2012 | [9] |
| USA-Nebraska | All | 23.0 | 3.6 | 9 323 174 | 2012 | [9] |
| USA-Nevada | All | 15.9 | - | 932 461 | 2012 | [9] |
| USA-South Dakota | All | 3.0 | - | 841 997 | 2012 | [9] |
| USA-Arizona | All | 1.3 | 0.3 | 12 878 | 2012 | [9] |
| Canada | All | 11 | - | - | 2010 | [13] |

| Country-Region-Field | Type of production | WOR | WGR | PW volume ($m^3$/year) | Year | Reference |
|---|---|---|---|---|---|---|
| Mexico | All | 3 | - | - | 2010 | [13] |
| China | All | 9 | - | - | 2010 | [13] |
| Australia | All | - | 0.2 | 33 000 000 | 2010 | [14; 15; 16] |
| USA-California-Kern River | Conventional | 15 | - | 52 227 328 | 2005, 2008 | [17; 18] |
| Saudi Arabia | Conventional | 1–3 | - | - | 2010, 2015 | [13; 19] |
| Saudi Arabia-Qatif and Khursaniyah | Conventional | 2.3 | - | - | 2009 | [20] |
| Saudi Arabia-Ghawar | Conventional | 0.4 | - | - | 2003 | [21] |
| Iraq-North Rumaila | Conventional | - | - | 16 828 806–46 424 292 | 2013 | [22] |
| Iraq-Kirkuk | Conventional | 2 | - | - | 2009 | [20] |
| Oman | Conventional | 10.0 | - | 292 000 000 | 2007 | [23] |
| Oman-South Fields | Conventional | 3 | - | - | 2007 | [24] |
| Oman-Nimr | Conventional | 10.0 | - | 98 550 000 | 2009 | [23] |
| Kuwait | Conventional | 0.4 | - | - | 2016 | [25] |
| Qatar | Conventional | 3.0 | | - | 2014, 2016 | [26; 27] |
| USA-Wyoming-Powder River | Coalbed methane | - | 15.4 | 63 531 643 | 2000 | [28] |
| USA-New Mexico-Colorado-San Juan | Coalbed methane | - | 0.2 | 4 481 395 | 2000 | [28] |
| USA-Colorado-Raton | Coalbed methane | - | 7.5 | 7 085 159 | 2000 | [28] |
| USA-Utah-Uinta | Coalbed methane | - | 2.4 | 4 903 276 | 2000 | [28] |
| Australia-Queensland-Surat | Coalbed methane | - | - | 125 000 000 | 2015 | [14] |
| Australia-New South Wales-Sydney | Coalbed methane | - | - | 4800 | 2012 | [29] |
| USA-Texas-Eagle Ford | Shale (tight) | 1.4 | 0.6 | 397 468 250 | 2014 | [30] |
| USA-Colorado | Shale (tight) | 2.5 | - | 56 979 299 | 2012, 2015 | [9; 31] |

| Country-Region-Field | Type of production | WOR | WGR | PW volume ($m^3$/year) | Year | Reference |
|---|---|---|---|---|---|---|
| USA-North Dakota-Montana-Bakken | Shale (tight) | 3 | - | 42 926 571 | 2014, 2015 | [32; 33] |
| USA-Utah | Shale (tight) | 3 | - | 26 542 135 | 2012 | [9; 31] |
| China-Liaoning-Liaohe | Heavy oil | - | - | 7 300 000 | 2011 | [34] |
| Mexico-Maya | Heavy oil | 3 | - | - | 2009 | [20] |
| Canada-Alberta | Oil sands | 0.4–5.0 | - | - | 2010, 2013 | [35; 36] |

[1](BP, 2017); [2](Dal Ferro and Smith, 2007); [3](SPE, 2011); [4](Khatib and Verbeek, 2002); [5](Fakhru'l-Razi et al., 2009); [6](Stanic, 2014); [7](Transparency Market Research, 2016); [8](Burnett, 2004); [9](Clark and Veil, 2015); [10](Digital H2O, 2015); [11](Sharr, 2014); [12](Waterfind, 2016); [13](Jacobs Consultancy, 2010); [14](Commonwealth of Australia, 2014); [15](IESC, 2014); [16](Blackam, 2017); [17](Robles, 2016); [18](Waldron, 2005); [19](Al-Haddabi et al., 2015); [20](Keesom, Unnasch and Moretta, 2009); [21] (Sorkhabi, 2010); [22](Kuraimid, 2013); [23](Breuer, 2011); [24](Al-Mahrooqi, Marketz and Hinai, 2007); [25](Alanezi, 2016); [26](Ahan, 2014); [27](Bin-Hilal Al-Kuwari et al., 2016); [28](Rice and Nuccio, 2000); [29](NSW Government, 2013); [30](Scanlon, Reedy and Nicot, 2014); [31](Gordon, 2015); [32](Kurz et al., 2016); [33](Terrel, 2015); [34](Vaz and Di Falco, 2011); [35](Williams and Simmons, 2013); [36](Miller, 2010)

## 3.3 Quality of produced water

PW contains a mixture of organic and inorganic materials (Table 3-2) including dissolved and dispersed oil, dissolved formation minerals, production chemical compounds, production solids (e.g. formation solids, corrosion and scale products, bacteria, waxes, and asphaltenes), naturally occurring radioactive materials (NORM) and dissolved gases (Deng et al., 2009; Ekins, Vanner and Firebrace, 2007; Fakhru'l-Razi et al., 2009; Hansen and Davies, 1994; McCormack, 2001; Neff, 2002; Neff, Lee and DeBlois, 2011; Stephenson, 1992; Veil et al., 2004; Wang et al., 2001). The detailed chemical composition and physical characteristics of PW partly depend on the type of hydrocarbon associated with PW. For example, PW from gas production usually has lower total dissolved solids (TDS), oil, and grease content than that from oil production. PW quality also differs according to the geology of the storage formation from which they are withdrawn, the operational conditions, the age of the well, and the chemicals used in process facilities (Abousnina, Nghiem and Bundschuh, 2015; Igunnu and Chen, 2014; Neff, Lee and DeBlois, 2011; Pichtel, 2016; Veil et al., 2004). In addition, like the volume, the composition of PW can vary over time within the same well (Veil et al., 2004).

**Table 3-2 Ranges of some physical and chemical parameters of typical oil and gas produced water compared to FAO guidelines for irrigation water and US EPA national discharge standards.**

| | COPW[1; 2; 3] | CGPW[1; 2; 4; 5] | TOPW[1] | SGPW[1; 6; 7] | CBMPW[1; 5; 8; 9; 10; 11] | FAO guidelines[12] or US EPA standards[13] |
|---|---|---|---|---|---|---|
| EC (µS/cm) | 621–359 000 | 621–359 000 | 78 400–373 400 | 0.03–763 000 | 9–40 380 | 0 < SAR < 3 if EC > 0.7 |
| SAR | 1–3759 | - | 430–1014 | 2–1497 | 4–1567 | 3 < SAR < 6 if EC > 1.26 |
| | | | | | | 6 < SAR < 12 if EC > 1.9 |
| | | | | | | 12 < SAR < 20 if EC > 2.9 |
| | | | | | | 20 < SAR < 40 if EC > 5 |
| pH | 4.3–10.0 | 3.1–7.0 | 3.9–11.2 | 3.2–11.8 | 5.4–10.4 | 6.5–8.4 |
| TDS (mg/L) | 80–472 000 | 4802–310 000 | 1517–349 056 | 35–358 000 | 150–177 000 | 0–3200 |
| $Cl^-$ (mg/L) | 80–292 000 | 3000–200 000 | 1–310 561 | 1–196 000 | 0.8–110 000 | 0–1050 |
| $HCO_3^-$ (mg/L) | 77–3990 | 100–6000 | 0.6–18 916 | 0.01–13 880 | 19–43 310 | 0–8.5 |
| $SO_4^{2-}$ (mg/L) | < 2–1650 | BDL–5000 | 0.7–11 300 | 0.1–3580 | BDL–1800 | 0–960 |
| $NO_3^-$ (mg/L) | - | - | - | - | 0.01 | 0–30 |
| $PO_4^{3-}$ (mg/L) | - | - | - | 0.03–51 | BDL–9199 | 0–2 |
| Na (mg/L) | 122 000 | 2000–100 000 | 49.9–124 400 | 3.6–434 403 | 2.6–51 700 | 0–920 |
| K (mg/L) | 24–4300 | BDL–750 | 7–8526 | 2–17 043 | 0.1–20 100 | 0–2 |
| Ca (mg/L) | 13–42 800 | 24 | 10–132 687 | 1.95–162 324 | 0.42–13 900 | 400 |
| Mg (mg/L) | 8–8,350 | BDL–2000 | 1–26 666 | 0.1–5747 | 0.01–15 | 60 |
| Al (mg/L) | 310–410 | BDL–83 | 0.09 | 0.04–2 | 0.01–3 | 0–5 |
| B (mg/L) | 5–95 | BDL–56 | 63–564 | 0.01–155 | 0.05–10 | 0–3 |
| Cd (mg/L) | < 0.005–0.2 | BDL–0.015 | 0.024–0.067 | 0.001–0.1 | 0.0001–1.4 | 0–0.01 |

| | COPW[1; 2; 3] | CGPW[1; 2; 4; 5] | TOPW[1] | SGPW[1; 6; 7] | CBMPW[1; 5; 8; 9; 10; 11] | FAO guidelines[12] or US EPA standards[13] |
|---|---|---|---|---|---|---|
| Cr (mg/L) | 0.02–1.1 | BDL–0.03 | 0.045–318 | 0.001–14 | 0.001–3.7 | 0–0.1 |
| Cu (mg/L) | < 0.002–1.5 | BDL–5 | 0.009–1.5 | 0.01–2.6 | 0.002–4.6 | 0–0.2 |
| Fe (mg/L) | < 0.1–100 | BDL–1100 | 0.05–800 | 0.18–1247 | 0.005–4180 | 0–5 |
| Li (mg/L) | 3–50 | 19–235 | 7.1–90.1 | 0.009–426 | BDL–36 | 0–2.5 |
| Mn (mg/L) | < 0.004–175 | 0.04–1 | 1.54–29.4 | 0.01–24 | 0.0018–6 | 0–0.2 |
| Ni (mg/L) | < 0.001–1.7 | BDL–9.2 | 0.183–0.397 | BDL–36.5 | 0.0001–19.2 | 0–0.2 |
| Pb (mg/L) | 0.002–8.8 | < 0.02–10.2 | 0.006–1.210 | 0.001–0.7 | 0.001–0.2 | 0–5 |
| Zn (mg/L) | < 0.01–35 | BDL–5 | 0.134–29 | BDL–182 | 0.001–51 | 0–2 |
| Oil and grease (mg/L) | 0.565 | 0.29–38.8 | - | - | 2.2 | 35[13] |

COPW: Conventional oil produced water; CGPW: Conventional gas produced water; TOPW: Tight oil produced water SGPW: Shale gas produced water; CBMPW: Coalbed methane produced water; BDL: Below Detection Level.

[1](Blondes et al., 2017); [2](Engle, Cozzarelli and Smith, 2014); [3](Pichtel, 2016); [4](Fakhru'l-Razi et al., 2009); [5](Xu, Drewes and Heil, 2008); [6](Alleman, 2011); [7](Maguire-Boyle and Barron, 2014); [8](Abousnina, Nghiem and Bundschuh, 2015); [9](Commonwealth of Australia, 2014); [10](Jackson and Myers, 2002); [11](Khan and Kordek, 2013); [12](Ayers and Westcot, 1985); [13](US EPA, 1995)

As we see in Table 3-2 the ranges of chemical concentration in the different kinds of O&G PW vary widely. From an agronomic point of view, PW typically has high TDS, high electrical conductivity (EC), high sodium adsorption ratio (SAR), acidic to alkaline pH. PW also contains moderate to high amounts of various heavy metals such as B, Cd, Cr, Cu, Pb, Ni and Zn (ALL Consulting, 2003; Clark and Veil, 2009; Hansen and Davies, 1994; Pichtel, 2016; Van Voast, 2003).

## 3.4 Management of produced water

Due to its complex composition, PW needs to be managed in order to avoid environmental damage. Treatment and reuse or disposal options depend on the constituents of PW, the location of the oil or gas field (e.g. onshore or offshore) and the environmental regulation of the territory where the hydrocarbon is produced. For example, oil and grease receive the most attention for both onshore and offshore PW, whereas salt content is of concern for onshore PW.

### 3.4.1 Treatment

The treatment options include de-oiling, desalination, degassing, suspended solids removal, organic compounds removal, heavy metal and radionuclides removal, and disinfection (SPE, 2011). These treatment goals are essentially the same for beneficial reuse or disposal, although the level of contaminant removal required for reuse in irrigation can be significantly higher, depending on the original quality of the PW and the type of reuse. Achieving the various treatment goals requires the use of multiple treatment technologies, including physical, chemical and biological treatment processes (Fakhru'l-Razi et al., 2009). The treatment cost strongly depends on the quality of PW (which can vary widely among production fields and change over time within a given field) and the regulatory environment. Therefore, technology solutions for treatment and reuse of PW would need to be adapted according to the properties of the PW and the amount of water to be treated (SPE, 2011).

### 3.4.2 Management options

The final destination of the PW (i.e. disposal or reuse) is highly dependent on its quality and also the location of the O&G field. Table 3-3 shows that most PW is reinjected into underground formations. When used to improve oil recovery, PW ceases to be a waste and becomes a useful resource. Surface discharge is the second most common practice while reinjection into disposal wells is the third. In these cases, PW is not used in a beneficial way and is considered as a waste. PW reuse (other than reuse for enhanced oil recovery) remains a minor practice although it is expected to develop in the future due to the reuse of higher proportion of PW that is currently discharged to the surface and reinjected for disposal (Global Water Intelligence, 2014). Despite the projected increase in PW volume, the shares of non-beneficial uses of PW (disposal and discharge) will decrease compared to beneficial uses (enhanced oil recovery and other beneficial reuses).

**Table 3-3 Global oil and gas produced water management practices in 2012 compared to 2020 forecast after Global Water Intelligence (2014)**

| Management option | Share of PW volume in 2012 (%) | Expected share of PW volume in 2020 (%) |
|---|---|---|
| Reinjection for enhanced oil recovery | 52 | 56 |
| Reinjection for disposal | 19 | 15 |
| Surface discharge | 21 | 17 |
| Other non-beneficial practices | 5 | 5 |
| Beneficial reuse | 3 | 7 |

Management practices vary between regions. In the USA for instance, in 2007, about 95% of the PW was managed through underground injection practices (i.e. 55% for enhanced oil recovery and 39% for disposal), the remaining 5% of water was discharged to surface water, stored in surface impoundments, reused for irrigation, or reused for hydraulic fracturing (Clark and Veil, 2009; Hladik, Focazio and Engle, 2014).

Management practices also differ between onshore and offshore fields. Most onshore O&G PW is reinjected whilst offshore O&G PW tends to be discharged, due to the isolation of offshore O&G facilities from potential reuse options. Indeed, globally, in 2014, an estimated 844 million $m^3$ of PW were discharged offshore (IOGP, 2014) representing 84% of the total volume of offshore PW in 2013 (Water Online, 2014). The variability of offshore PW management practices is less compared to onshore PW. For example, the estimated total volume of PW generated in the USA's federal waters in 2007 was about 93 million $m^3$, 91% was treated and discharged to the ocean and only 9% of this PW was reinjected underground for enhanced recovery or disposal (Clark and Veil, 2009). In Europe's offshore waters (mainly the North Sea), about 419 million $m^3$ of PW were discharged in 2014 whereas about 100 million $m^3$ were reinjected in 2012 (Garland, 2005; IOGP, 2014).

PW that is discharged, disposed of, and not used beneficially represented 45% of global PW volume in 2012 (Table 3-3). Thus, considering the 18.86 billion $m^3$ of PW generated in 2015 (Table 3-1), about 8.5 billion $m^3$ of PW is potentially available for agricultural irrigation.

## 3.5 Potential of produced water for reuse in irrigation

### 3.5.1 Experience of irrigation with oil and gas produced water

Among the possible beneficial reuses of PW, agricultural irrigation (especially of food crops) could be particularly relevant in drylands. Table 3-4 presents theoretical research, laboratory and field experiments, as well as examples of large-scale use of PW for irrigation in different parts of the world. Table 3-4 helps to identify the challenges faced when PW is used for irrigation in dry zones. It also supports the idea that PW in conjunction with adapted management has an important potential to increase water resources in drylands.

**Table 3-4 Cases of irrigation of food crops and non-food crops with oil and gas produced water and main outcomes**

| Country (Region) | Type of O&G field associated to the PW used | Water treatment | Quality of the water applied | Soil type | Soil amendments applied | Crop irrigated | Main observations | Reference |
|---|---|---|---|---|---|---|---|---|
| USA (Wyoming) | Conventional oil field PW | Untreated | TDS = 3220 mg/L<br>Na = 642 mg/L<br>SAR = 9.79 | Soilless cultivation (hydroponic) | Fertilisers: $KNO_3$; $Ca(NO_3)_2$; $MgSO_4$<br><br>pH regulator: $H_2SO_4$ | Tomato | Yield reduction (3 times lower compared to control).<br>More Na and metals absorption by plants than in control. | 1 |
| USA (Wyoming) | CBM PW | Untreated | TDS = 1390 mg/L<br>Na = 555 mg/L<br>SAR = 5.73 | Clay loam | Fertilisers: NPK (18-6-12) | Corn, switchgrass, spearmint, Japanese corn mint, lemongrass, common wormwood | Increase Na and decrease $Ca^{2+}$ and $Mg^{2+}$ concentrations in soil.<br>Elevated leaf Na content in plant.<br>Untreated CBM PW can be used for short periods (2 years). | 2 |
| USA (Alabama) | CBM PW | Blending with fresh water | EC = 10 600 µS/cm<br>TDS = 6780 mg/L<br>SAR = 73 | Sand = 28.9 %<br>Silt = 50.5 %<br>Clay = 20.6 % | Fertilisers: N (30 mg/kg of soil) | Sorghum-sudangrass | CBM PW (TDS = 2000 mg/L) can be applied to highly weathered soils.<br>Plant growth of summer annual grasses will be optimised if an irrigation system is used to apply PW at a rate to maintain soil moisture at or near field capacity. | 3 |
| USA (California) | Conventional oil field PW | Mechanical separation, sedimentation, air flotation and filtration | TDS = 500 mg/L<br>Na = 130 mg/L | Saline-alkaline soils with diverse texture | - | Grape, almond, citrus, pistachio, apple, peach, plum, melon, potato, vegetables | Trace of organic chemical below drinking standards. Water considered safe for irrigation. | 4; 5: 6 |

| Country (Region) | Type of O&G field associated to the PW used | Water treatment | Quality of the water applied | Soil type | Soil amendments applied | Crop irrigated | Main observations | Reference |
|---|---|---|---|---|---|---|---|---|
| Oman | Conventional oil field PW | Reed, solar distillation | TDS ≤ 50 mg/L | - | None | Eucalyptus, Kuwaiti tree, seashore paspalum, cotton | The PW is desalinised using a commercial solar powered system called 'Solar Dew' which is especially adapted to arid environments. The desalination cost 0.5–2 USD/$m^3$ is thus much lower compared to an electric or fuel-powered desalination unit. After treatment by reeds, the PW is saline (TDS = 6980 mg/L). The solar desalination system produced an effluent reaching WHO potable standards (TDS ≤ 50 mg/L). | [7; 8; 9] |
| Oman | Conventional oil field PW | Air flotation, anthracite filtration, activated carbon | EC = 8 dS/cm<br>TDS = 3000–6000 mg/L | Mixture of gravel (top layer 8 cm), sand (40 cm) and OM | None except than OM initially added to create an experimental soil | Alfalfa, barley, Rhodes grass | Increased soil salinity and sodicity.<br>Decrease of soil salinity when low-salinity water is frequently used to leach salts. | [10] |
| Mexico | Conventional oil field PW | Dilution with fresh water | EC = 1.1–1.2 dS/cm<br>TDS = 726–769 mg/L<br>Na = 100–103 mg/L<br>SAR = 2.85–2.92 | Pots of peat moss and perlite substrate (3:1) | Nutrient solution is applied but its composition is not detailed | Tomato | Raw PW is unsuitable for irrigation due to the high levels of EC.<br>Diluted PW with fresh water to adjust the EC to 1500 µS/cm is suitable for irrigation of tomato under greenhouse conditions. | [11] |
| Qatar | Conventional gas field PW | - | TDS = 162–179 mg/L<br>Na = 2.8–3.3 mg/L<br>SAR = 0.34–0.35<br>EC = 0.3–3 dS/cm | Sand = 87 %<br>Silt = 2 %<br>Clay = 11 %<br>OM = 4.3 % | None | Alfalfa | The fresh weight of the plant was significantly reduced at irrigation with gas PW. Crude fiber was significantly higher. Gas PW can result in a reasonable production with acceptable quality. | [12] |

| Country (Region) | Type of O&G field associated to the PW used | Water treatment | Quality of the water applied | Soil type | Soil amendments applied | Crop irrigated | Main observations | Reference |
|---|---|---|---|---|---|---|---|---|
| Yemen | Conventional oil field PW | Constructed wetland (reed bed) | NaCl = 15 000 mg/L | Clayed-sandy | None | Cotton and hemp | Hemp was affected by salinity but not cotton | 13 |

CBM: Coalbed Methane, COD: Chemical Oxygen Demand, EC: Electrical Conductivity, OM: Organic Matter, SAR: Sodium Adsorption Ratio, TDS: Total Dissolved Solids.

[1](Jackson and Myers, 2002); [2](Burkhardt et al., 2015); [3](Mullins and Hajek, 1998); [4](Cawelo Water District, 2015); [5](Heberger and Donnelly, 2015); [6](Robles, 2016); [7](Breuer, 2017); [8](Breuer, 2011); [9](Sluijterman et al., 2004); [10](Hirayama et al., 2002); [11](Martel-Valles et al., 2014); [12](Ibrahim, Marroff and Wafi, 2009); [13](Rambeau et al., 2004)

### 3.5.2 Agro-environmental risks associated with irrigation with oil and gas produced water

The concentration ranges of salts (measured through TDS and EC) particularly sodium and some heavy metals (Al–Zn) are very often over the values recommended by the FAO guidelines that we use as a reference for the quality of irrigation water (Table 3-2) (Alley et al., 2011; Ayers and Westcot, 1985). These components remain in high concentration even after conventional treatment, which mainly targets organic pollutants (Fakhru'l-Razi et al., 2009). The other components of PW represent lower risks to the soil because they are either initially present in low concentrations (e.g. nutrients and radioactive elements) or their concentrations are highly reduced during treatment processes and are particularly targeted by regulation (e.g. hydrocarbons) (Fakhru'l-Razi et al., 2009). Thus, hydrocarbons represent a minor hazard for soil compared to salts and heavy metals. Indeed, oil and grease concentration in most documented PW is quite low compared to US EPA standards for agricultural use of PW (Table 3-2). PW that could be reused at a large scale would otherwise be disposed or discharged into the environment and would therefore be treated up to tertiary level, having a final oil and grease concentration below 10 mg/L (SPE, 2011); which is also below US EPA standards. In addition, hydrocarbons do not tend to accumulate in the long term as salts or metals do, this is because of their organic nature enabling biological degradation in soil (Pichtel, 2016).

As a result, the challenging components of PW remain in dissolved formation minerals (i.e. salts and sodium) and heavy metals. If PW is used in agricultural irrigation, these elements can accumulate in the soil; creating risks of soil salinisation and sodification as observed in most case studies (Table 3-4). These risks are not specific to PW but they are also related to irrigation with both municipal and industrial wastewaters that are often saline and sodic (Elgallal, Fletcher and Evans, 2016; Maassen, 2016).

#### 3.5.2.1 Risks related to the salinity and sodicity of produced water

Generally, salinity and sodicity are closely linked because the main ions in PW are sodium ($Na^+$) and chloride ($Cl^-$). Other cations such as $K^+$, $Ca^{2+}$, $Mg^{2+}$, $Ba^{2+}$, $Sr^{2+}$, $Fe^{2+}$ and anions like $SO_4^{2-}$, $CO_3^{2-}$, $HCO_3^-$ also affect PW salinity and buffering capacity (Hansen and Davies, 1994), but at a lower scale than $Na^+$ and $Cl^-$ due to lower concentrations in PW. However, on some sites that use seawater for enhanced oil recovery, $SO_4^{2-}$ concentration is high and contribute significantly to PW salinity (Neff, 2002). The salt concentration of most PW varies from 1000 to 300 000 mg/L classifying it between 'slightly saline' to 'brine' (Jacobs et al., 1992; Rhoades, Kandiah and Mashali, 1992).

The misuse of PW in irrigation can increase soil salinity and sodicity to unsustainable levels for crops and soil's health even on a short term (Burkhardt et al., 2015; Hirayama et al., 2002; Rambeau et al., 2004) (Table 3-4).

Excessive salinity and sodicity of PW used for irrigation can dramatically and irreversibly alter soil structure in drylands. Salt accumulates in soil, particularly in the root zone, as a result of high rates of evaporation and low precipitation (Burkhardt et al., 2015; Elgallal, Fletcher and Evans, 2016; Safriel et al., 2006). The build-up of salt could lead to elevated levels of exchangeable sodium and SAR in soil if $Na^{+}$ is the dominant ion (Ayers and Westcot, 1985; Beletse et al., 2008; Johnston, Vance and Ganjegunte, 2008; Stefanakis, 2016; Toze, 2006) causing a decrease in water infiltration and dispersion of clay which destroy clay-humus complex and finally lead to possible nutrient deficiencies, such as $Ca^{2+}$ and $Mg^{2+}$, which are displaced by the high $Na^{+}$ content, or unavailable because the roots cannot penetrate into the subsurface (Hillel, 2003). A vicious circle can set up once soils are sodic. Indeed, when sodic soils are wet, they become sticky, and when they dry, they form a crusty layer that is nearly impermeable. Then more water is lost due to evaporation or runoff and salts accumulate even more in the topsoil, this worsens salinity and sodicity problems. Elevated salinity affects the ability of plants to uptake water to facilitate biochemical processes such as photosynthesis and plant growth (Vance, King and Ganjegunte, 2008).

For example, a 2-year study conducted in the Powder River Basin (USA) showed that irrigation with untreated CBM PW increased soil sodicity from 1.4 to 2.8 mmol/L (measured on a saturated extract) while concentrations of $Ca^{2+}$ and $Mg^{2+}$ decreased reaching levels that are potentially toxic to the crop (Burkhardt et al., 2015). Another study in the same area showed that CBM PW increased the soil EC about two-fold compared to pre-irrigation level (Johnston, Vance and Ganjegunte, 2008). Similar results were observed in Alabama (USA) where CBM PW was used continuously for 30 days to irrigate sorghum and sudangrass. The exchangeable Na percentage reached 40% indicating that long-term use of CBM PW could lead to degradation of soil physical properties (Mullins and Hajek, 1998). In Oman, irrigation with conventional oil field PW increased soil EC from 1.63 to 7.08 dS/m after 102 days of irrigation although fresh water was used at a regular frequency (28 days totally) to leach salts, in the meantime, the SAR increased dramatically from 2.31 to 68.10 (Hirayama et al., 2002).

#### 3.5.2.2 Risks related to heavy metals of produced water

Heavy metals are generally studied because of their impact on human health and on the environment, although some metals such as boron are known to be phytotoxic at high concentration and are therefore an agronomic issue too (Qadir and Drechsel, 2016; Tal, 2016). Heavy metals do not biodegrade like organic pollutants, they chemically and physically interact with naturally occurring substances, which alter their mobility. In fact, some heavy metals are adsorbed, or bound to other particles, reducing their chance of migration or absorption into plants. The degree to which different heavy metals are immobilised in the soil is determined by the natural composition of the soil, pH, water content, and temperature (Dube et al., 2001) although still not completely documented (Pedrero et al., 2010). Heavy metals concentrate in plants (particularly leafy vegetables) and can transfer into the food chain posing a threat to humans (Farrag, Elbastamy and Ramadan, 2016; Rattan et al., 2005). There is evidence of accumulation of Cu and Zn in soil using PW for irrigation in Qatar (Ibrahim, Marroff and Wafi, 2009).

### 3.5.3 Adapting produced water to irrigation

From an agronomic perspective, soil salinisation and sodification are critical as they can immediately impact soil structure and fertility because of the high loads of salt brought by irrigation with saline-sodic PW. In contrast, heavy metals concentrations in PW may create problems of toxicity to plants over a longer term (Table 3-2). Therefore, in order to use PW for irrigation in dry areas, the water salinity and sodicity have to be within the suitable EC-SAR ranges described in Table 3-2. Figure 3-2 shows that a limited proportion of PW can be used without reduction of their salinity (EC) and sodicity (SAR), indeed, over 474 samples of PW collected in the USA, Australia, South Africa and Qatar, only 8.4% of PW samples meet the requirements for being used in irrigation, of which only 10% meet the requirements for unrestricted irrigation.

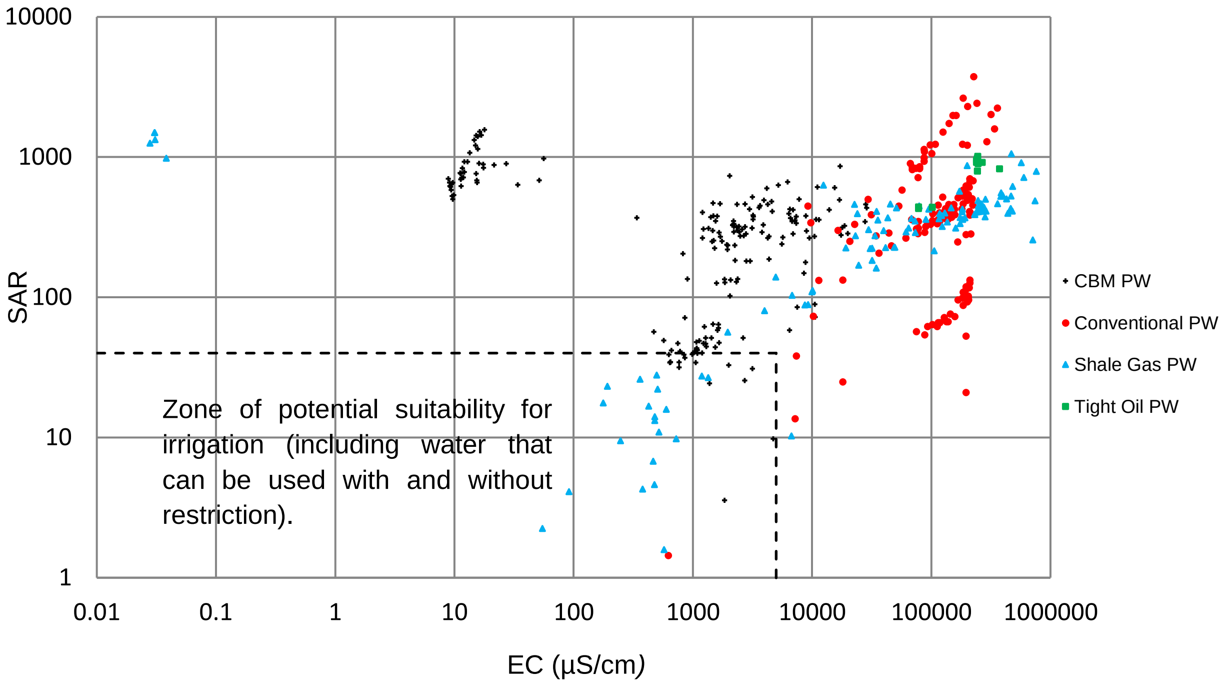


**Figure 3-2 Sodicity (SAR) and salinity (EC) of 474 samples of PW associated to different hydrocarbon types (CBM, conventional shale gas and tight oil) compared to irrigation water quality guidelines based on salinity and sodicity hazard**

adapted from (ALL Consulting, 2003; Ayers and Westcot, 1985; Beletse et al., 2008; Blondes et al., 2017; Brown et al., 2010; Burkhardt et al., 2015; Dresel and Rose, 2010; Ganjegunte, Vance and King, 2005; Jackson and Myers, 2002; Janson et al., 2015; Johnston, Vance and Ganjegunte, 2007; Mullins and Hajek, 1998; Myers, 2014; Szép and Kohlheb, 2010; Xu, Drewes and Heil, 2008).

Although most PW cannot be sustainably used for irrigation, there are solutions for reducing EC and SAR of PW in order to use it for irrigation. Blending of PW with low salinity water and PW desalination using reverse osmosis are the two principal solutions commonly cited in the literature (Fisher et al., 2010; Guerra, Dahm and Dundorf, 2011; Hagström et al., 2016; Jakubowski et al., 2014; Sullivan Graham, Jakle and Martin, 2015; Xu, Drewes and Heil, 2008).

In California, the oil firm Chevron supplies Cawelo Water District with 44 million m$^3$ of treated PW which is then blended with fresh water to irrigate18,600 ha of food crops (Arnold et al., 2004; Heberger and Donnelly, 2015; Martel-Valles, Foroughbakchk-Pournavab and Benavides-Mendoza, 2016). Another study in the Powder River Basin (USA) showed that PW is suitable for irrigation when mixed with fresh water in 1:3 ratio (Burkhardt et al., 2015). PW blending does not necessarily require a source of high-quality-fresh water. Treated municipal sewage, for example, can be mixed with PW to obtain water suitable for irrigation.

Desalination can also be used to reduce PW salinity and sodicity. In the USA, CBM PW has been treated to irrigation standards using ultra-low pressure reverse osmosis (ULPRO) at an estimated cost of \$0.24/$m^3$ (Xu, Drewes and Heil, 2008). Although desalination cost has always been a limitation for using desalinated water in irrigation, the value of water resources increases with water scarcity (Maton et al., 2010). Thus, in dry regions with developed economies, such as the Gulf States, Israel and Spain, desalination could be justified for high-value crops (Burn et al., 2015). Moreover, treating relatively low salinity PW instead of more saline alternatives (e.g. brackish groundwater or seawater) might be economic (Kaner et al., 2017; Qadir et al., 2007).

In addition to reducing the salinity and sodicity of PW, soil and crop management can be adapted to be more resilient against the risks of soil salinisation and sodification. Selecting salt-tolerant crops was found to be the principal factor for the sustainability of wastewater irrigation (Ayers and Westcot, 1985; Maas and Grattan, 1999). Suitable crops should also demonstrate a good marketing value in order to compensate the associated costs of using PW (Fonseca et al., 2007).

Soil ameliorants help to counter undesirable effects of salinity and sodicity of PW. In fact, irrigation with PW in combination with gypsum ($CaSO_4$) and sulphur increase the sulphate content of the soil, helping to mitigate soil dispersion by $Na^+$ (Johnston, Vance and Ganjegunte, 2008). These soil ameliorants individually and/or in combination are used in Australia and in the USA for CBM PW application to agricultural croplands and grasslands (Biggs et al., 2013; Fisher et al., 2010). Gypsum is used as a surface soil ameliorant to increase the level of $Ca^{2+}$ in the system (Amezketa, Aragüés and Gazol, 2005; Guerra, Dahm and Dundorf, 2011; Mace, Amrhein and Oster, 1999). Sulphur is used as a surface soil ameliorant to decrease soil pH and enhance calcite ($CaCO_3$) dissolution to release $Ca^{2+}$ into the soil solution to counter $Na^+$ (Johnston, Vance and Ganjegunte, 2008). The addition of significant organic amendments such as poultry manure (rich in calcium) can contribute to re-balance the SAR (Pichtel, 2016). Other types of soil improvers may prove to be beneficial in treating soil irrigated with PW. For example, use of synthetic polymers (e.g., polyacrylamides) to stabilise aggregates has proved to be successful in improving the physical properties of sodic soils (Alberta Environmental Sciences Division, 2001; Sumner, 1993).

Soil dilution may relieve salinity problems following the release of PW. Indeed, in arid and semi-arid climates, contaminants tend to accumulate in the topsoil. Mixing of the less-contaminated deeper soil with the surface soil can result in dilution of contaminants (Wolf, Brye and Gbur, 2015).

Leaching salts below the root zone helps to control soil salinity. It also contributes to the restoration of the SAR to a suitable range of values by leaching excess sodium (Johnston, Vance and Ganjegunte, 2008). The volume of water and the frequency of leaching fractions depend on the PW quality, crop and climate.

Combining leaching and soil ameliorants (sulphur burners) has been proved to be efficient to stabilise soil sodicity when CBM PW has been used for irrigation (Vance, King and Ganjegunte, 2008).

## 3.6 Conclusions

A significant part of current and forecast volumes of PW will be produced in drylands where water scarcity demands alternative irrigation water sources. PW could be an effective resource in drylands; indeed, at the global scale, about 45% of PW is discharged, disposed of, or not reused in a beneficial way. However, quality remains the principal challenge for the reuse of this massive quantity of PW in irrigation. In fact, most PW are high in salts ([TDS] = 35–472 000 mg/L) and sodium ([Na] = 3–435 000 mg/L). As a consequence, the main risks for the soil of using PW in irrigation are soil salinisation and sodification as observed in the reviewed experiences of irrigation with PW. Nonetheless, these issues are not unique to PW, and dryland farming is often prone to challenges in soil salinity management.

Of the PW samples from around the world summarised in this paper, only a limited proportion (8.4%) were potentially suitable for irrigation in terms of EC-SAR, and for most PW, water treatment, water blending and/or farm-based management techniques would be required to mitigate the risks of soil degradation. The costs of achieving the desired water quality will be very site-specific and will depend, for example, on the PW quality, the cost of energy, and the opportunity cost and availability of alternative water supplies. Similarly, the benefit of using PW for irrigation will depend on the local market for the crop produced and the cost of alternative PW disposal methods. However, in arid areas, where alternative water sources are not available and where the desalination industry is well established with competitive costs, using treated PW to produce an economic output may provide social, economic and environmental advantages over alternative methods of disposal.

Although well-documented studies exist, they are often limited to particular cases (e.g. field experiments in specific locations with their specific soils, climates and economic backgrounds) and cannot easily be extrapolated to world drylands. Also, the reuse of PW for the irrigation of food crops is still not widely considered compared to non-food crops, although food crops could be a resource of primary interest in drylands. Further integrated research is necessary regarding the understanding of the sustainability of food crop irrigation with PW in drylands including its economic feasibility.

## 3.7 References

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# 4 SELECTION AND TESTING OF A MATHEMATICAL MODEL FOR LONG-TERM PREDICTION OF SOIL SALINITY AND SODICITY AS A RESULT OF IRRIGATION

## 4.1 Introduction

The agro-environmental sustainability of irrigation with PW in drylands mainly depends on the severity of soil and groundwater salinisation and sodification. A specific model is needed to predict these phenomena and assess the agro-environmental sustainability of reusing PW of different quality, on different soil types, and in different climates representative of drylands. Ultimately, the model would be used to identify agro-environmentally sustainable irrigation strategies with PW in a regional (Chapter 6) and in an industrial context (Chapter 7).

The aim of this chapter is to select, characterise and test a soil-water model predicting the long-term impacts of irrigation with PW on soil and groundwater salinity and sodicity in drylands.

## 4.2 Agro-environmental sustainability indicators

The long-term soil and groundwater salinisation and sodification need to be quantified. This can be done through specific agro-environmental indicators:

- The electrical conductivity of the soil saturation extract ($EC_e$)

Soil salinity refers to the presence of the major dissolved inorganic solutes (i.e. essentially $Na^+$, $Mg^{2+}$, $Ca^{2+}$, $K^+$, $Cl^-$, $SO_4^{2-}$, $HCO_3^-$, $NO_3^-$ and $CO_3^{2-}$) in the soil-water solution (Rhoades, Chanduvi and Lesch, 1999). Salinity is quantified in terms of the total concentration of such soluble salts, or more practically, in terms of the $EC_e$ as both parameters are closely related (Allison et al., 1954). This close correlation between soluble salts and the $EC_e$, the easiness and rapidity of measuring the $EC_e$ makes the latter commonly used as an expression of soil salinity (Ezlit, Smith and Raine, 2010). Thus, as the $EC_e$ is frequently used, using it in this research would facilitate comparisons with other research results.

- The sodium adsorption ratio of the soil saturation extract ($SAR_e$)

The soil sodicity is usually described in terms of the relative proportion of sodium cations ($Na^+$), compared to the divalent cations $Ca^{2+}$ and $Mg^{2+}$ in solution. The $SAR_e$

is one of the most appropriate indicators to evaluate the negative $Na^+$ effect along with the exchangeable sodium content (ESC) (Cook and Muller, 1997). However, the use of the ESC as an indicator of the level of sodicity instead of the $SAR_e$ or the exchange sodium percentage (ESP) is still limited in the literature. In addition, the $SAR_e$ requires fewer parameters to be calculated (i.e. the soil contents in $Na^+$, $Ca^{2+}$ and $Mg^{2+}$), and can be determined from the same soil-water extract than the one used to evaluate the $EC_e$ (Qadir and Schubert, 2002). Additionally, the SAR and the EC of soil or of irrigation water are often used conjunctly to characterise soil and irrigation water quality (Ayers and Westcot, 1985; Shaw et al., 2011). For the same reasons as for the $EC_e$, the wide use of the $SAR_e$ as a soil sodicity indicator in other research also facilitates the comparison between the results obtained in this research and those available in the literature.

- The pH ($pH_e$) and alkalinity ($Alk_e$) of the soil saturation extract

Additionally, the $pH_e$ and the $Alk_e$ can be used as complementary indicators to assess the sodicity hazard and therefore, the risk of soil structural stability degradation. Indeed, an increase in $pH_e$ may enhance the preference of $Na^+$ to be adsorbed on clay colloids and therefore, increase clay deflocculation, and thus soil structure destabilisation (Ezlit, Smith and Raine, 2010). The $pH_e$ is a common indicator for measuring the acidity in the soil (Allison et al., 1954; Rhoades, Chanduvi and Lesch, 1999). Soil alkalinity indicates the soil pH buffer capacity, that is, the ability of the soil to resist pH change when challenged by acid or alkaline inputs such as irrigation water and soil amendments (Hillel, 2000).

- The electrical conductivity of the drainage water ($EC_d$)

The $EC_d$ is used to quantify the drainage water salinity, and thus, the possibility of impacting the groundwater EC when drainage water reaches the aquifer.

- The sodium adsorption ratio of the drainage water ($SAR_d$)

The $SAR_d$ is used to quantify the drainage water sodicity, and therefore, the possibility of changing the groundwater SAR when drainage water reaches the aquifer.

## 4.3 Model selection criteria

Considering the wide range of modelling approaches and the large number of soil-water models, selection criteria were established to screen candidate models and select the most relevant for this study.

There are essential criteria that the model must have to match the research objectives:

- Model purpose: the model must simulate the evolution of the agro-environmental sustainability indicators at the field scale as a function of irrigation water quality (i.e. water EC and SAR), irrigation management (i.e. volume of water applied and frequency of irrigation), type of climates (i.e. from hyper-arid to dry sub-humid), type of soils (i.e. texture and pH) and type of crop (i.e. annual and perennial).
- Spatial scale: the model should operate at the soil profile scale and in the root zone depth (0–100 cm); the depth where soil salinisation affects crops (Hussain et al., 2002);
- Time scale: the model should reflect the long-term conditions without a fixed time limit;
- Applicability: the model must be applicable to dry climates (i.e. 0 ≤ aridity index $= \frac{Precipitation}{Evapotranspiration} \leq 0.65$). The model should also be applicable to different soil types (i.e. texture, layers and depth). The model should be applicable to different crop types (i.e. annual and perennial);
- Balance between components: the agro-environmental sustainability indicators need to be modelled with the same degree of accuracy as these parameters are closely linked to assess the level of soil and groundwater degradation resulting from irrigation;
- Data requirements: the model does not require input data to be collected on the field and can be run with limited inputs obtained from the literature. This is a crucial point as it is not possible to carry out field sampling in the diverse environments that need to be simulated;
- Acceptance: the model must have been peer-reviewed in published articles. This is to ensure that the model has received critics from the research community and has been applied with relative success;
- Usability: the model must be freely available (i.e. open access);

- Target user: the model must be simple to use without having to attend a specific training;
- Run-time and system requirements: the model should run under the Microsoft Windows operating system and require reasonable computer performance (i.e. processor speed ≤ 3 GHz);
- Model documentation: the model running steps should be clearly explained either in a user manual or in peer-reviewed papers.

Also, there are desirable criteria that would make the model more powerful for reaching the research objectives and deepen the research analysis:

- Accuracy: the model does not necessarily need to be validated in the various dry climates that are studied in this research. Indeed, as this research studies drylands (which are diverse in terms of climate and soil) over a long period of time, the accuracy of model predictions is not as important as it would be for a case study. However, it is an advantage if the model has been validated in one or several dry climates;
- Clarity and parsimony: it is preferable that the model is as much as possible transparent to understand the soil-water system more easily. Thus, transparency makes easier the identification of key parameters and processes impacting irrigation sustainability. For these reasons, a parsimonious mechanistic model is desirable;
- Portability: input and output data should be easily imported or exported to common computer programmes such as Microsoft Excel in order to perform further work with the model outputs (e.g. calculations, graphic representations, etc.);
- Error trapping: it is helpful if the model informs the user when unexpected or out-of-range data are entered. It prevents simulation errors and helps to identify where errors come from.

## 4.4 Candidate models

Various models have been developed for simulating water and solute movement in the soil (Bastiaanssen et al., 2007; Subbaiah, 2013) and salinisation dynamics (Ditthakit, 2011; Goel and Tiwari, 2013). These models tend to vary greatly in their characteristics, ranging from simple to sophisticated, from crop specific to general, and from primary crop-based to soil-based. Soil-based models generally use a sophisticated numerical and mechanistic approach to water and solute movement in the soil. However, crop water uptake is calculated by a simple sink term, which is generally an empirical relationship between soil water pressure and root water absorption. Moreover, plant growth dynamics is generally not considered or is dealt in a quite simplistic way. Conversely, crop-based models are more sophisticated regarding the crop response to soil salinity but they are much simpler regarding the modelling of water and solute movement in the soil profile (Castrignanò, Katerji and Mastrorilli, 2002). More precisely, this type of model does not usually model both soil salinity and soil sodicity.

The criteria previously described were used to prioritise the model selection. Soil-water models fulfilling all or most of the essential criteria are compared in Table 4-1 to choose the most adapted one for this study.

**Table 4-1 Comparison of irrigation models to select the most adapted to the current study**

| | SALFPREDICT | SALTIRSOIL_M | SWAP | HYDRUS | WATSUIT |
|---|---|---|---|---|---|
| **MODEL INFORMATION** | | | | | |
| Reference | 1 | 2 | 3 | 4 | 5 |
| Model type | SS | TS | TS | TS | SS |
| **ESSENTIAL CRITERIA** | | | | | |
| Purpose of the model | | | | | |
| *Predicts the evolution of the $EC_e$ and $SAR_e$* | 2 | 2 | 2 | 2 | 2 |
| *Predicts the evolution of the $pH_e$ and $Alk_e$* | 2 | 2 | 2 | 2 | 2 |
| *Predicts the evolution of the $EC_d$ and $SAR_d$* | 2 | 2 | 2 | 2 | 2 |
| *Calculates the salt balance* | 2 | 2 | 2 | 2 | 2 |
| Applicability | | | | | |
| *Applicable to different types of soils from light shallow soils to heavy deep soils* | 2 | 2 | 2 | 2 | 1 |
| *Applicable to different types of climates from hyper-arid to dry sub-humid* | 0 | 2 | 1 | 2 | 1 |
| *Applicable to different types of crops* | 2 | 2 | 2 | 2 | 2 |
| *Applicable to different crop rotations* | 2 | 1 | 1 | 2 | 1 |
| *Applicable to different types of irrigation systems* | 1 | 2 | 1 | 2 | 1 |
| *Applicable to different water qualities (EC and SAR)* | 2 | 2 | 2 | 2 | 2 |
| *Suitable for simulating over-irrigation, PW blending and PW desalination to mitigate soil salinisation and sodification* | 2 | 2 | 2 | 2 | 2 |
| Balance between components | 2 | 2 | 2 | 2 | 2 |
| Spatial scale | 2 | 2 | 2 | 2 | 2 |
| Time scale | 1 (MA) | 2 (ES) | 2 (MA) | 2 (MA) | 2 (ES) |
| Data requirements | 0 | 2 | 0 | 0 | 2 |
| Acceptance | | | | | |
| *Amount of academic papers referred in Scopus and Science Direct in which the model is used or studied* | 0 (0) | 1 (5) | 2 (522) | 2 (54) | 1 (12) |
| *The model has been applied in dry environments* | 2 | 2 | 2 | 2 | 2 |
| Usability | 2 | 2 | 2 | 2 | 2 |
| Target user | 2 | 2 | 1 | 2 | 2 |
| Run-time and system requirements | 2 | 2 | 2 | 2 | 2 |
| Model documentation | 2 | 2 | 2 | 2 | 2 |
| **DESIRABLE CRITERIA** | | | | | |
| Accuracy | 2 | 2 | 2 | 2 | 2 |
| Clarity and parsimony | 2 | 2 | 0 | 1 | 2 |
| Portability | 2 | 2 | 2 | 2 | 2 |
| Error trapping | 0 | 2 | 2 | 2 | 2 |
| **TOTAL SCORE** | **39** | **45** | **38** | **44** | **42** |

[1](Shaw and Kitchen, 2016), [2](Visconti et al., 2014), [3](Van Dam, 2000), [4](Suarez and Simunek, 1997), [5](Rhoades and Merrill, 1976)

ES: equilibrium state, MA: multi-annual, SS: steady-state, ST: steady-state

| | |
|---|---|
| 2 | Able/Convenient |
| 1 | Not fully able/Mean |
| 0 | Unable/Not convenient |

Most soil-water models are mechanistic and deterministic making them relatively transparent (i.e. white box models). However, they mainly differ in terms of time dimension and system state (Letey and Feng, 2007). Therefore, candidate models are discriminated into two groups: steady-state models and transient-state models.

Steady-state models such as SALFPREDICT (Shaw and Kitchen, 2016) and WATSUIT (Rhoades and Merrill, 1976) are characterised by the assumption that water and salts have a steady-state movement through the soil over time. SALFPREDICT demands input data that are difficult to obtain without field measurements (e.g. soil ESP, soil air-dry moisture content, etc.). WATSUIT is less demanding but has less flexibility in terms of simulating different types of soils, climates, crops and irrigation systems due to the limited input parameters considered (Table 4-1). SALFPREDICT is rigid in terms of climatic parameters (i.e. only the precipitation amount can be parameterised by the user) because it was originally conceived for being used in Queensland (Australia) a region with dry sub-humid to semi-arid climate. Lastly, SALFPREDICT has not been used or cited in academic publications (Table 4-1) although it has been used in surveys and consultancies in Australia (Biggs et al., 2013).

In salinity modelling, the steady-state assumption has been controversial, and this has led to the development of transient-state models (Corwin, Rhoades and Šimůnek, 2007; Letey et al., 2011). Transient-state models such as SWAP (Van Dam, 2000), HYDRUS-UNSATCHEM (Suarez and Simunek, 1997) and SALTIRSOIL_M (Visconti et al., 2014) can simulate time-dependent parameters, such as irrigation scheduling, rainfall pattern, crop growth and water quality on soil salinity throughout time. Therefore, this category of models usually better represents the ‘reality’ of the system and provides more accurate predictions of soil salinity than steady-state models (Letey and Feng, 2007). However, most transient-state models use the Richards’ equation and convection-dispersion equations to simulate water and solute flow (Oster et al., 2012) and therefore, they require data which are highly variable and not routinely determined during land surveys (e.g. parameters of the soil water characteristic curve, dispersivity and molecular diffusion coefficients, pore size distributions, etc.). SWAP and HYDRUS-UNSATCHEM are part of these data-demanding models (Table 4-1) that would require field data collection, and this constraint limits their applicability to this study. SALTIRSOIL_M, however, is situated between data-demanding models (i.e HYDRUS-UNSATCHEM, SWAP and SALFPREDICT) and less data-demanding model (i.e. WATSUIT). SALTIRSOIL prolongs the WATSUIT concepts in terms of detailed conceptualisation of irrigation and crop management practices, which enables the user to simulate different irrigation techniques, and soil characterisation through the inclusion of soil hydrology parameters. The flexibility of SALTIRSOIL_M without having to provide inputs that are difficult to obtain are the main advantages of this model compared to SALFPREDICT, WATSUIT, HYDRUS-UNSATCHEM and SWAP (Table 4-1). In addition, SALTIRSOIL_M simulates the average composition of the soil solution at the equilibrium state of the soil-water system. This characteristic matches the objective of modelling the long-term in order to study the system sustainability. In brief, the flexibility, the amount and the type of data required as well as the time dimension of the SALTIRSOIL_M make this model more suitable for the current research.

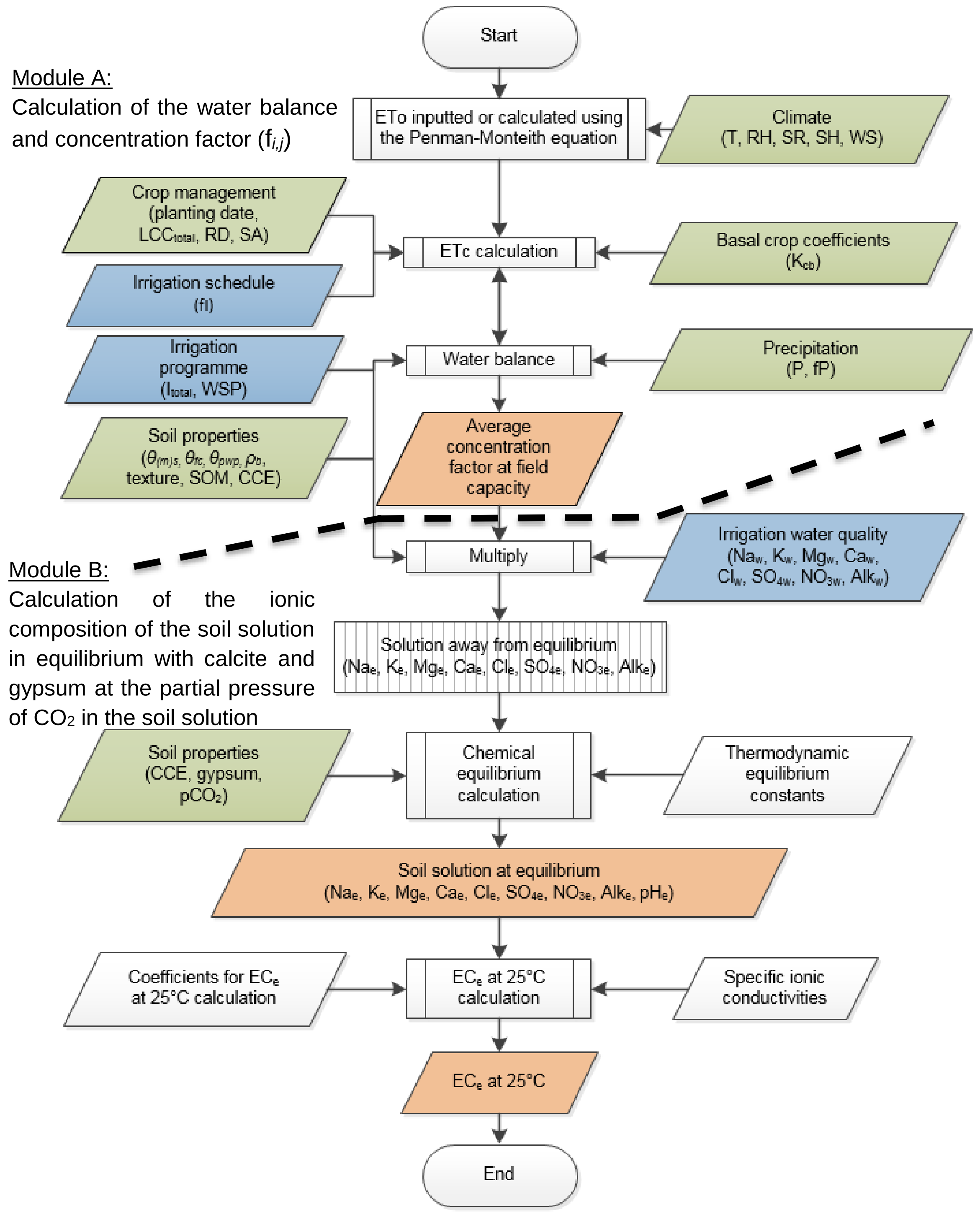
## 4.5 Description of the selected model: SALTIRSOIL_M

### 4.5.1 Model description

SALTIRSOIL_M —SALTs in IRrigation SOILs Monthly assessment— (Visconti et al., 2014) is a one-dimensional, deterministic, transient-state model with a monthly time step that has been developed to improve SALTIRSOIL (Visconti et al., 2011), the original version of the model which is a steady-state model with an annual time step. SALTIRSOIL_M aims at simulating soil salinity in well-drained salt-threatened irrigated lands. Specifically, this model has been developed for calculating the monthly major ion composition (i.e. $Na^+$, $K^+$, $Mg^{2+}$, $Ca^{2+}$, $Cl^-$, $SO_4^{2-}$ and $NO_3^-$), alkalinity, pH, and EC at 25°C of the soil solution and of the drainage water. These parameters can be calculated at three distinct soil-water ratios: field moisture, field capacity and saturation, and in two different parts of the crop rooting depth: average to a given depth (i.e. from the soil surface to the given depth), and at that precise depth. Besides, the model gives the evolution (in percent) of the content in calcium carbonate equivalent and in gypsum of the soil solution. SALTIRSOIL_M is also a water balance model; therefore it includes the calculation of the monthly average field moisture and field capacity from the surface down to the simulation depth or at that particular depth. The monthly volume of drainage water is calculated at the maximum rooting depth and at the simulation depth. Finally, it also calculates the average gravimetric water content at saturation from the surface down to the simulation depth ($\theta_{(m)s}$), and the average gravimetric water content at field capacity from the surface to the simulation depth ($\theta_{(m)fc}$).

### 4.5.2 Model conception

The used equations and algorithms in both SALTIRSOIL and SALTIRSOIL_M models have been exhaustively described and explained in several journal articles (Visconti et al., 2011, 2014) and in the user's manual of SALTIRSOIL_M (Visconti, 2013). In a nutshell, the model is based on a tipping bucket algorithm for simulating the soil water downward movement where the soil is conceptualised as a succession of layers. The model is divided into two complementary units; module A calculates the water balance and the concentration factor of the soil solution. Module B calculates the soil solution composition in equilibrium with calcite and gypsum at the partial pressure of $CO_2$ ($pCO_2$) in the soil solution. SALSOLCHEM (SALine SOLution CHEMistry) is an intermediate module that calculates the chemical equilibrium of the soil solution (Figure 4-1).




Input variables depending on natural factors; Input variables depending on irrigation management; Output variables; Calculations; Intermediate results

Alk: alkalinity, CCE: calcium carbonate equivalent, $EC_e$: electrical conductivity of the soil saturation extract, ETc actual evapotranspiration, ETo: reference evapotranspiration, $I_{total}$: total irrigation amount, $LCC_{total}$: total length of the crop cultivation, fI: number of irrigation days, fP: number of rainy days $K_{cb}$: basal crop coefficient, RD: root depth, RH: relative humidity, SA: shaded area, SH: sunshine hours, SR: solar radiation, P: precipitation amount, $pCO_2$: partial pressure of $CO_2$ in the soil solution, SOM: soil organic matter, T: temperature, WS: wind speed, WSP: wetted soil pattern, $\rho_b$: soil bulk density, $\theta_{fc}$: soil volumetric water content at field capacity, $\theta_{(m)s}$: soil gravimetric water content at saturation $\theta_{pwp}$: soil volumetric water content at permanent wilting point.

**Figure 4-1 Flowchart of the SALTIRSOIL_M model adapted from Visconti et al (2011)**

### 4.5.3 Model assumptions and limitations

The SALTIRSOIL_M model assumes:

- Free or facilitated drainage through the soil profile; SALTIRSOIL_M is suitable to model salt accumulation in soils where the water table level is controlled.

Soils in drylands have generally coarse texture with a high percentage of sand and limited organic matter. Thus, there are usually draining soils (Ffolliott et al., 2002). Furthermore, it is assumed that PW is reused in farms with a high level of technicity regarding salinity management, therefore drainage is supposed to be facilitated when necessary.

- For each month simulated, the model assumes steady-state movement of water and solutes through the soil; for given climatic, water, irrigation, crop and management conditions, there is an equilibrium point where the amount of salt input equals the amount of salt output by leaching and change of soil salinity every year is similar.

Although SALTIRSOIL_M is a transient-state model at the annual scale (the model parameters can vary on a monthly time step) it remains a steady-state model at the monthly scale as input parameters do not vary weekly or daily. In brief, SALTIRSOIL_M simulates the average year of irrigation in the long-term when the equilibrium state is reached. Nevertheless, steady-state-like conditions is a common simplification over long periods in irrigation modelling, especially when the sustainability of irrigation is appraised (Hillel, 2000).

- The soil is conceptualised as a finite number of continuous, homogeneous reservoirs (soil layers) located one over the other; a concentration factor is calculated for each soil layer from the surface to the simulation depth limit, then an average concentration factor is calculated for the entire simulated soil depth.

The soil conceptualisation as a superposition of reservoirs is common in soil-water models, and this allows the simulation of different type of soils that occur in drylands.

- The $EC_e$ is calculated for a soil sample temperature of 25°C.

The standard temperature for measuring the $EC_e$ is 25°C (ISO 11265, 1994). A soil sample can be easily maintained at 25°C in laboratory conditions but in field conditions soil temperature constantly changes. Soil temperature changes affect soil physical and chemical processes. For instance, the $EC_e$ and soil temperature are positively correlated (Visconti and de Paz, 2016), heavy metals absorption by plants (Yu et al., 2013) and organic compounds degradation (Hayat et al., 2011) also vary depending on soil temperature. As the SALTIRSOIL_M model does not account for soil temperature variations, the impact of these on the $EC_e$ are not considered.

### 4.5.4 Use of the model

The SALTIRSOIL_M model was successfully used and validated against measurements obtained in an irrigated field in semi-arid Spain. The model predicted the soil water content and the $EC_e$ of the soil saturated extract at 15 and 45 cm of depth. The model results were used to provide irrigation recommendations regarding the long-term soil salinity and the water-efficiency of irrigation (Visconti et al., 2014). SALTIRSOIL_M has been integrated into a GIS platform to perform soil salinity predictions in the context of climate change (De Paz et al., 2010). The model has been also integrated within the online decision support system DSS-SALTIRSOIL for water management recommendation in salt-threatened lands in the Valencian Region, Spain (Visconti and De Paz, 2016).

In this research, SALTIRSOIL_M is used to simulate the impacts of different irrigation scenarios using PW to irrigate crops in drylands on:

- the soil and drainage water physical parameters: $EC_e$, $pH_e$, $Alk_e$ and $EC_d$;
- the soil and drainage water chemical composition: ionic composition (i.e. soil and drainage water content in $Na^+$, $K^+$, $Mg^{2+}$, $Ca^{2+}$, $Cl^-$, $SO_4^{2-}$ and $NO_3^-$) to calculate the $SAR_e$ and $SAR_d$;
- the water balance.

## 4.6 Sensitivity analysis

In order to test how the variation in the principal model outputs can be attributed to variations of its input factors, the Morris' method (Morris, 1991) also known as the one-factor-at-a-time sensitivity analysis (OFATSA) was applied to the SALTIRSOIL_M model. The OFATSA shows the sensitivity of a calculated output to the changes in a variable if all other variables are kept constant at some value (Pianosi et al., 2016).

The reasons for conducting an OFATSA are to:

- verify the consistency of the model behaviour;
- identify which parameters contribute most to output variability;
- identify which parameters contribute less to output variability;
- assess the consequences of output sensitivity to input variability on the research objectives.

### 4.6.1 Methodology

Four model outputs, which are the soil agro-environmental sustainability indicators were selected: $EC_e$, $SAR_e$, $pH_e$ and $Alk_e$. The sensitivity of these four output parameters to all 40 model input parameters was studied at soil saturation and between 0–50 cm of soil depth (Table 4-2). The time step is a month and the time period spreads from the first month simulated until the equilibrium state (i.e. the stabilised long-term state of the system) is reached.

Four climates were created using climatic averages of the American cities of Oklahoma City (dry sub-humid), Tucson (semi-arid), Las Vegas (arid) and Yuma (hyper-arid). These climates cover the degrees of climate aridity defining drylands. A mean climate was created from these four climates with the average of the values of each climatic parameter of the four climates. This mean climate is the climate of the mean scenario used for the OFATSA (Table 4-2).

The same way, three types of soils (i.e. sandy, loamy and clayey), four types of irrigation water qualities (i.e. fresh water, slightly saline water, saline water and very saline water) and two crops were created. Crop A has a low crop basal coefficient ($K_{cb}$) whereas crop B has a high $K_{cb}$ were created. Crop $K_{cb}$ is defined as the ratio of the crop evapotranspiration over the reference evapotranspiration (ETc/ETo). It has a role in determining the irrigation requirement of a crop. Therefore, it also determines soil salinity as the irrigation amount is proportional to the salt load brought by the irrigation water to the soil.

A mean soil, a mean irrigation water and a mean crop were created on the same principle as for the mean climate and the addition of these, compose the mean scenario in Table 4-2.

Two additional scenarios, one which has its parameters' values increased by 50% compared to the mean scenario and another scenario which has its parameters decreased by 50% compared to the mean scenario were considered (Table 4-2). The sensitivity of the model parameters was evaluated by comparing the percentage change in simulated annual $EC_e$, $SAR_e$, $pH_e$ and $Alk_e$ in the 50 cm soil profile at the equilibrium state. The number of soil layers was set to one and the evaporative soil layer depth was set at 15 cm (default value).

The weight of each input parameter in the determination of the output parameters was determined according to Equation (4-1):

$$\Delta = \left(\frac{Px(-50\%) + Px(+50\%)}{\sum Pn\ (-50\%) + Pn\ (+50\%)}\right) \times 100 \quad (4\text{-}1)$$

where

Δ is the percentage change in the output parameter that can be attributed to the input parameter P$x$. The higher Δ is, the more sensitive the output is to the tested input parameter.

$Px(-50\%)$ is the value of the output parameter in the -50% scenario.

$Px(+50\%)$ is the value of the output parameter in the +50% scenario.

$Pn(-50\%)$ is the value of each output parameter in the -50% scenario.

$Pn(+50\%)$ is the value of each output parameter in the +50% scenario.

**Table 4-2 Values of the parameters of the three simulated scenarios**

| Class | Parameter (abbreviation) | Units | Reference | Mean scenario | -50% scenario | +50% scenario |
|---|---|---|---|---|---|---|
| Climate | Annual precipitation amount (P) | mm | 1 | 489 | 244 | 733 |
| | Annual reference evapotranspiration amount (ETo) | mm | 1 | 1875 | 938 | 2813 |
| | Frequency of precipitation (fP) | day/year | 1 | 38 | 19 | 56 |
| | Annual average temperature (T) | °C | 1 | 21.0 | 10.5 | 31.5 |
| | Annual average relative humidity (RH) | % | 1 | 42 | 21 | 63 |
| | Annual solar radiation (SR) | $MJ/m^2$ | 1 | 7441 | 3720 | 11 161 |
| | Annual sunshine hours (SH) | h/year | 1 | 3883 | 1941 | 5824 |
| | Annual average wind speed (WS) | m/s | 1 | 2 | 1 | 3 |
| Soil | Sand content (Sand) | g/100g | 2 | 33.3 | 16.7 | 50 |
| | Silt content (Silt) | g/100g | 2 | 33.3 | 16.7 | 50 |
| | Clay content (Clay) | g/100g | 2 | 33.3 | 16.7 | 50 |
| | Stone content (Stone) | g/100g | 3 | 16.5 | 8.3 | 24.8 |
| | Calcium carbonate equivalent (CCE) | g/100g | 3 | 50 | 25 | 75 |
| | Soil organic matter (SOM) | g/100g | 3 | 2 | 1 | 3 |
| | Gypsum content (Gypsum) | g/100g | 3 | 0.4 | 0.2 | 0.6 |
| | Initial soil pH ($pH_i$) | - | 3 | 7 | 4 | 11 |
| | Log of $CO_2$ partial pressure ($pCO_2$) | atm | 3 | -3 | -1.5 | -4.5 |
| | Volumetric water content at field capacity ($\theta_{fc}$) | % | 3 | 37 | 18 | 55 |

| Class | Parameter (abbreviation) | Units | Reference | Mean scenario | -50% scenario | +50% scenario |
|---|---|---|---|---|---|---|
| | Gravimetric water content at saturation ($\theta_{(m)s}$) | % | 3 | 0.37 | 0.18 | 0.55 |
| | Volumetric water content at permanent wilting point ($\theta_{pwp}$) | % | 3 | 0.16 | 0.08 | 0.24 |
| | Bulk density ($\rho_b$) | $g/cm^3$ | 3 | 2 | 1 | 3 |
| | Bottom limit (BL) | cm | 3 | 100 | 50 | 150 |
| Irrigation | Total irrigation ($I_{total}$) (mm) | mm | 3 | 1250 | 625 | 1875 |
| | Number of irrigation days (fI) | days | 3 | 151 | 76 | 227 |
| | Fraction of the soil area wetted by irrigation (WSP) | % | 3 | 67 | 34 | 100 |
| Water | Electrical conductivity ($EC_w$) | dS/m | 3 | 2.4 | 1.2 | 3.6 |
| | Sodium content ($Na_w$) | mmol/L | 3 | 165.9 | 82.9 | 248.8 |
| | Potassium content ($K_w$) | mmol/L | 3 | 16.6 | 8.3 | 24.9 |
| | Calcium content ($Ca_w$) | mmol/L | 3 | 8.9 | 4.4 | 13.3 |
| | Magnesium content ($Mg_w$) | mmol/L | 3 | 4.4 | 2.2 | 6.7 |
| | Sodium adsorption ratio ($SAR_w$) | - | 3 | 18 | 9 | 27 |
| | Chlorine content ($Cl_w$) | mmol/L | 3 | 8.9 | 4.4 | 13.3 |
| | Nitrate content ($NO_{3w}$) | mmol/L | 3 | 8.9 | 4.4 | 13.3 |
| | Sulphate content ($SO_{4w}$) | mmol/L | 3 | 8.9 | 4.4 | 13.3 |
| | Alkalinity ($Alk_w$) | $mmol_c/L$ | 3 | 5.0 | 2.5 | 7.5 |
| | Acidity ($pH_w$) | - | 3 | 7.0 | 3.5 | 10.5 |

| Class | Parameter (abbreviation) | Units | Reference | Mean scenario | -50% scenario | +50% scenario |
|---|---|---|---|---|---|---|
| Crop | Root depth (RD) | cm | [3] | 100 | 50 | 150 |
| | Total length of the crop cultivation ($LCC_{total}$) | days | [3] | 90 | 45 | 135 |
| | Average basal crop coefficient ($K_{cb}$) | - | [3] | 0.72 | 0.36 | 1.08 |
| | Maximum shaded area (SA) | % | [3] | 75 | 38 | 100 |

[1] Average of the American cities of Oklahoma City (dry sub-humid climate), Tucson (semi-arid climate), Las Vegas (arid climate) and Yuma (hyper-arid climate) (Diebel, Norda and Kretchmer, 2017; Weather Atlas, 2017)

[2] Adapted from (USDA, 2013)

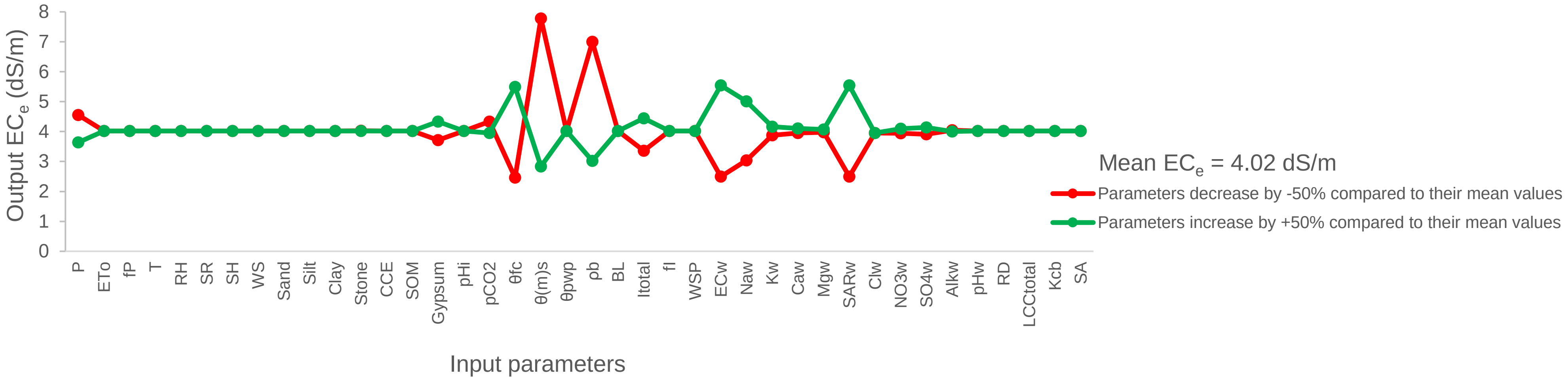
[3] Adapted from (Visconti et al., 2010)

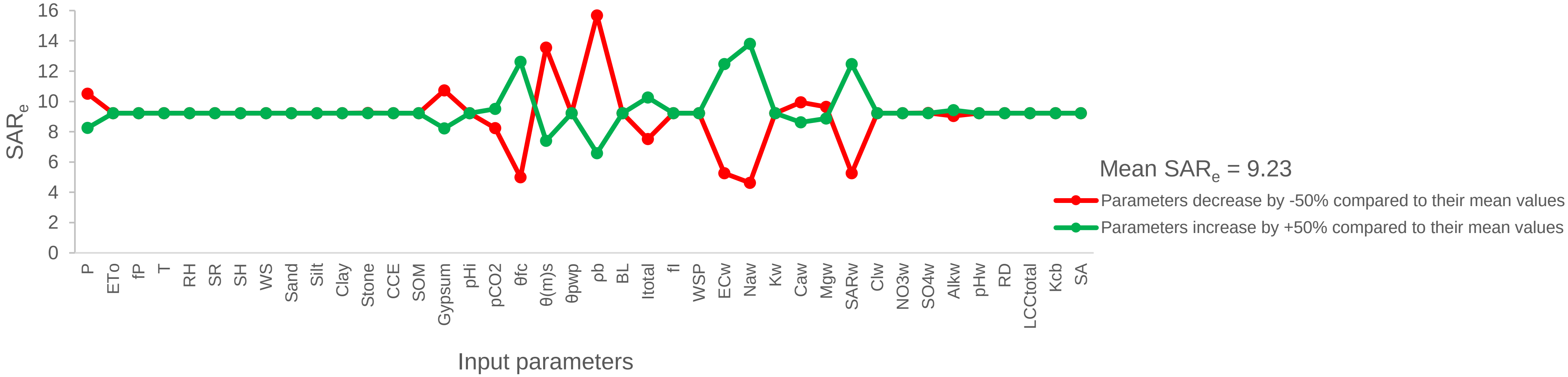
### 4.6.2 Results and discussion

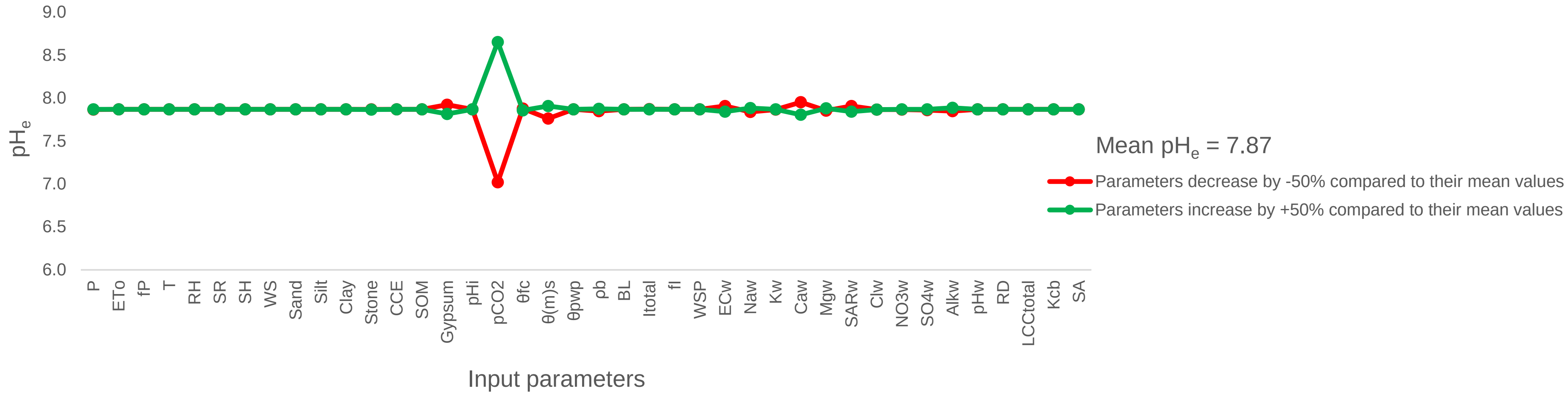
The OFATSA reveals that the $EC_e$ is mostly influenced by soil parameters, mainly $\theta_{(m)s}$, $\rho_b$, $\theta_{fc}$, Gypsum and $pCO_2$ (Figure 4-2) which together represent about 54% of the variability of the output $EC_e$ (Table 4-3). Then the irrigation water quality parameters are also determinant for the calculation of the $EC_e$ (Figure 4-2), indeed, the $SAR_w$, the $EC_w$, $Na_w$ and $K_w$ represent about 35% of the variability of this output (Table 4-3). Finally, the irrigation amount ($I_{total}$) and precipitation (P) represent about 5% and 4% respectively of the $EC_e$ variability (Table 4-3).

The output $SAR_e$ is influenced by the same input parameters as for the $EC_e$ (Figure 4-2) but in a different proportion. In fact, soil parameters: $\theta_{(m)s}$, $\rho_b$, $\theta_{fc}$, Gypsum and $pCO_2$ represent around 46% of the $SAR_e$ variability (Table 4-3) whereas the irrigation water quality parameters: $SAR_w$, $EC_w$, $Na_w$, $Ca_w$ and $Mg_w$ represent approximately 44% of the $SAR_e$ variability. As for the $EC_w$, the irrigation amount ($I_{total}$) and precipitation (P) represent about 5% and 4% respectively of the $SAR_e$ variability (Table 4-3).

The soil alkalinity indicators ($pH_e$ and $Alk_e$) have a different sensitivity profile than the soil salinity and sodicity indicators ($EC_e$ and $SAR_e$). In fact, the OFATSA shows that the outputs $pH_e$ and $Alk_e$ are mostly influenced by the soil parameters $\theta_{(m)s}$, $\rho_b$, gypsum content and $pCO_2$ (Figure 4-2) which together represent around 82% and 81% of the variability of the output $pH_e$ and $Alk_e$ respectively, with a dominant influence of $pCO_2$ in both cases (Table 4-3). The irrigation water quality is less determinant for soil alkalinity indicators. Indeed, $SAR_w$, $EC_w$, $Na_w$, $Ca_w$, $Mg_w$ and $Alk_w$ represent about 16% and 18% of the $pH_e$ and $Alk_e$ variability respectively (Table 4-3).


Output ECe (dS/m)
Input parameters
Mean ECe = 4.02 dS/m
Parameters decrease by -50% compared to their mean values
Parameters increase by +50% compared to their mean values
P ETo fP T RH SR SH WS Sand Silt Clay Stone CCE SOM Gypsum pHi pCO2 θfc θ(m)s θpwp ρb BL Itotal fl WSP ECw Naw Kw Caw Mgw SARw Clw NO3w SO4w Alkw pHw RD LCCtotal Kcb SA


SARe
Input parameters
Mean SARe = 9.23
Parameters decrease by -50% compared to their mean values
Parameters increase by +50% compared to their mean values
P ETo fP T RH SR SH WS Sand Silt Clay Stone CCE SOM Gypsum pHi pCO2 θfc θ(m)s θpwp ρb BL Itotal fl WSP ECw Naw Kw Caw Mgw SARw Clw NO3w SO4w Alkw pHw RD LCCtotal Kcb SA



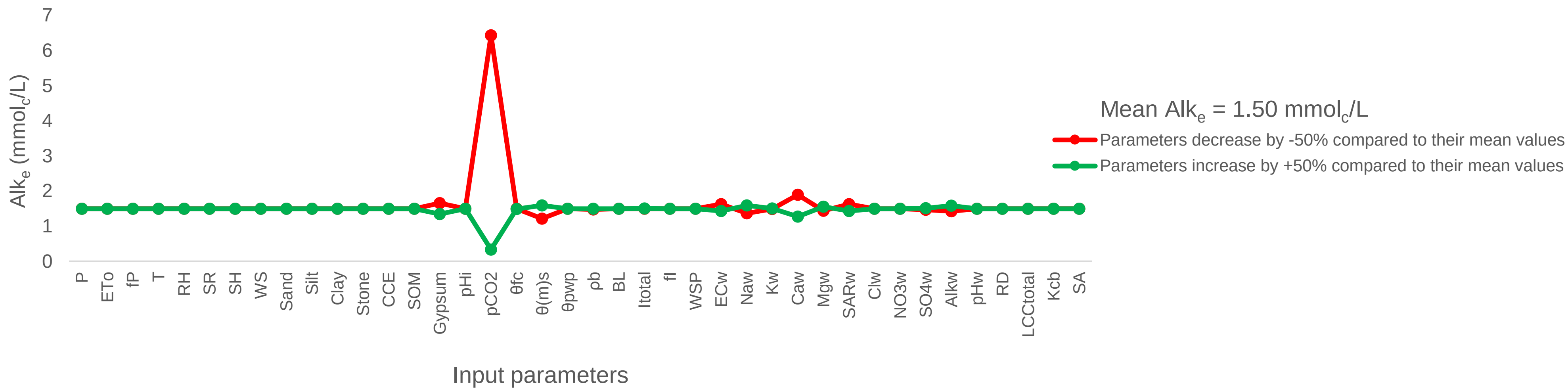




**Figure 4-2 Sensitivity analysis of the model output parameters: (a) $EC_e$, (b) $SAR_e$, (c) $pH_e$ and (d) $Alk_e$ when model input parameters increase or decrease by 50% compared to mean values.**

**Table 4-3 Weight of each model input parameter in the sensitivity of the model output parameters $EC_e$, $SAR_e$, $pH_e$ and $Alk_e$ (in % of responsibility in the total change of the model output parameter).**

| Input parameters \ Output parameters | $EC_e$ | $SAR_e$ | $pH_e$ | $Alk_e$ |
|---|---|---|---|---|
| $\theta_{(m)s}$ | **20.49583** | **10.63785** | **6.30205** | **4.46695** |
| $\rho_b$ | **16.49963** | **15.74999** | **1.16815** | 0.27466 |
| $SAR_w$ | **12.62697** | **12.46714** | **2.79556** | **2.28490** |
| $EC_w$ | **12.62679** | **12.46674** | **2.79561** | **2.28499** |
| $\theta_{fc}$ | **12.55205** | **13.21679** | 0.88287 | 0.14366 |
| $Na_{iw}$ | **8.18422** | **15.89838** | **1.85942** | **2.63694** |
| $I_{total}$ | **4.50767** | **4.74909** | 0.19795 | 0.09805 |
| P | **3.80710** | **3.92493** | 0.15940 | 0.02859 |
| Gypsum | **2.54649** | **4.35415** | **4.62297** | **3.69766** |
| $pCO_2$ | **1.59795** | **2.20661** | **69.67133** | **72.68608** |
| $K_w$ | **1.19101** | 0.00368 | 0.03320 | 0.10684 |
| $SO_{4w}$ | 0.92150 | 0.01741 | 0.42772 | 0.50527 |
| $Ca_w$ | 0.64452 | **2.29813** | **6.25309** | **7.40109** |
| $NO_{3w}$ | 0.63489 | 0.00224 | 0.02567 | 0.06501 |
| $Cl_w$ | 0.57389 | 0.00205 | 0.02238 | 0.05955 |
| $Mg_w$ | 0.39678 | **1.33458** | **1.11205** | **1.37651** |
| $Alk_w$ | 0.18129 | 0.64784 | **1.66534** | **1.87932** |
| Stone | 0.00984 | 0.02077 | 0.00464 | 0.00350 |
| RD | 0.00152 | 0.00159 | 0.00053 | 0.00038 |
| SA | 0.00003 | 0.00003 | 0.00009 | 0.00006 |

The results of the OFATSA can be compared to the results of the global sensitivity analysis (GSA) carried out by Visconti et al. (2010) who attempted to ascertain what input parameters were more influential on the $EC_e$, $SAR_e$ and $pH_e$. Globally, the results of the OFATSA are in concordance with the results of the GSA, however, the GSA reveals a much higher importance of climatic parameters such as P and ETo which could represent together about 39% and 26% of the $EC_e$ and $SAR_e$ variability respectively (Visconti et al., 2010) compared to 4% for both output parameters in the OFATSA (Table 4-3). The same way, the GSA suggests that crop parameters such as the $K_{cb}$ could explain 16% and 13% of the $EC_e$ and $SAR_e$ variability respectively whereas this input parameter does not appear to be determinant in the OFATSA (Figure 4-2). The difference of methodology between the OFATSA and the GSA could be the cause of the underestimation of the influence of P, ETo and $K_{cb}$ on the determination of the $EC_e$ and $SAR_e$ in the OFATSA. Indeed, the OFATSA approach does not take into account the simultaneous variation of input variables. This means that, unlike the GSA (Saltelli et al., 2002), the OFATSA cannot detect the presence of interactions between input variables (Czitrom, 1999). Also, these differences in terms of results could be due to one or several parameters values that make the influence of P, ETo and $K_{cb}$ undetectable in the simulated scenarios.

The results of both sensitivity analyses (OFATSA and GSA) imply that the soil hydro-chemistry (i.e. $\theta_{(m)s}$, $\rho_b$, $\theta_{fc}$, Gypsum and $pCO_2$) and the PW quality (i.e. $SAR_w$, $EC_w$, $Na_w$, $K_w$, $Ca_w$ and $Mg_w$) are critical for the agro-environmental sustainability of irrigation in terms of soil salinity, sodicity and alkalinity. Although soil gypsum content does not stand out as being particularly important for the determination of the $EC_e$, $SAR_e$, $pH_e$ and $Alk_e$, this could be due to the fact that gypsum solubility is limited by the volume of water present into the soil (Visconti et al., 2011). Therefore, a high soil gypsum content significantly impacts the $EC_e$, $SAR_e$, $pH_e$ and $Alk_e$, only if there is sufficient water to dissolve gypsum into the soil solution.

Soil hydro-chemical and PW quality parameters can be managed by irrigation practices and water treatment, whereas other influential parameters such as climate (i.e. P and ETo) and crop (i.e. $K_{cb}$) cannot be controlled. This means that the geographical location with its climate and soil are determinant for the agro-environmental sustainability of a project of irrigation with PW. The possibility of changing these natural parameters is limited, therefore, the agro-environmental sustainability of an irrigation project can eventually be improved through adapting the irrigation management (e.g. changing the irrigation amount) and through water treatment (e.g. changing PW quality).

## 4.7 Conclusions

Modelling is a relevant technique for appraising the agro-environmental sustainability of irrigation practices in an agricultural system. Models have the advantages of being cheap and quick to run compared to field trials, they are also very flexible as many scenarios can be simulated whereas field experiments are dependent on natural conditions. SALTIRSOIL_M, a one-dimensional, deterministic, transient-state model has been selected mainly because of the possibility to predict the required agro-environmental sustainability indicators (i.e. $EC_e$, $SAR_e$, $pH_e$, $Alk_e$, $EC_d$ and $SAR_d$) as a result of long-term irrigation in dry climates. Different irrigation practices can be simulated with SALTIRSOIL_M such as over-irrigation and water treatments (i.e. PW blending and desalination). Finally, most of the data used in the model can be obtained from the literature and do not necessarily need to be obtained through field sampling. This last point is crucial as the research explores wide ranges of soils and climates, thus, conducting field experiments for such diversity is not possible within the allocated timeframe.

The OFATSA shows that the soil agro-environmental sustainability indicators $EC_e$ and $SAR_e$ are most sensitive to the soil hydrology (i.e. $\theta_{(m)s}$, $\rho_b$ and $\theta_{fc}$), the soil chemistry (i.e. gypsum content and $pCO_2$) and the PW quality (i.e. $SAR_w$, $EC_w$, $Na_w$, $K_w$, $Na_w$, $Ca_w$ and $Mg_w$) whereas $pH_e$ and $Alk_e$ are principally sensitive to soil hydro-chemistry (particularly $pCO_2$). Nevertheless, the GSA conducted by Visconti et al. (2010) suggests that climatic parameters (P and ETo) and a crop parameter ($K_{cb}$) have also a significant influence on the $EC_e$ and on the $SAR_e$.

The consequences of selecting the SALTIRSOIL_M model for this research are that the agro-environmental sustainability of irrigation would mainly depend on naturally determined parameters (i.e. the soil water properties and climate aridity) and on PW quality (i.e. $EC_w$ and $SAR_w$). Thus, it is expected that the likelihood of attaining agro-environmental sustainability would decrease as the soil water retention and the PW salinity and sodicity increase. Moreover, the possibility of improving the agro-environmental sustainability of irrigation with PW in a given environment would be simulated by controlling the irrigation amount (i.e. to simulate over-irrigation) and by reducing the PW EC and SAR (i.e. to simulate PW dilution and desalination).

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# 5 ASSESSING THE AGRO-ENVIRONMENTAL SUSTAINABILITY OF IRRIGATION WITH OIL AND GAS PRODUCED WATER IN DRYLANDS[2]

## 5.1 Introduction

Oil and gas (O&G) extraction generates considerable volumes of ‘produced water’ (PW) which is the main by-product of the O&G industry (Veil, 2011). PW mostly originates from water which is naturally present with the hydrocarbons in the reservoir but can also include water that is artificially added to the reservoir and flows back to the surface during enhanced oil recovery and hydraulic fracturing (Engle, Cozzarelli and Smith, 2014). About half of the global PW volume is injected into disposal wells or discharged on the surface after treatment without being beneficially reused (Echchelh, Hess and Sakrabani, 2018). These disposal practices have limits. Deep-well injection is energy intensive, and thus is expensive and is responsible for high $CO_2$ emissions (Arthur et al., 2011). Furthermore, it is environmentally hazardous, as it can pollute groundwater (Hagström et al., 2016) and induce seismic activity (Walsh and Zoback, 2015). Surface discharge can also contaminate soils (Konkel, 2016) and receiving water bodies (Christie, 2012). As a consequence, stricter environmental regulations are being developed requiring extensive PW treatment before discharging (Fakhru’l-Razi et al., 2009) or prohibiting discharge entirely, e.g. Zero Liquid Discharge (Igunnu and Chen, 2014). The increasingly stringent regulation increases PW management cost for O&G firms (Stanic, 2014). As global PW volume is expected to rise drastically (Dal Ferro and Smith, 2007), there is a need for sustainable alternatives to current PW management practices.

PW reuse for irrigation could potentially provide a considerable amount of water to farmlands situated within O&G basins (Echchelh, Hess and Sakrabani, 2018). This option is of the utmost interest in drylands which host a significant part of the world’s hydrocarbon production and reserves (EIA, 2018), and where water

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scarcity is likely to be exacerbated as a result of climate change (Feng and Fu, 2013) and population growth (Safriel et al., 2006). Therefore, to respond to both water scarcity and the environmental-economic limits of traditional PW disposal practices, the reclamation of PW for irrigation in dry areas must be considered.

Despite the large volume available, PW salinity, sodicity and heavy metals contents often exceed the maximum levels recommended in the FAO irrigation water quality guidelines (Alley et al., 2011), thus preventing its application to the soil without adequate treatment. In Oman, for instance, following irrigation with de-oiled PW; the electrical conductivity ($EC_e$) and the soil sodium adsorption ratio ($SAR_e$) of the soil saturation extract dramatically increased from 1.6 to 7.1 dS/m for the $EC_e$ and from 2.3 to 68.1 for the $SAR_e$ after 102 days of irrigation. As a result, the soil saturated hydraulic conductivity decreased from $1.42 \times 10^{-3}$ to $1.6 \times 10^{-6}$ m/s (Hirayama et al., 2002). Similarly, in semi-arid USA, when untreated PW was used to irrigate camelina, the soil $EC_e$ increased from 1.4 to 1.9 dS/m while the soil $SAR_e$ rose from 0.2 to 2.0 (Sintim et al., 2017). Comparable observations have been reported in other semi-arid regions of the USA (Burkhardt et al., 2015; Johnston, Vance and Ganjegunte, 2008), Northeast Brazil (Sousa et al., 2017), South Africa (Beletse et al., 2008) as well as in dry sub-humid Australia (Biggs et al., 2013). Although these changes to soil properties may not immediately affect crop productivity, the long-term implications of irrigation using PW without soil salinity and sodicity management are uncertain.

Most research addressing the impacts of irrigation with PW is composed of short-term field experiments (1–3 years) whereas, O&G fields longevity varies from 5 to more than 50 years (Encana, 2011; Total, 2015). Moreover, field trials are carried out under specific climates and on particular soils, so their results cannot be easily extrapolated to other types of drylands. Also, the qualities of the PWs used in these trials do not necessarily represent the diversity of PW qualities. Therefore, there is a need for extending the study of the impacts on soil fertility of irrigation with PW of different qualities on soil fertility in the long-term and under different climates and soil types.

To this end, simulation with soil-water models such as SALTIRSOIL_M (Visconti et al., 2014) is an adequate methodology for studying the long-term impacts of irrigation with a range of representative PWs on multiple soils and climates typical of drylands. Modelling is an appropriate tool, firstly because it reduces the time needed for obtaining results compared to field experiments. Also, models can be run with 'what-if' scenarios describing different situations without the need for a large number of field trials. Lastly, models allow the simulation of extreme scenarios without any adverse consequences on the environment (Graves et al., 2002). Although Mallants et al. (2017) and Jakubowski et al. (2013) modelled the impacts of irrigation with PW on soil salinity on the medium-term (1–10 years), they did not consider the long-term agro-environmental sustainability of this practice. In addition, these studies were limited to dry sub-humid Queensland, Australia.

This paper aims to estimate the agro-environmental sustainability of irrigation with PW in dry conditions and to determine how it is affected by environmental parameters (i.e. PW quality, climate and soil type). Here, agro-environmentally sustainable irrigation refers to maintaining soil fertility in the long-term (i.e. indefinitely), which means to preserve soil structural stability and maintain a crop yield of at least 50% of optimum potential. For that, the salinity ($EC_e$), sodicity ($SAR_e$) and pH ($pH_e$) of the soil saturation extract must be preserved from the effects of the irrigation water salinity ($EC_w$), sodicity ($SAR_w$) and pH ($pH_w$). Agro-environmentally sustainable irrigation also includes appropriate management of drainage water (DW) depending on its salinity ($EC_d$) and sodicity ($SAR_d$). The impacts of irrigation with PW on soil fertility, DW quality and crop yield are discussed from an agro-environmental perspective.

## 5.2 Methods

### 5.2.1 Soil-water model

SALTIRSOIL_M is a one-dimensional, deterministic, transient-state model with a monthly time step (Visconti, 2013). Based on a tipping-bucket algorithm, it simulates the water movement through a number of soil layers (*n*) and down to a specific soil depth chosen by the user. As a result, the model calculates a concentration factor of the soil solution regarding the irrigation water ($f_{i,j}$ = $C_{SS_{i,j}}/C_{I_i}$), for each month *i* and soil layer *j* with Equations (5-1) and (5-2), where $C_{SS_{i,j}}$ is the concentration of the *k*th ion in the soil solution of the *j*th layer in the *i*th month, and $C_{I_i}$ is the concentration of the *k*th ion in the irrigation water in the *i*th month. Equation (5-1) expresses the soil solution concentration factor for the first soil layer (*j* = 1), and Equation (5-2) for subsequent layers.

$$f_{i,1} = \frac{V_{i-1,1} f_{i-1,1} \left(\frac{C_{Ii-1}}{C_{Ii}}\right) + I_i}{V_{i,1} + D_{i,1}} \tag{5-1}$$

$$f_{i,1} = f_{i,j} = \frac{V_{i-1,j} f_{i-1,j} \left(\frac{C_{Ii-1}}{C_{Ii}}\right) + D_{i,j-1} f_{i,j-1}}{V_{i,j} + D_{i,j}} \tag{5-2}$$

In Equations (5-1) and (5-2), $f_{i,j-1}$ is the concentration factor in the previous month, $V_{i,\mathrm{j}}$ and $V_{i-1,\mathrm{j}}$ are, respectively, the soil water content of the soil layer $j$ in the month $i$ and in the previous ($i-1$) month, $D_{i,j}$ and $D_{i,j-1}$ are, respectively, the drainage amount from the soil layer *j* and from the overlaying ($j-1$) layer in the month $i$, $C_{Ii-1}/C_{Ii}$ is the quotient of the irrigation water concentration the previous ($i-1$) regarding the present ($i$) month, and finally, $I_i$ is the irrigation water amount in the present month ($i$).

The main ion concentrations in the irrigation water ([*k*] where *k* = $Na^+$, $K^+$, $Mg^{2+}$, $Ca^{2+}$, $Cl^-$, $SO_4^{2-}$ and $NO_3^-$) are multiplied by the monthly averages of the soil solution concentration factors from the first down to the *n*th layer chosen by the user ($\overline{f_i}$), and besides, by the quotient of the soil water content at saturation ($\theta_S$) to the soil water content at field capacity ($\theta_{fc}$) (Equation (5-3)).

$$[k]_{e,i} = \frac{\theta_s}{\theta_{fc}} \overline{f_i} [k]_i \tag{5-3}$$

As a result, the main ion composition of monthly soil saturation extracts away from chemical equilibrium is obtained. These ion concentrations are then entered into a chemical equilibrium model that calculates the soil solution ionic composition at equilibrium by letting calcite ($CaCO_3$) and gypsum ($CaSO_4 \cdot 2H_2O$) precipitate or dissolve, if present, at the specified $CO_2$ partial pressure ($pCO_2$).

Finally, the soil $pH_e$ and the $EC_e$ at 25ºC are calculated, the latter by using, in addition to ion concentrations, their ionic conductivities.

The month-by-month year-round ionic composition, $pH_e$ and $EC_e$ calculated with the model represents the steady state that would be reached in the long-term under constant irrigation water composition, irrigation management, climate features, soil physical properties and crop.

The SALTIRSOIL_M model has been successfully used to predict the equilibrium soil ionic composition and $EC_e$ of irrigated semi-arid lands in Spain (Visconti et al., 2014). In this paper, the model is used to calculate the equilibrium $EC_e$, $SAR_e$, and $pH_e$ of the soil saturation extract and of the DW.

### 5.2.2 Model parameterisation

Locations from the Western USA, preferably near O&G fields, were chosen to represent the different types of dry climates (Table 5-1). Dry climates are classified using the UNEP aridity index (AI) which is defined as the ratio of precipitation and potential evapotranspiration (Cherlet et al., 2018). A climate is hyper-arid if AI < 0.05, arid if 0.05 ≤ AI < 0.20, semi-arid if 0.20 ≤ AI < 0.50 and dry sub-humid if 0.5 ≤ AI < 0.65. Monthly climatic averages were calculated from daily time series for the period 1990–2016. Temperature, relative humidity, precipitation, reference evapotranspiration (ETo), wind speed, and downward solar radiation, were sourced from the University of Idaho's METDATA (Abatzoglou, 2013) and the average number of days with precipitation from Weatherbase (Canty, Frischling and Frischling, 2018). The number of sunshine hours was estimated using the adapted equation of Ångström-Prescott (Viswanadham and Ramanadham, 1969).

The Harmonised World Soil Database (FAO, 2009) was used to select four representative soil types according to FAO's Reference Soil Groups (RSG) classification (IUSS Working Group WRB, 2015). The selected soil types (Gypsisol, Arenosol, Planosol and Vertisol) represent ~22%, ~12%, ~5% and ~5% respectively of soils in drylands (Koohafkan and Stewart, 2008). All soil samples belonging to the same soil type were grouped and their parameters values averaged to create an indicative soil for each soil type. The soil volumetric water contents at saturation and at field capacity were estimated from the soil texture and organic matter content (Saxton and Rawls, 2006). The soil organic matter content (SOM) was estimated from the total organic carbon content using a Van Bemmelen factor of 1.72 (Soil Survey Staff, 1996). The soil $CO_2$ partial pressure ($pCO_2$) was estimated from soil pH (Thomas, 1996) (Table 5-2).

Tropical sugar beet was selected as an exemplar crop for the following reasons. Firstly, it is salt-tolerant (Tanji and Kielen, 2002), sodium and chloride-tolerant (Wakeel, Steffens and Schubert, 2010) and it can be grown in a wide range of soils (SESVanderHave, 2016) and under dry climates (Chatin et al., 2004; Nilsson, 2005). Secondly, sugar beet usually adapts well to drip irrigation, which is the most suitable system in water-scarce drylands (Rhoades, Kandiah and Mashali, 1992). Finally, this crop has multiple uses such as foodstuff (sugar), animal feed (pellets and molasses) and biofuel. The planting date was set on November 1st, a typical planting date in the Northern Hemisphere regions with Mediterranean arid climates (FAO, 2018a). Crop coefficients, growth stages lengths and root depths were obtained from FAO (2018a). The shaded area values were sourced from Webb et al. (1997).

CROPWAT 8.0 (FAO, 2018b) was used to estimate the crop water requirements and to set the irrigation schedule for each climate (Table 5-1). No deliberate leaching nor amendments were included

**Table 5-1 Parameters of climate, crop development and irrigation schedules used in the simulations of irrigation of sugar beet with PW in dry climates.**

| | Parameter | January | February | March | April | May | June | July | August | September | October | November | December | Total |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Hyper-arid: Yuma, Arizona (AI= 0.04) | P (mm) | 10 | 10 | 7 | 3 | 1 | 0 | 6 | 8 | 11 | 6 | 6 | 10 | 78 |
| | ETo (mm) | 80 | 93 | 144 | 184 | 232 | 254 | 269 | 249 | 194 | 138 | 90 | 72 | 2000 |
| | I (mm) | 105 | 133 | 218 | 285 | 349 | 306 | 115 | 0 | 0 | 0 | 43 | 59 | 1612 |
| Arid: Bakersfield, California (AI = 0.09) | P (mm) | 31 | 31 | 27 | 13 | 6 | 1 | 0 | 0 | 2 | 9 | 12 | 26 | 157 |
| | ETo (mm) | 47 | 64 | 112 | 157 | 220 | 254 | 265 | 238 | 180 | 125 | 66 | 45 | 1773 |
| | I (mm) | 33 | 65 | 147 | 244 | 346 | 329 | 123 | 0 | 0 | 0 | 20 | 13 | 1320 |
| Semi-arid: Santa Fe, New Mexico (AI = 0.23) | P (mm) | 16 | 14 | 18 | 18 | 25 | 27 | 62 | 55 | 40 | 35 | 22 | 25 | 357 |
| | ETo (mm) | 50 | 64 | 106 | 144 | 190 | 215 | 180 | 159 | 137 | 108 | 67 | 47 | 1468 |
| | I (mm) | 58 | 85 | 147 | 202 | 252 | 224 | 53 | 0 | 0 | 0 | 14 | 19 | 1054 |
| Dry sub-humid: Dallas, Texas (AI = 0.64) | P (mm) | 70 | 66 | 92 | 94 | 123 | 102 | 50 | 62 | 68 | 113 | 74 | 79 | 991 |
| | ETo (mm) | 64 | 74 | 113 | 146 | 174 | 201 | 226 | 203 | 153 | 117 | 77 | 62 | 1609 |
| | I (mm) | 13 | 36 | 74 | 123 | 142 | 139 | 63 | 0 | 0 | 0 | 0 | 0 | 590 |
| Crop growth | $K_{cb}$ | 0.70 | 1.15 | 1.20 | 1.20 | 0.93 | 0.70 | 0.41 | 0 | 0 | 0 | 0.41 | 0.70 | - |
| | Root depth (cm) | 49 | 56 | 84 | 92 | 100 | 100 | 100 | 0 | 0 | 0 | 15 | 30 | - |

AI: aridity index, P: precipitation; ETo: reference evapotranspiration; I: irrigation; $K_{cb}$: basal crop coefficient.

**Table 5-2 Parameters of the four soils used in the simulations.**

| Soil type (FAO's RSG) | Soil layer (cm) | Hydrophysical | | | USDA texture (%) | | | Chemical | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | $\rho_b$ (g/cm$^3$) | $\theta_{fc}$ (%) | $\theta_{pwp}$ (%) | Sand (%) | Silt (%) | Clay (%) | pH | Gypsum (%) | CCE (%) | SOM (%) | log $pCO_2$ |
| Arenosol | Topsoil 0–30 | 1.70 | 10 | 5 | 89 | 6 | 5 | 6.1 | 0.02 | 0.74 | 0.58 | 0 |
| | Subsoil 30–100 | 1.69 | 10 | 5 | 89 | 5 | 6 | 6.1 | 0.02 | 0.81 | 0.27 | 0 |
| Gypsisol | Topsoil 0–30 | 1.42 | 28 | 14 | 45 | 34 | 21 | 7.9 | 12.57 | 6.42 | 0.57 | -3 |
| | Subsoil 30–100 | 1.38 | 31 | 11 | 41 | 33 | 26 | 7.9 | 16.99 | 5.62 | 0.30 | -3 |
| Planosol | Topsoil 0–30 | 1.43 | 36 | 25 | 51 | 29 | 20 | 5.7 | 0.01 | 0.16 | 1.45 | 0 |
| | Subsoil 30–100 | 1.33 | 36 | 22 | 40 | 25 | 35 | 6.3 | 0.01 | 0.58 | 0.59 | 0 |
| Vertisol | Topsoil 0–30 | 1.22 | 42 | 30 | 22 | 24 | 54 | 7.2 | 0.14 | 2.39 | 1.81 | -2 |
| | Subsoil 30–100 | 1.21 | 42 | 30 | 21 | 23 | 57 | 7.6 | 0.19 | 3.64 | 0.99 | -2 |

FAO's RSG: FAO Reference Soil Groups, $\rho_b$: bulk density; $\theta_{fc}$: soil volumetric water content at field capacity; $\theta_{pwp}$: soil volumetric water content at permanent wilting point; CCE: calcium carbonate equivalent.

### 5.2.3 Produced water quality

Data on PW origin and quality for 33 PWs were sourced from the USGS National Produced Waters Geochemical Database (Blondes et al., 2017). An exploratory data analysis of the ten physicochemical water properties (EC, pH, [$Na^+$], [$K^+$], [$Mg^{2+}$], [$Ca^{2+}$], [$Cl^-$], [$NO_3^-$], [$SO_4^{2-}$] and alkalinity (Alk)) was carried out in the 33 PWs. The distributions of these properties fulfilled the requirements for log-normally distributed variables with the exception of pH, being this last one normally distributed. However, inspection of the histograms revealed some data clustering that could diminish the precision of the sustainability assessment. Since having more regularly distributed data would be optimal, a stochastic PW generator (SPWG) was developed on the basis of the 33 PW according to the methodology outlined in the ensuing paragraph.

First of all, the original water properties were log-transformed with the exception of pH, and their means and standard deviations assessed. Second, a principal components analysis (PCA) was performed on the log-transformed data table of 33 PWs and, as a consequence, its matrix of eigenvectors was obtained. Third, independent random values were obtained from a marginal normal distribution with zero mean and one standard deviation for each of the 10 principal components (PCs) of a set of 1000 synthetic waters. Four, the logarithmic values of the 10 physicochemical water properties in all these synthetic PWs were calculated using the previously obtained matrix of eigenvectors in addition to the corresponding means and standard deviations, which we know from the first step. Five, these logarithmic values for the ten properties in the set of 1000 synthetic PWs were back-transformed to become normal. Six, the charge balance errors (CBE) were calculated in every synthetic PW and the PWs exceeding ± 2% were deleted from the dataset. Finally, just 15 PWs regularly covering a wide range from 0.3 to 130.3 dS/m were kept and used in the simulations (Table 5-3).

**Table 5-3 Quality of the different PWs used for irrigation simulations ranked by increasing $EC_w$ (all ions contents are expressed in mmol/L, alkalinity as $[CaCO_3]$ equivalent in $mmol_c/L$, and $EC_w$ in dS/m).**

| | $[Na^+]$ | $[K^+]$ | $[Ca^{2+}]$ | $[Mg^{2+}]$ | $[Cl^-]$ | $[NO_3^-]$ | $[SO_4^{2-}]$ | $Alk_w$ | $EC_w$ | $SAR_w$ | $pH_w$ |
|---|---|---|---|---|---|---|---|---|---|---|---|
| PW1 | 1.1 | 0.0 | 0.0 | 0.0 | 0.1 | 0.0 | 0.0 | 0.9 | 0.3 | 10 | 7.1 |
| PW2 | 9.1 | 0.1 | 0.4 | 0.1 | 3.3 | 0.0 | 0.1 | 6.8 | 0.9 | 13 | 6.8 |
| PW3 | 7.4 | 0.1 | 4.9 | 0.2 | 8.8 | 0.0 | 0.1 | 8.1 | 1.6 | 3 | 6.0 |
| PW4 | 36.0 | 0.3 | 4.2 | 2.7 | 47.6 | 0.0 | 0.2 | 2.7 | 4.5 | 14 | 5.6 |
| PW5 | 46.4 | 0.7 | 11.5 | 0.7 | 65.7 | 0.1 | 0.2 | 3.9 | 6.3 | 13 | 7.5 |
| PW6 | 58.6 | 0.2 | 26.4 | 0.9 | 102.2 | 0.0 | 0.2 | 10.6 | 9.9 | 11 | 6.2 |
| PW7 | 51.8 | 0.1 | 38.9 | 2.9 | 127.9 | 0.1 | 0.3 | 6.7 | 12.1 | 8 | 5.6 |
| PW8 | 124.4 | 6.7 | 8.7 | 9.3 | 166.2 | 0.0 | 0.7 | 2.0 | 14.4 | 29 | 6.7 |
| PW9 | 190.8 | 1.9 | 2.7 | 2.6 | 195.8 | 0.0 | 0.5 | 1.3 | 16.7 | 83 | 6.5 |
| PW10 | 179.5 | 1.4 | 23.2 | 2.1 | 198.2 | 0.0 | 0.4 | 28.1 | 18.9 | 36 | 6.7 |
| PW11 | 103.7 | 0.5 | 101.9 | 7.4 | 307.3 | 0.0 | 0.9 | 1.9 | 27.9 | 10 | 6.6 |
| PW12 | 466.4 | 0.2 | 2.0 | 5.2 | 488.0 | 0.0 | 0.4 | 3.8 | 38.5 | 174 | 7.6 |
| PW13 | 559.4 | 6.4 | 17.2 | 6.4 | 589.3 | 0.0 | 0.9 | 2.1 | 47.3 | 115 | 6.8 |
| PW14 | 759.1 | 2.1 | 36.7 | 31.6 | 918.4 | 0.0 | 1.7 | 2.6 | 71.3 | 92 | 6.8 |
| PW15 | 866.4 | 1.2 | 220.5 | 126.8 | 1572.2 | 0.0 | 3.9 | 3.9 | 130.3 | 47 | 7.2 |

### 5.2.4 Model scenarios

The 240 simulated scenarios represent the irrigation with each PW (15) on each soil type (4) and under each climate (4). The soil depth chosen for the simulation was 60 cm because this is the depth where sugar beet root density is the highest (Draycott, 2006). All results of soil composition were expressed for a saturated extract at chemical equilibrium.

### 5.2.5 Agro-environmental sustainability assessment

The sustainability assessment is based on a comparison between (1) indicators values estimating the long-term soil fertility and crop yield resulting from irrigation with PW and (2) thresholds values below which, the soil fertility and crop yield are preserved (Figure 5-1).

Soil fertility was appraised using the calculated indicators $SAR_e$ and $EC_e$, which were compared to threshold values. Threshold $SAR_e$ values for soil structural stability were based on the Australian and New Zealand Environment Conservation Council guidelines (ANZECC, 2000) which have been used as a reference to study the risks and feasibility of irrigating with PW under dry climates in Australia and in sub-Saharan Africa (Horner et al., 2011; Mallants et al., 2017). The thresholds for $SAR_e$ were set at 20 for Arenosol (sandy soil with clay content < 15%), 20 for Gypsisol (loamy soil with 15% < clay content < 24%), 13 for Planosol (clay loam soil with 25% < clay content < 34%) and 5 for Vertisol (clayey soil with 55% < clay content < 64%). Due to the critical importance of the $SAR_e$ for soil stability, no scenario could be considered agro-environmentally sustainable if the simulated soil $SAR_e$ exceeded the ANZECC guidelines thresholds.

The $EC_e$ was evaluated through the expected effects on sugar beet yield considering the FAO salt tolerance parameters given by Shaw et al (2011). That is, an $EC_e$ of 7 dS/m for a maximum yield and a productivity decrease of 5.9% per dS/m increase of $EC_e$. Therefore, taking a minimum yield of 50% of its potential, the resulting maximum $EC_e$ is 15.5 dS/m.

The quality of DW can affect the sub-soil and the aquifer. In fact, DW can carry dissolved salts into the aquifer and depending on its depth, it may result in groundwater salinisation (Shannon, Cervinka and Daniel, 1997). DWs qualities are ranked according to their $EC_d$ (Rhoades et al., 1992) to consider their potential impacts on groundwater.

In addition, the $pH_e$ was used as a complementary indicator for assessing the risk of nutrient deficiencies which have an impact on crop yield and quality (McEnroe and Coulter, 1964). The $pH_e$ threshold values are the suitable range of values for sugar beet cultivation (SESVanderHave, 2016).

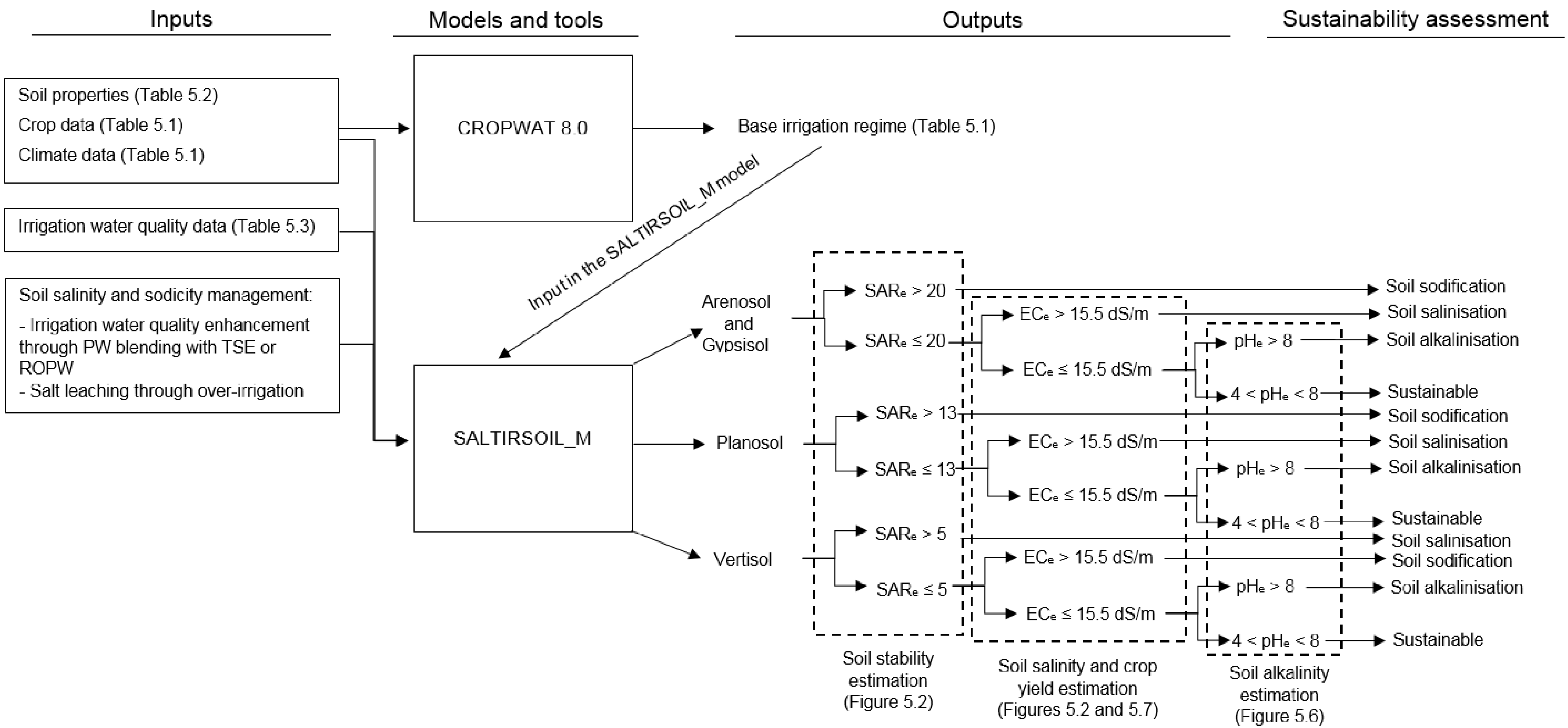



**Figure 5-1. Research methodology flowchart and decision tree for the sustainability assessment.**

## 5.3 Results

The impact of irrigation with PW on the long-term soil salinity and sodicity of the 240 scenarios are presented in Figure 5-2. The DWs resulting from irrigation are classified by their level of salinity in Figure 5-3. The salinity and sodicity balances between the different salt reservoirs (irrigation water, soil and DW) are described by the slope of the curves in Figure 5-4 and Figure 5-5.

Soil salinity and sodicity showed a remarkable linear dependence on PW salinity and sodicity according to the high $R^2$ in Figure 5-4 and Figure 5-5. The curves $EC_e = f(EC_w)$ in Figure 5-4 indicate how prone the different soils are to salinisation because of salt transfer from the irrigation water to the soil. The slope of the curve $EC_e = f(EC_w)$ was the steepest for Vertisol (1.01), Planosol (0.87), Gypsisol (0.65) and finally Arenosol (0.23). On the other hand, the curves $EC_d = f(EC_w)$ illustrate how dependent on PW is DW salinity in each soil type. In this case, the slope of this curve was the highest for Arenosol (2.096), Vertisol (1.850), Planosol (1.836) and finally Gypsisol (1.823). Likewise, the curves $SAR_e = f(SAR_w)$ indicate the soil sensitivity to sodification due to the transfer of sodium from the irrigation water to the soil. The slope of this curve was the steepest for Vertisol (0.94), Planosol (0.88), Arenosol (0.54) and Gypsisol (0.42). Next, the curves $SAR_d = f(SAR_w)$ indicate the ability of the soil to buffer the calcium and magnesium concentrations of the water that percolates through it. In this case, the slope of this curve was the highest for Planosol (1.319), Vertisol (1.316), Arenosol (1.046) and Gypsisol (0.904).

The impact of irrigation with PW on soil salinity and sodicity can be amplified by the increasing aridity (Figure 5-5). Indeed, all soils combined, the slopes of the curves $EC_e = f(EC_w)$ and $SAR_e = f(SAR_w)$ from highest to lowest were as follow: hyper-arid (0.886 and 0.963), arid (0.885 and 0.753), semi-arid (0.675 and 0.663) and dry sub-humid (0.402 and 0.445) respectively. Whereas the slopes of the curves $EC_d = f(EC_w)$ and $SAR_d = f(SAR_w)$ from highest to lowest were as follow: arid (2.56 and 1.35), hyper-arid (2.25 and 1.33), semi-arid (1.70 and 1.01) and dry sub-humid (1.08 and 0.81) respectively.

According to $R^2$, soil $pH_e$ was not dependent on PW quality in Gypsisol and Vertisol, whereas it was somewhat more dependent in the case of Arenosol and Planosol

(Figure 5-6). There was no risk for crop yield due to unsuitable $pH_e$ as tropical sugar beet can be grown in soil with pH ranging from 4 to 8. Instead, crop yield responded negatively to increasing $EC_e$ (Figure 5-7).

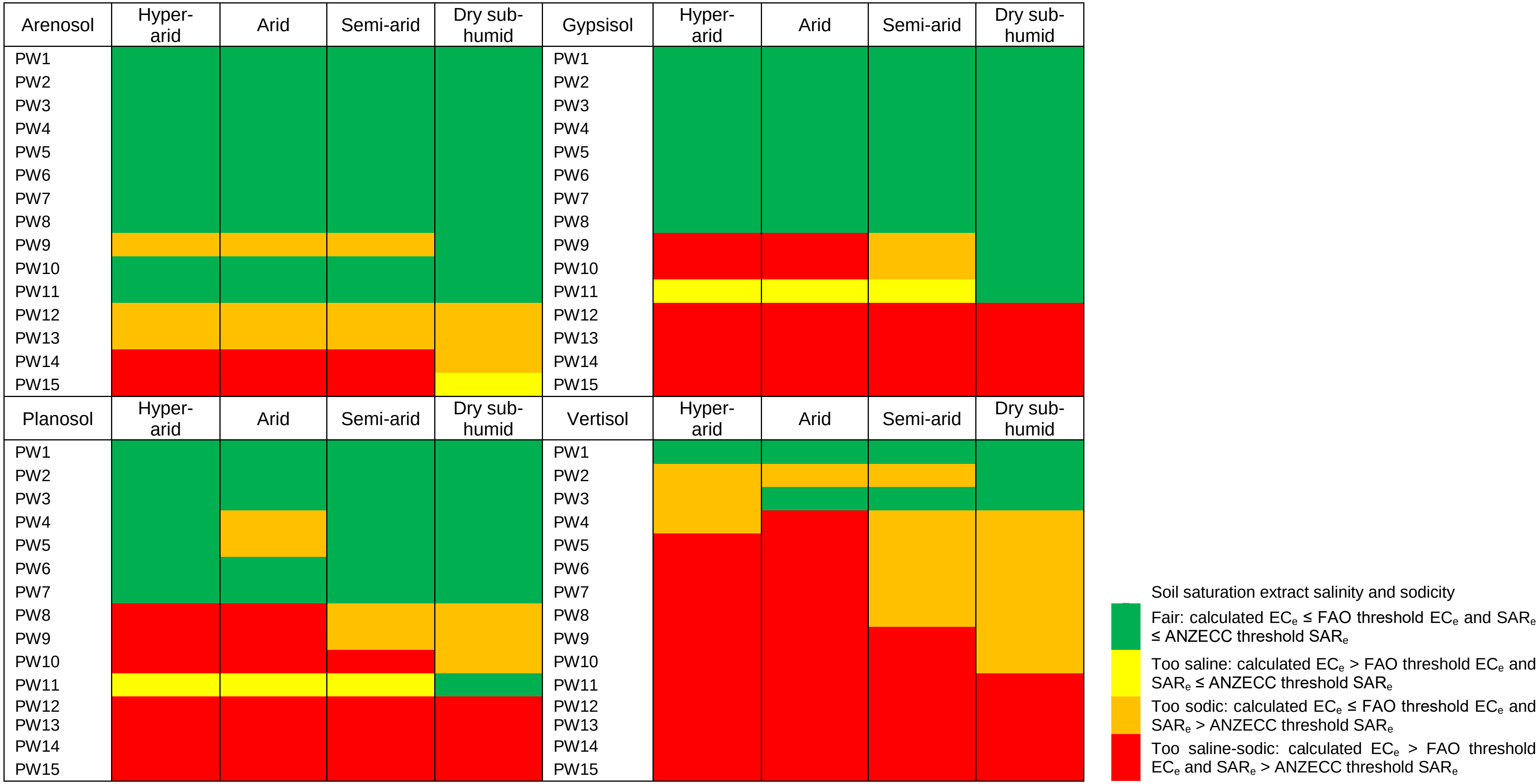



**Figure 5-2 Salinity and sodicity of the soil solution in the long-term as a result of irrigation with 15 PWs under hyper-arid, arid, semi-arid and dry sub-humid climates and on Arenosol, Gypsisol, Planosol and Vertisol. The $SAR_e$ threshold values are 20 for Arenosol and Gypsisol, 13 for Planosol and 5 for Vertisol. The limit $EC_e$ value for sugar beet cultivation is 15.5 dS/m, below this value crop yield is lower than 50% of its optimum.**

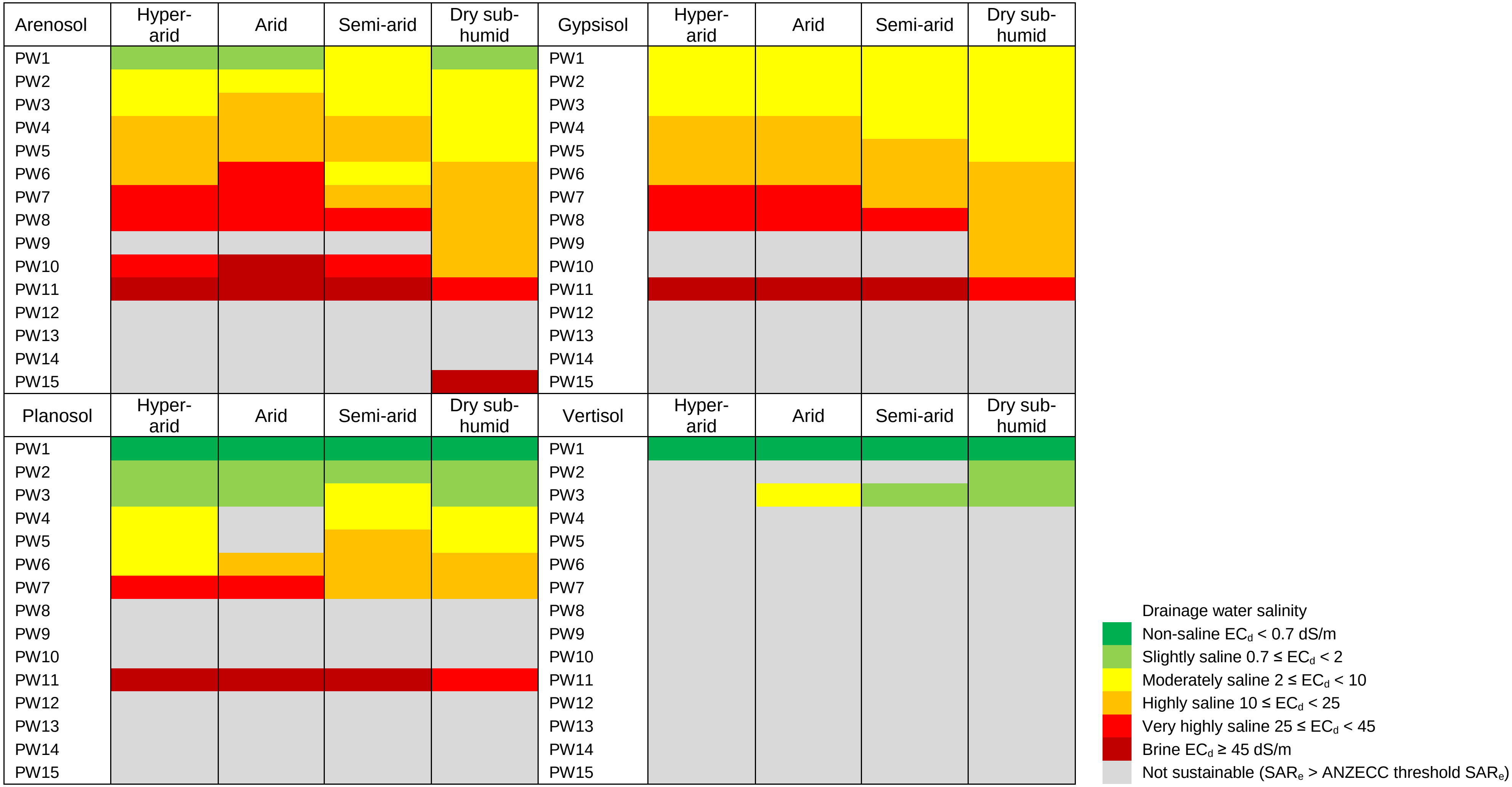
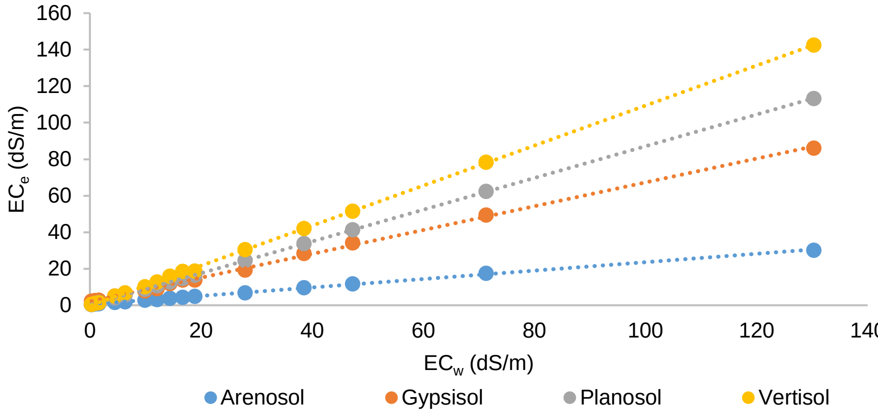
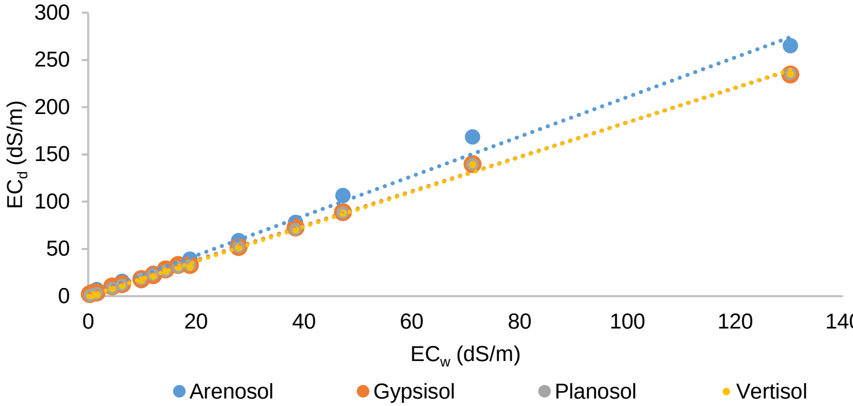


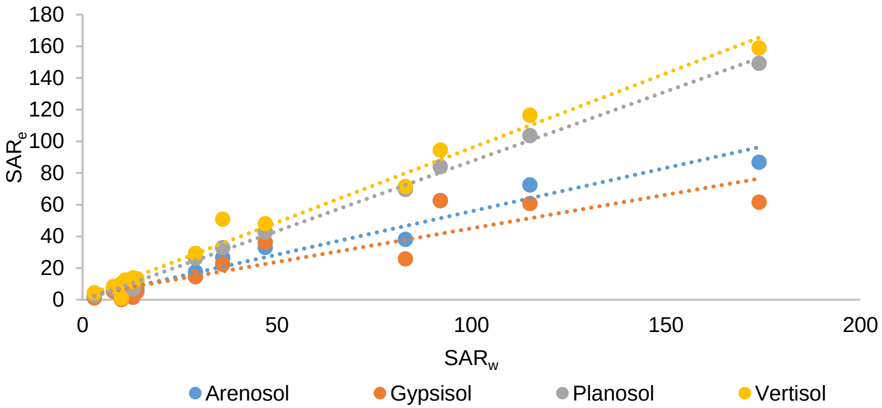

**Figure 5-3 Drainage water salinity leaving the root zone (0-60 cm) of the selected agro-environmentally sustainable (fair soil salinity) and likely agro-environmentally sustainable scenarios (too saline soils) in Figure 5-2.**



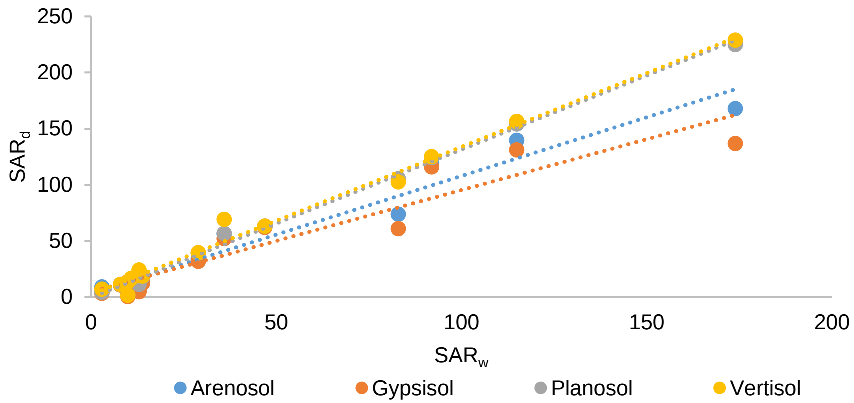

| Soil | Arenosol | Gypsisol | Planosol | Vertisol |
|---|---|---|---|---|
| Curve equation | y = 0.2307x + 0.5134 | y = 0.6522x + 2.0578 | y = 0.8692x + 0.167 | y = 1.0961x - 0.2392 |
| $R^2$ | 0.9993 | 0.9989 | 1 | 0.9998 |




| Soil | Arenosol | Gypsisol | Planosol | Vertisol |
|---|---|---|---|---|
| Curve equation | y = 2.0962x + 1.117 | y = 1.8225x + 1.6751 | y = 1.8361x + 0.3944 | y = 1.8399x - 0.1344 |
| $R^2$ | 0.9931 | 0.9982 | 0.9984 | 0.9982 |

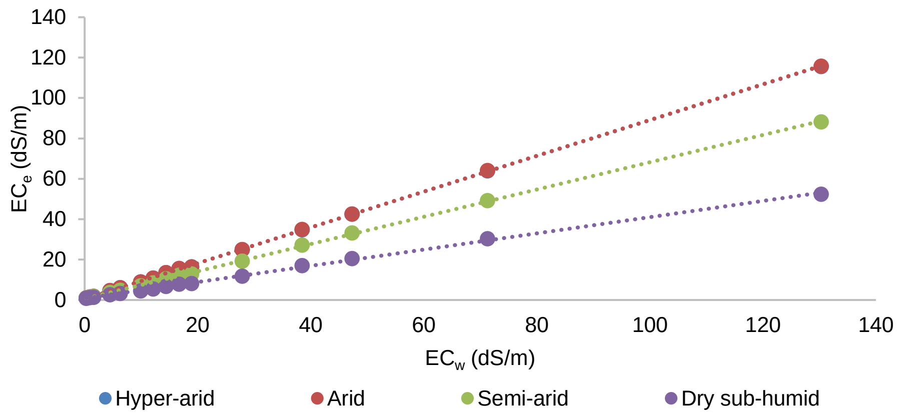



| Soil | Arenosol | Gypsisol | Planosol | Vertisol |
|---|---|---|---|---|
| Curve equation | y = 0.547x + 1.1376 | y = 0.4242x + 2.6289 | y = 0.8812x - 0.7401 | y = 0.9411x + 1.7476 |
| $R^2$ | 0.9545 | 0.8398 | 0.9956 | 0.9829 |

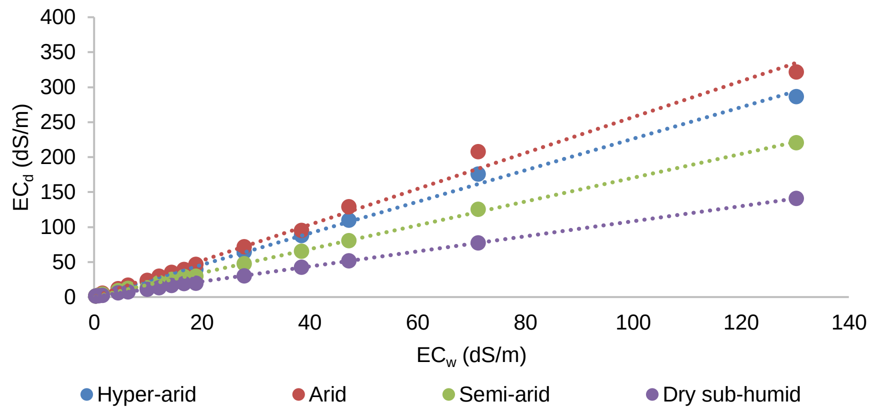



| Soil | Arenosol | Gypsisol | Planosol | Vertisol |
|---|---|---|---|---|
| Curve equation | y = 1.0457x + 3.1721 | y = 0.9066x + 4.519 | y = 1.3185x - 0.5524 | y = 1.3155x + 2.0803 |
| $R^2$ | 0.9543 | 0.9041 | 0.9951 | 0.9886 |

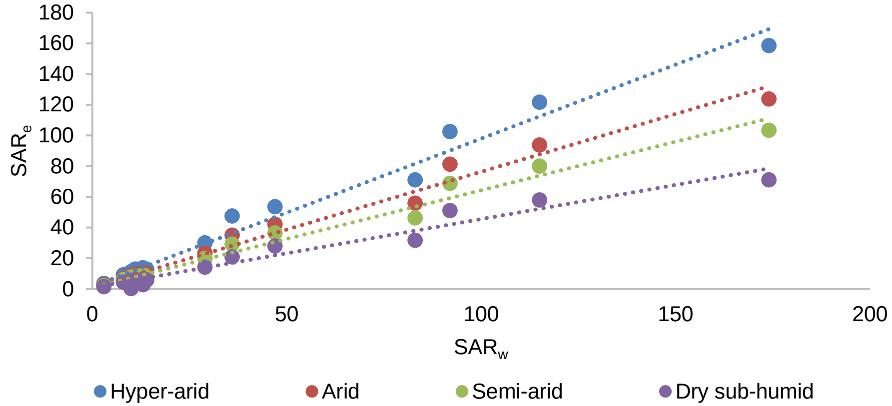
**Figure 5-4 Influence of soil type on the ratios $EC_e/EC_w$, $EC_d/EC_w$, $EC_e/EC_d$, $SAR_e/SAR_w$, $SAR_d/SAR_w$ and $SAR_e/SAR_d$, all climates combined**



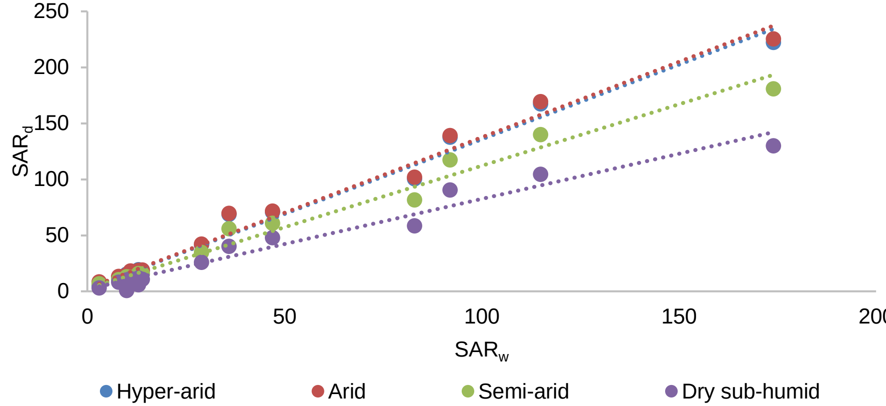

| Climate | Hyper-arid | Arid | Semi-arid | Dry sub-humid |
|---|---|---|---|---|
| Curve equation | y = 0.8864x + 0.4876 | y = 0.8851x + 0.4889 | y = 0.6752x + 0.676 | y = 0.4016x + 0.8466 |
| $R^2$ | 0.9999 | 0.9999 | 0.9998 | 0.9991 |




| Climate | Hyper-arid | Arid | Semi-arid | Dry sub-humid |
|---|---|---|---|---|
| Curve equation | y = 2.2527x + 1.0602 | y = 2.5644x + 0.8079 | y = 1.7003x + 0.5167 | y = 1.0772x + 0.6673 |
| $R^2$ | 0.9966 | 0.9923 | 0.9991 | 0.9998 |

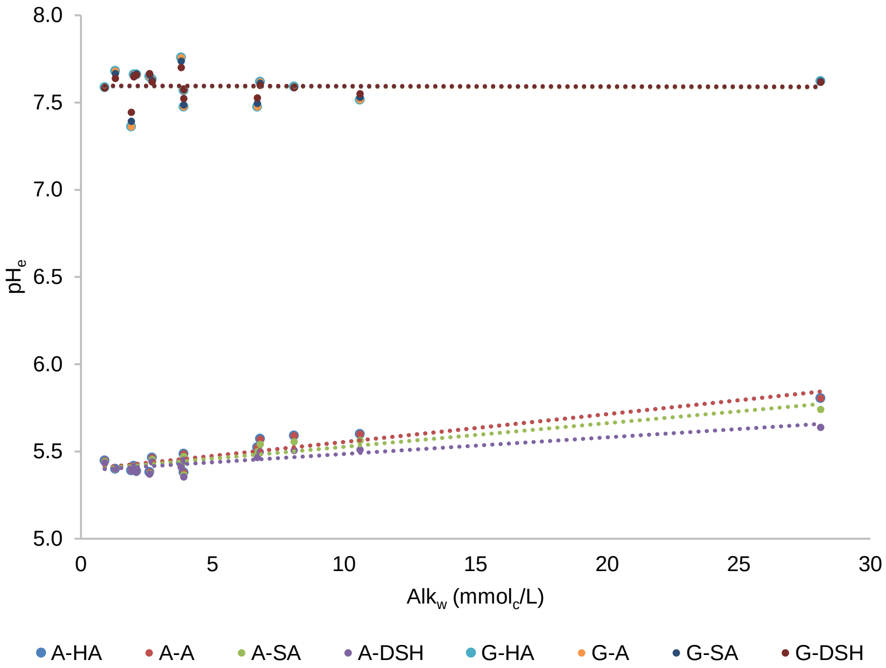



| Climate | Hyper-arid | Arid | Semi-arid | Dry sub-humid |
|---|---|---|---|---|
| Curve equation | y = 0.9634x + 1.6134 | y = 0.7525x + 1.1241 | y = 0.6326x + 1.0169 | y = 0.4452x + 1.0196 |
| $R^2$ | 0.9768 | 0.9783 | 0.9738 | 0.9577 |

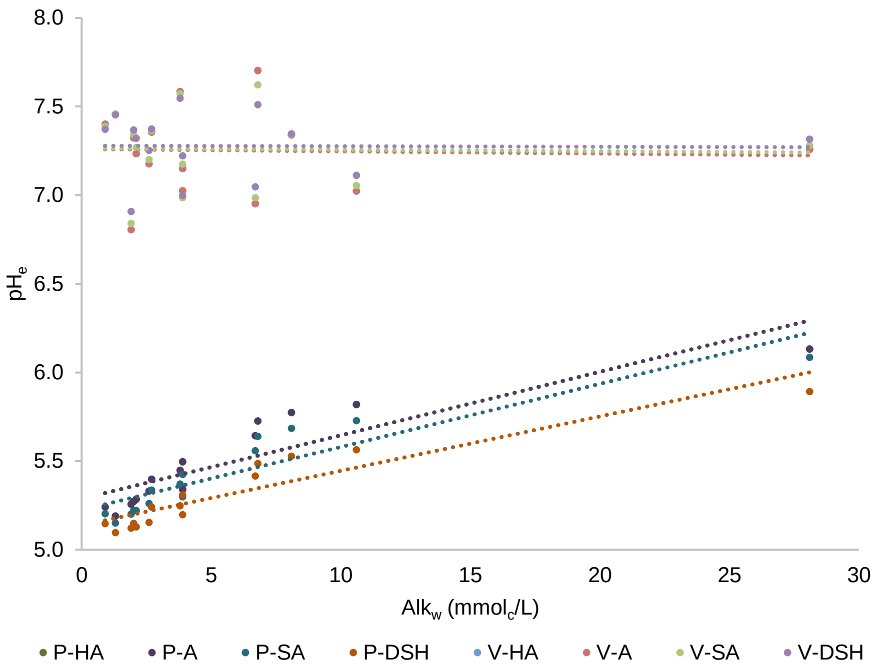



| Climate | Hyper-arid | Arid | Semi-arid | Dry sub-humid |
|---|---|---|---|---|
| Curve equation | y = 1.3336x + 2.3575 | y = 1.3484x + 2.6874 | y = 1.0983x + 2.2757 | y = 0.8062x + 1.8984 |
| $R^2$ | 0.9812 | 0.9814 | 0.9762 | 0.9642 |

**Figure 5-5 Influence of climate aridity on the ratios $EC_e/EC_w$, $EC_d/EC_w$, $EC_e/EC_d$, $SAR_e/SAR_w$, $SAR_d/SAR_w$ and $SAR_e/SAR_d$, all soils combined.**




| Soil/climate | Curve equation | $R^2$ |
|---|---|---|
| Arenosol/Hyper-arid (A-HA) | y = 0.016x + 5.3951 | 0.8557 |
| Arenosol/Arid (A-A) | y = 0.0159x + 5.395 | 0.8556 |
| Arenosol/Semi-arid (A-SA) | y = 0.0136x + 5.391 | 0.8396 |
| Arenosol/Dry sub-humid (A-DSH) | y = 0.0095x + 5.3904 | 0.7926 |
| Gypsisol/Hyper-arid (G-HA) | y = -0.0002x + 7.5922 | 0.0002 |
| Gypsisol/Arid (G-A) | y = -0.0002x + 7.5922 | 0.0002 |
| Gypsisol/Semi-arid (G-SA) | y = -0.0002x + 7.5944 | 0.0002 |
| Gypsisol/Dry sub-humid (G-DSH) | y = -0.0002x + 7.5973 | 0.0004 |




| Soil/climate | Curve equation | $R^2$ |
|---|---|---|
| Planosol/Hyper-arid (P-HA) | y = 0.0358x + 5.2865 | 0.8012 |
| Planosol/Arid (P-A) | y = 0.0358x + 5.287 | 0.8007 |
| Planosol/Semi-arid (P-SA) | y = 0.0356x + 5.2235 | 0.8428 |
| Planosol/Dry sub-humid (P-DSH) | y = 0.0307x + 5.1377 | 0.8618 |
| Vertisol/Hyper-arid (V-HA) | y = -0.0013x + 7.2605 | 0.0013 |
| Vertisol/Arid (V-A) | y = -0.0013x + 7.2605 | 0.0013 |
| Vertisol/Semi-arid (V-SA) | y = -0.0008x + 7.2641 | 0.0006 |
| Vertisol/Dry sub-humid (V-DSH) | y = -0.0003x + 7.2791 | 0.0001 |

**Figure 5-6 Soil $pH_e$ and irrigation water alkalinity of the average soil depth 0-60 cm at equilibrium following irrigation with the 15 PWs on Arenosol, Gypsisol, Planosol and Vertisol under hyper-arid, arid, semi-arid and dry sub-humid climates.**

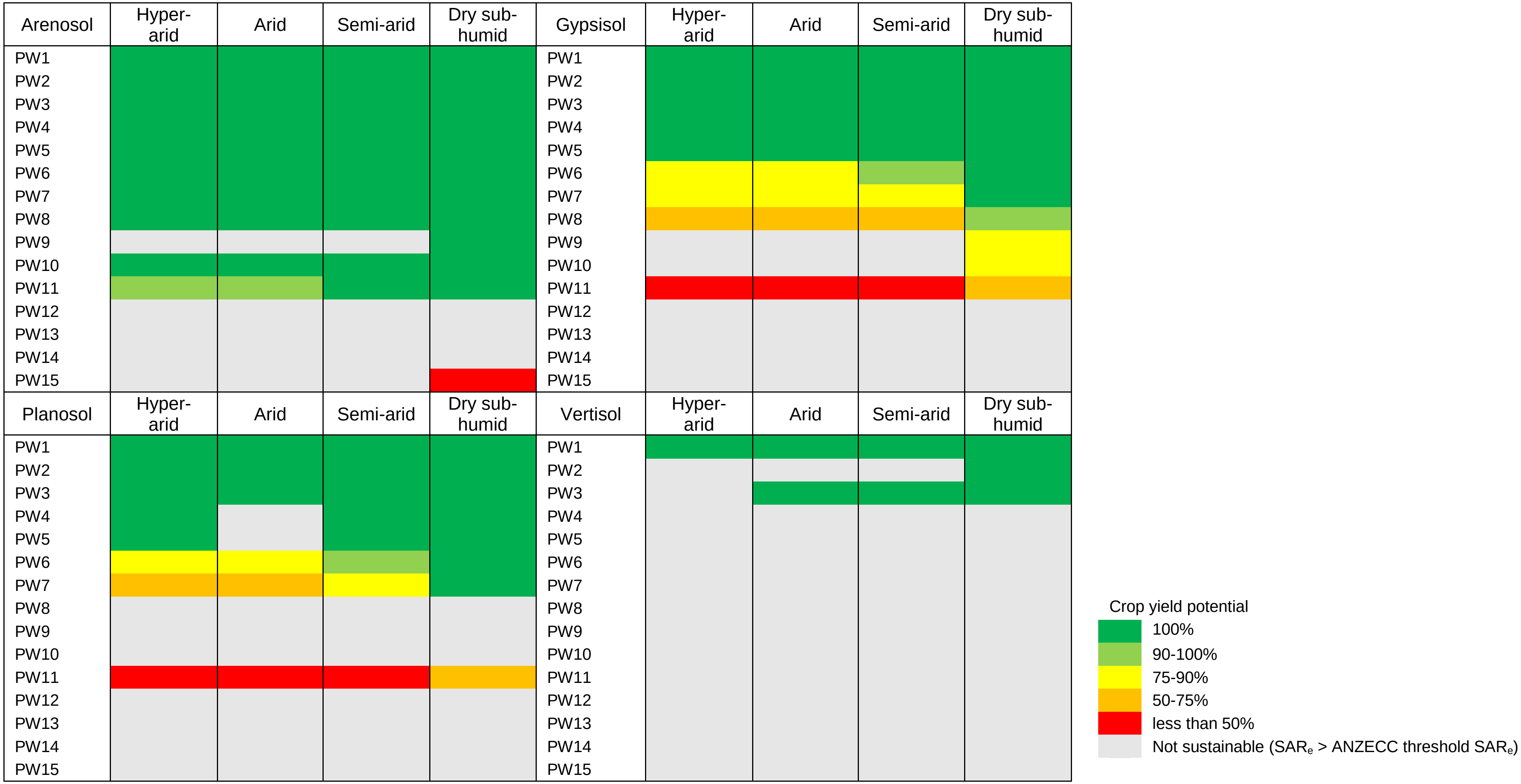



**Figure 5-7 Estimated crop yield potential of sugar beet irrigated with 15 PWs under hyper-arid, arid, semi-arid and dry sub-humid climate and on Arenosol, Gypsisol, Planosol and Vertisol.**

## 5.4 Discussion

### 5.4.1 Soil salinity and sodicity

In the long-term, PW quality is the most influential factor on soil salinity and sodicity. In all scenarios, increasing $EC_w$ and $SAR_w$ led to a higher degree of soil salinisation and/or sodification (Figure 5-2). This is illustrated by the values of the coefficients of determination ($R^2$) between $EC_w$ and $EC_e$ on the one hand, and between $SAR_w$ and $SAR_e$ on the other hand. Therefore, independently of soil type (Figure 5-4) and climate aridity (Figure 5-5); irrigation using PWs with higher $EC_w$ led to higher soil $EC_e$. Similarly, irrigation using PWs with higher $SAR_w$ led to higher soil $SAR_e$. However, $R^2$ was lower for soil and water SAR than for soil and water EC, thus, soil sodicity ($SAR_e$) was less dependent on water sodicity ($SAR_w$) than soil salinity ($EC_e$) was on water salinity ($EC_w$) as other parameters related to the soil interfere and must be taken into account to predict the $SAR_e$.

The simulations have shown that the soil types differed regarding their levels of vulnerability to sodification (Figure 5-2). The clay content on which the ANZECC $SAR_e$ threshold values are based (Shaw et al., 2011), has a key role in determining the sensitivity of soil to sodification. Indeed, high $SAR_e$ causes high exchangeable sodium percentage (ESP), which destabilises soil particles due to clay swelling and dispersion. As a result, soil pores clog and its hydraulic conductivity and thus, the ability to supply water to crops, decreases. Eventually, the sensitivity of lands to erosion and desertification are both amplified (Dregne, 1983; Qadir and Schubert, 2002).

Besides, the soil factors that most influence the long-term $SAR_e$ are gypsum content and drainage ability. Firstly, soil gypsum buffers soil sodicity by dissolving into the soil solution. Secondly, the drainage properties of the soil moderate soil sodification because as the percentage of sand increases, the water content at field capacity decreases and leaching increases and thus, the sodium concentration in the soil solution decreases, however, calcium concentration is more constant because this ion also dissolves into the soil solution from calcium minerals (calcite and gypsum). As a result, the soils with the highest gypsum content, highest sand content and thus, lowest field capacity were the less prone to sodification. This was shown by the simulations, in which Vertisol and Planosol resulted to be the most sensitive to sodification whereas Gypsisol and Arenosol were the least vulnerable to sodification (Figure 5-4). Field experiments have confirmed that the sensitivity of soil to sodification can be anticipated knowing the soil clay content and the water retention properties (Levy, Goldstein and Mamedov, 2005). The buffer effect of soil gypsum on the $SAR_e$ has also been highlighted in an experimental-modeling study with PW in semi-arid Wyoming, USA (Engle et al., 2011).

The long-term $EC_e$ and therefore, the risk of soil salinisation also depends on soil type (Figure 5-2). Indeed, soils with low water content at field capacity, in general, drain more easily as they usually have large pores. Thus they retain less water and leach more salt compared to soils with a higher field capacity. As a consequence, Vertisol and Planosol were the most sensitive to salinisation whereas Gypsisol and Arenosol were the least vulnerable to salinisation (Figure 5-4). An irrigation trial with PW conducted in semi-arid Northeast Brazil on a sandy soil showed that the high porosity of Arenosols decreased $EC_e$ through facilitated drainage (Sousa et al., 2017).

Climate affects the relationships between water and soil salinity as well as between water and soil sodicity. Indeed, this was shown by the simulations in which the slopes of the curves $EC_e = f(EC_w)$ and $SAR_e = f(SAR_w)$ increased following aridity from dry sub-humid to hyper-arid (Figure 5-5) making irrigation with PW less sustainable (Figure 5-2). This is explained by the double effect of rain which both dilutes the soil solution and transports salts out of the root zone reducing $EC_e$. Equally important, higher evaporation increases the salt concentration of the soil-water. Lower aridity or higher humidity decreases the concentration of soil sodium while the concentrations of magnesium and, over all, calcium, are buffered by the minerals calcite and gypsum generally present in dryland soils (Koohafkan and Stewart, 2008). The ability of climate aridity to influence $EC_e$ and $SAR_e$ were observed in a field trial carried out with PW under humid sub-tropical climate in Alabama, USA (Mullins and Hajek, 1998). Thus, the

AI should be considered when assessing the agro-environmental sustainability of irrigation with PW.

The crop has an indirect effect in determining the $EC_e$ and $SAR_e$. Sugar beet was considered in this study, however, other crops would have required different irrigation amounts and schedules. As the irrigation volume and its distribution play a key role for the $EC_e$ and $SAR_e$, a crop with lower water needs compared to sugar beet implies less irrigation water and thus less salt input to the soil. Also, if the crop requires water in the period of the year when rainfall is the highest and evaporation the lowest, it could significantly reduce the soil $EC_e$ and $SAR_e$ due to less irrigation, more salt leaching, and less water evaporation.

In brief, well-drained soils with low clay content and significant gypsum content in the relatively most humid regions must be chosen in priority for preventing soil salinisation and sodification. In addition, a drought-resistant crop growing when the AI is the highest during the year must be privileged for improving the agro-environmental sustainability of irrigation with PW.

Most of the studies referring to the suitability of using PW for irrigation use the FAO guidelines (Ayers and Westcot, 1985) for assessing potential risks to the soil and crop (Beletse et al., 2008; Guerra, Dahm and Dundorf, 2011; Martel-Valles et al., 2014; Martel-Valles, Benavides-Mendoza and Valdez-Aguilar, 2017; Martel-Valles, Foroughbakchk-Pournavab and Benavides-Mendoza, 2016; Myers, 2014). The results obtained in the current paper could help to refine these standards when assessing the agro-environmental sustainability of irrigation with PW. Indeed, the limitations of the FAO guidelines are that they are not specific, therefore they may be too conservative for environments with low vulnerability to salinisation and sodification (e.g. well-drained soils in dry sub-humid climates). Although the ANZECC guidelines are more specific by discriminating among soil types according to their clay content, they do not consider the degree of aridity in the determination of threshold $EC_w$ and $SAR_w$ values to prevent soil salinisation and sodification.

### 5.4.2 Soil pH

The simulations have shown that the $pH_e$ was positively influenced, although in a limited proportion, by irrigation water alkalinity ($Alk_w$) with $0.79 < R^2 < 0.86$ for Arenosol and Planosol (which have low carbonate content) whereas the $R^2$ were low (> 0.01) for Gypsisol and Vertisol which both have the highest carbonate content (Figure 5-6). In fact, soil $pH_e$ is mainly determined by the soil $CaCO_3$-$CO_2$ system. The limited influence of irrigation water on $pH_e$ has also been highlighted in an irrigation trial with PW on a Vertisol in dry sub-humid Australia (Bennett et al., 2016). Decreasing climate aridity slightly reduced $pH_e$ on Arenosol and Planosol (Figure 5-6), that is a common observation in arid environments (Jiao et al., 2016).

Soil amendments and fertilisers, which are not considered in this study may have a significant effect on the $pH_e$ which must be anticipated if they are used along with irrigation.

### 5.4.3 Crop yield

Crop yield can be maximised by reducing the $EC_e$ below 7 dS/m which is the crop threshold value for an optimal yield (Figure 5-7). The irrigation volume can be increased to leach more salt or PW can be blended with another water of lower salinity to reduce the $EC_e$. Eventually, a crop with a higher tolerance to salinity can be cultivated if it is adapted to climates and soils in drylands.

Although crop production is of primary relevance for farmers, the O&G industry does not necessarily have the same target. If managing PW in an irrigation project remains less expensive compared to conventional disposal options, yield as low as 50% of crop optimum could be satisfactory.

### 5.4.4 Drainage water

The simulations demonstrated that the qualities of DW and irrigation water were closely related. Increasing $EC_w$ and $SAR_w$ led to higher $EC_d$ and $SAR_d$ (Figure 5-3) due to positive correlations between $EC_w$ and $EC_d$ on the one hand and between $SAR_w$ and $SAR_d$ on the other hand. Furthermore, the soil type was determinant in defining DW quality. The slopes of the curves $EC_d = f(SAR_w)$ and $SAR_d = f(SAR_w)$ show that well-drained soils such as Arenosol generated the most saline DWs whereas Planosol and Vertisol generated the most sodic DWs. Climate interferes as decreasing aridity lowered $EC_d$ and $SAR_d$ by diluting the salinity of DW because of lower evaporation and/or higher precipitation. The crop indirectly determines the $EC_d$ and $SAR_d$ through the irrigation volume and irrigation schedule. Also, if the crop requires water in the period of the year when rainfall is the highest and evaporation the lowest, rain could either increase the $EC_d$ and $SAR_d$ due to more salt leaching or reduce the $EC_d$ and $SAR_d$ due to increasing dilution of DW. Notwithstanding, if the crop requires more water when it rains more, then less PW will be used accordingly, and therefore, less salt will be introduced into the soil.

If irrigation can be agro-environmentally sustainable from a soil-plant point of view, DW leaving the root zone must be properly managed to avoid transferring the salinity and sodicity hazards from the soil to the groundwater. Indeed, in the simulations, DW was always more saline and more sodic than the associated irrigation water. DW would continue to percolate deeper into the soil, eventually reaching the aquifer. This risk must be anticipated if the $EC_d$ is higher than the EC of the aquifer although it might not be a problem in some dry areas where groundwater is deep and/or already brackish (Vengosh, 2014). Alternatively, DW can be captured by means of drainage systems and reused, treated or disposed of. Disposal options such as pond evaporation, discharge to the sea or deep-well injection could be considered (Jiménez et al., 2018). Notwithstanding, irrigation would at least reduce the volume of saline water that had to be disposed of, compared to the original volume of PW, therefore it would be cheaper to manage.

### 5.4.5 Limitations

The carried out simulations are exploratory and limitations related to the model, the method and the guidelines used in this study are acknowledged.

The SALTIRSOIL_M model does not simulate crop salt uptake, therefore, where this is significant, it could overestimate the soil salinity. Although salt uptake is usually negligible compared to the salt load brought by irrigation water, sugar beet salt uptake can reach more than 1.2 t/ha of sodium and potassium annually (Cumo, 2013). Given that irrigation of sugar beet under dry sub-humid climate requires 590 mm of water annually (Table 5-1), irrigation would bring between 0.15 to 118 t/ha/year of sodium and potassium, respectively if PW1 and PW15 are used, that is between 13% and 10000% of the salt load that would be exported by the crop. Thus, the salt load extracted from the soil solution would not significantly change the salt concentration of the worst case scenarios (e.g. irrigation in a hyper-arid climate with PW15) but would positively contribute to the agro-environmental sustainability of the simulated scenarios where the salt load brought by irrigation water was relatively low.

Tolerance to salinity and optimum soil $pH_e$ vary widely among crops, consequently, different crop threshold levels will also impact soil salinity and sodicity as well as crop yield. The soil salinity and sodicity, and crop yield patterns described in Figure 5-2 and Figure 5-7 would be different if another crop would have been chosen instead of sugar beet.

From an agricultural point of view, the agro-environmental sustainability of irrigation with PW is mainly, but not exclusively a salinity issue. Other constituents of concern, such as heavy metals, organic compounds and radioelements exist in PW, and their presence and concentrations depend on PW origin (Alley et al., 2011) and treatment processes (Fakhru'l-Razi et al., 2009). On the one hand, the high $pH_e$ and the low SOM content of most soils in dryland limit the bioavailability of heavy metals, but on the other hand, high soil $EC_e$ increases this risk (Singh et al., 2009). Although the risks linked to other components of PW are not as concerning as those related to salts, they still deserve to be specifically assessed and included in potential guidelines or frameworks aiming to support PW reuse in irrigation.

Although the SALTIRSOIL_M model has been calibrated and validated against field results in a dry region with slightly to moderately saline irrigation water (Visconti et al., 2014), this has not yet been done for the environments simulated in the current paper. Therefore, the model results should be used in the context of refining conceptual and mathematical models for future research based on a comparison of simulated and field results under specific environments. This would also help to define the sensitivity of agro-environmental sustainability indicators (e.g. $EC_e$, $SAR_e$, ions contents, $pH_e$, and alkalinity of the soil solution) to parameters and processes that are considered or not considered in the model and which in this case, would require further characterisation and study.

## 5.5 Conclusions

PW is generated continuously, independent of climatic conditions and could be a useful water resource for irrigators in drylands. For petroleum firms, its reuse in irrigation is an alternative to conventional disposal practices which are environmentally risky, increasingly regulated and costly. Depending on the soil and climate, the low quality of PW, particularly its high salinity and sodicity, can degrade soil fertility and aquifers to varying degrees.

Irrigation water quality and climatic aridity drive the balance of salt inputs and outputs of the system, while the irrigation practice and soil type control the salt removal processes and leaching through drainage. The main threat to the soil from irrigation with PW is sodification, the risk of which largely depends on the clay content of the soil, PW sodicity ($SAR_w$), and climate aridity. If PW quality cannot be improved (e.g. by blending PW with fresh water or by desalinating PW), PW irrigation can only be used in the long term, in environments that are less vulnerable to soil salinisation and sodification. Well-drained soils with low water content at field capacity (e.g. Arenosol) are less vulnerable to salinisation, whilst a relatively high gypsum content (e.g. Gypsisol) provides resistance against sodification. On the contrary, clayey soils with a high field capacity water content and a low gypsum content must be avoided, as the soil structural stability, as well as a tolerable $EC_e$ for the crop, cannot be maintained on the long-term.

Simulations with the sugar beet crop in drylands demonstrated that crop yield could be adequate (> 50% of optimum) and even improved by using PW with lower EC in well-drained soils. Soil $pH_e$, which also impacts crop yield through nutrient availability, was not significantly affected by irrigation water quality since it largely depends on the natural soil $CaCO_3$-$CO_2$ content.

Finally, drainage water quality is closely linked to the quality of PW but is also influenced by the soil type and climate aridity. The impact of drainage water on the aquifer must be considered and measures such as drainage-water reuse or disposal implemented accordingly for achieving agro-environmentally sustainable irrigation with PW.

The modelling has demonstrated the importance of the clay and gypsum contents of the soil, and of climate aridity (i.e. AI) to assess the suitability of PW for irrigation. Based on the simulation results, irrigation with PW is likely to be agro-environmentally sustainable on sandy soils if PW has an EC ≤ 28 dS/m and a SAR ≤ 36. Loamy and gypsiferous soils can cope with PW with an EC ≤ 14 dS/m and a SAR ≤ 29 unless the climate is dry sub-humid (0.50 ≤ AI < 0.65), in this case, PW with an EC as high as 28 dS/m and a SAR ≤ 83 can be used for long-term irrigation of halotolerant crops. Salinisation and sodification can be avoided in sandy clay loam soils if the PW has an EC ≤ 12 dS/m and a SAR ≤ 6. Lastly, clayey soils should not be irrigated with PW with an EC ≥ 2 dS/m and a SAR ≥ 10 except if the climate is dry sub-humid (or wetter) where PW SAR can be as high as 13. These thresholds values need to be confirmed through further field study and would only be adopted to manage PW through irrigation without targeting optimum crop yield. On a sample of 474 PWs collected worldwide, about 6% to 8% of PWs fall within the threshold values for the least vulnerable environments (i.e. sandy soil and loamy gypsiferous soil in dry sub-humid climate) whereas only 2% of the PWs corresponded to the required quality for irrigation on clayey soil (Echchelh, Hess and Sakrabani, 2018).

Future work should be carried out to explore how management practices such as over-irrigation, PW blending with fresh water, PW desalination and gypsum amendments could help to improve irrigation agro-environmental sustainability with the PWs that are too saline and/or too sodic to be used in long-term irrigation. A complete sustainability assessment would also require an analysis of the impacts of other constituents of concern such as heavy metals, organic compounds and radioelements on soil, crop and groundwater. These studies could be synthesised in a sustainability assessment framework specifically designed for the O&G sector to encourage cooperation between the O&G industry and irrigators for sustainable reuse of PW in drylands.

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# 6 AGRO-ENVIRONMENTAL SUSTAINABILITY AND FINANCIAL COST OF REUSING PRODUCED WATER FOR AGRICULTURAL IRRIGATION IN QATAR[3]

## 6.1 Introduction

Oil and gas (O&G) exploitation generates large volumes of 'produced water' (PW) which is the main waste stream derived from this industry (Veil, 2011). PW is naturally present in the hydrocarbon-bearing strata and flows up to the surface when O&G are extracted. It can also be water returning to the surface after being artificially injected to enhance O&G production (Engle, Cozzarelli and Smith, 2014). Whereas half of global PW volume is beneficially reused to increase hydrocarbon recovery, the other half is managed through injection into deep disposal wells or treated and discharged on the surface without being reused (Echchelh, Hess and Sakrabani, 2018). This is problematic because deep-well injection has a high energy and carbon intensity, and therefore is costly (Arthur et al., 2011). Besides, this practice is environmentally risky, as it can contaminate aquifers (Hagström et al., 2016) and induce earthquakes (Walsh and Zoback, 2015). Surface discharge is also controversial due to the risks of soil and water pollution (Christie, 2012; Konkel, 2016). Consequently, harsher environmental regulations are being developed demanding advanced PW treatment before discharging (Fakhru'l-Razi et al., 2009) or simply banning it completely (Igunnu and Chen, 2014). In this context, sustainable alternatives to existing PW management practices are needed. Reusing PW for irrigation of crops is an opportunity to reduce the dependence of the O&G industry on traditional disposal techniques while providing significant volumes of water to croplands located in O&G basins (Echchelh, Hess and Sakrabani, 2018).

Qatar is an example of how the O&G industry's quest for reducing PW disposal could meet the country's environmental and agricultural ambitions (Raja and El-Hadi, 2012). Qatar has a hyper-arid climate (Table 6-1) with an aridity index of 0.02 (Cherlet et al., 2018), it has very limited rainfall making its agriculture totally dependent on irrigation using the country's scarce freshwater resources which are almost totally located in its aquifers (FAO, 2009b). Groundwater reserves and quality have been declining, mainly

[3] This chapter is in the form of a paper to be submitted to Agricultural Water Management.

because of overexploitation by the agricultural sector which accounts for 92% of groundwater abstraction (Ministry of Development Planning and Statistics, 2017). The government aims to restore aquifers by limiting the volume of groundwater extracted and by developing the reuse of treated sewage effluent (TSE) in irrigation (Jasim et al., 2016). In the meantime, Qatar operates the largest gas reservoir in the world known as North Field (Fulks and Kumar, 2015). North Field generates about 1.4 million $m^3$/year of PW, representing the largest wastewater stream in the country (Al-Kaabi, 2016), and the equivalent of 3.2% of Qatar's average annual water balance, and 0.6% of the annual groundwater volume used in agriculture (Ministry of Development Planning and Statistics, 2017). Moreover, North Field PW is of relatively good quality with an EC of 7.1 dS/m (Table 6-2) whereas the PW of Qatar's offshore oil fields have an EC above 100 dS/m (Ahan, 2014). Therefore, this potential supply of irrigation water could help Qatar to reduce groundwater abstraction while increasing crop production and achieve its food security plan (Qatar e-government, 2019a). Short term risks such as the economic blockade on Qatar as well as longer-term trends such as population growth and climate change reinforce the need for developing local non-conventional irrigation water resources (Miniaoui, Irungu and Kaitibie, 2018).

Unfortunately, PW reuse in irrigation is challenging mainly because PW salinity, sodicity and its content in heavy metals frequently exceed the threshold irrigation water quality levels (Alley et al., 2011; Echchelh, Hess and Sakrabani, 2018). In fact, a greenhouse experiment conducted in Qatar, used PW diluted with tap water in a 1:9 ratio to irrigate several crops on a sandy soil amended with peat moss. The diluted PW had an electrical conductivity (EC) of 33.6 dS/m and a sodium adsorption ratio (SAR) of 14. Its application to the soil dramatically increased the soil contents in $Na^+$, $K^+$, $Ca^{2+}$, $Mg^{2+}$, $Cl^-$, $NO_3^-$ and $SO_4^{2-}$ compared to the soil irrigated with tap water. Irrigation with PW resulted in reduced longevity of crops. Indeed, the average number of plant survival days was 35 days for saltwort, 30 days for alfalfa, 20 days for sorghum, and less than 5 days for maize and sunflower (Atia, 2017). Similarly, another greenhouse experiment carried out in Qatar with a higher quality of PW (EC = 2.7 dS/m, SAR = 0.34) demonstrated the detrimental effect of PW on crops. Indeed, although the soil salinity and ionic composition were not significantly different between the soil irrigated with PW and the soil irrigated with tap water, the alfalfa irrigated with PW had lower average fresh weight, leaf area, number of leaves per plant and plant height compared to the alfalfa irrigated with tap water (Ibrahim, Marroff and Wafi, 2009). These observations are not exclusive to Qatar as several experiments carried out in dry areas have shown that irrigation with PW increases the soil EC and SAR (Echchelh, Hess and Sakrabani, 2018).

However, these short-term (1–3 years) field experiments do not inform about the agro-environmental sustainability of irrigation with PW in Qatar, that is, the extent of soil degradation caused by PW sodicity and the decline of crop productivity caused by PW salinity in the long-term (i.e. indefinitely). This information is critical as Qatari gas reserves are projected to last 138 years at the current production level (The Oil & Gas Year, 2019), thus, PW could potentially be used in irrigation for decades. Ideally, long-term field experiments combined with models could be conducted to provide better predictions of the sustainability of irrigation with PW.

Another limit of the field experiments conducted in Qatar is that they were not applicable to large irrigation schemes. Indeed, Atia (2017) diluted PW with tap water to mitigate the negative impacts of PW salinity and sodicity on the soil and on the crop, but this would be extremely costly at a commercial scale. Cheaper water resources, such as TSE or desalinated PW could be used to blend PW and improve irrigation water quality. Besides, other techniques such as over-irrigation to increase salt leaching could be used in conjunction with PW blending to control soil salinity and sodicity.

Finally, along with the possibility of having agro-environmentally sustainable irrigation with PW, the cost of this management practice in Qatar remains unknown.

This paper aims to address these knowledge gaps by first, identifying possible agro-environmentally sustainable irrigation strategies on sugar beet with gas-field PW in Qatar, using over-irrigation, PW blending and PW desalination to protect the soil and the aquifer from salinisation and sodification. The second objective of this study is to estimate the costs of these potential agro-environmentally sustainable irrigation scenarios.

In greenhouse and field experiments, halotolerant crops such as barley, sorghum or cotton are generally chosen because of their tolerance to PW salinity (Echchelh, Hess and Sakrabani, 2018). Targeting a successful crop cultivation requires the selection of a salt-tolerant crop that is adapted to the hyper-arid climate and poor shallow soil typical of Qatar. Thus, potential crops include sugar beet, courgettes, kale, barley, sorghum, cotton, oat, soya bean, date palm and Natal plum. In this study, tropical sugar beet was chosen as an exemplar crop. Although sugar beet is not currently grown in Qatar (Ministry of Development Planning and Statistics, 2016), it could be of interest to partly supply the country's first sugar refinery (Saul, Finn and El Dahan, 2018). Developing sugar production is also in alignment with Qatar's policy aiming to improve national food security. From an agronomic point of view, sugar beet is particularly appropriate because of its tolerance to salt (Tanji and Kielen, 2002), sodium and chloride (Wakeel, Steffens and Schubert, 2010), its adaptation to dry climates (Chatin et al., 2004; Nilsson, 2005) and to sandy soils, such as Calcisols (SESVanderHave, 2016), the dominant soil type in northern Qatar where North Field is located (IUSS Working Group WRB, 2015) (Table 6-3). Finally, this crop is a raw material for multiple products such as foodstuff (i.e. sugar), animal feed (i.e. pellets and molasses) and biofuel.

**Table 6-1. Parameters of climate, crop development and reference irrigation regime used in the simulations**

| | Parameter | January | February | March | April | May | June | July | August | September | October | November | December | Total |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Doha Airport meteorological station | P (mm) | 10 | 20 | 10 | 10 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 10 | 60 |
| | ETo (mm) | 102 | 104 | 155 | 214 | 302 | 342 | 302 | 281 | 215 | 188 | 138 | 108 | 2450 |
| | I (mm) | 199 | 168 | 122 | 0 | 0 | 0 | 0 | 116 | 94 | 122 | 101 | 142 | 1064 |
| Crop growth | $K_{cb}$ | 1.15 | 0.70 | 0.70 | 0 | 0 | 0 | 0 | 0.43 | 0.70 | 0.80 | 1.20 | 1.20 | - |
| | Root depth (cm) | 100 | 100 | 100 | 0. | 0 | 0 | 0 | 30 | 49 | 56 | 84 | 92 | - |

P: precipitation; ETo: reference evapotranspiration; I: base irrigation regime covering 100% of the crop water needs; $K_{cb}$: basal crop coefficient.

**Table 6-2. Quality of the different waters used for irrigation simulations (all ions contents are expressed in mmol/L, or in $mmol_c$/L of [$CaCO_3$] equivalent for the alkalinity, and the electrical conductivity in dS/m).**

| | [$Na^+$] | [$K^+$] | [$Ca^{2+}$] | [$Mg^{2+}$] | [$Cl^-$] | [$NO_3^-$] | [$SO_4^{2-}$] | $Alk_w$ | $EC_w$ | $SAR_w$ | $pH_w$ |
|---|---|---|---|---|---|---|---|---|---|---|---|
| PW | [a]52.12 | [a]2.58 | [a]7.13 | [a]1.85 | [a]82.39 | [a]0.04 | [b]0.56 | [c]3.00 | [a]7.04 | [a]17.39 | [a]4.43 |
| ROPW | [d]0.42 | [d]0.07 | [d]0.00 | [d]0.01 | [d]1.07 | [d]0.00 | [d]0.34 | [d]0.00 | [d]0.17 | [d]4.33 | [d]6.12 |
| TSE | [e]15.70 | [e]0.95 | [e]12.40 | [e]6.22 | [e]14.10 | [f]0.14 | [e]25.00 | [e]3.92 | [e]3.83 | [e]3.64 | [e]5.15 |

PW: produced water, TSE: treated sewage effluent, ROPW: reverse osmosis-treated produced water, $EC_w$: water electrical conductivity, $SAR_w$: sodium adsorption ratio of the water, $Alk_w$: water alkalinity as [$CaCO_3$] equivalent.

[a](Al-Kaabi, 2016), [b](Janson et al., 2015), [c](Ahan, 2014), [d](Ersahin et al., 2018), [e](Ahmad, 1989), [f](Dalahmeh and Baresel, 2014).

**Table 6-3. Soil parameters used in the simulations**

| Soil type (FAO's RSG) | Soil layer (cm) | Hydrophysical | | | USDA texture (%) | | | Chemical | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | $\rho_b$ (g/cm$^3$) | $\theta_{fc}$ (%) | $\theta_{pwp}$ (%) | Sand | Silt | Clay | pH | Gypsum (%) | CCE (%) | SOM (%) | log $pCO_2$ |
| Calcic Yermosol | Topsoil 0–30 | 1.7 | 12 | 5 | 86 | 9 | 5 | 8.1 | 0.1 | 5.9 | 0.55 | -3 |
| | Subsoil 30–100 | 1.6 | 12 | 5 | 80 | 11 | 9 | 8.2 | 0.9 | 3.0 | 0.40 | -3 |

FAO's RSG: FAO's Reference Soil Groups, $\rho_b$: bulk density; $\theta_{fc}$: soil volumetric water content at field capacity; $\theta_{pwp}$: soil volumetric water content at permanent wilting point; CCE: calcium carbonate equivalent, SOM: soil organic matter, log $pCO_2$: log value of the $CO_2$ partial pressure.

## 6.2 Methods

This paper combines a modelling approach to simulate the impacts of irrigation with PW on soil salinity and sodicity with a cost analysis to estimate the operating costs of potential agro-environmentally sustainable irrigation scenarios (Figure 6-1).

### 6.2.1 Definition of agro-environmental sustainability

Sustainability is generally defined as meeting current human needs without compromising the ability of future generations to meet their own needs (Held, 2001). In this paper, irrigation with PW is considered agro-environmentally sustainable if it conserves the existing soil and groundwater capital for future generations. For this, it is necessary to prevent the salinisation and sodification of the soil and of the aquifer as a result of irrigation with PW. In order to quantify these degradations, indicators were selected.

### 6.2.2 Agro-environmental sustainability indicators and agro-environmental sustainability assessment

To estimate the risk of the destabilising the soil structure, the sodium adsorption ratio of the soil saturation extract ($SAR_e$) was selected as an indicator. This indicator is widely used to estimate the risk of soil sodification as a result of irrigation (Hillel, 2000) and can be compared to the Australian and New Zealand Environment Conservation Council threshold $SAR_e$ values informing about the risk of soil structural instability (ANZECC, 2000). The ANZECC guidelines were used as a reference to study the risks and feasibility of using PW in irrigation under dry climates in Australia and in sub-Saharan Africa (Horner, Castle and Rodgers, 2011; Mallants, Šimůnek and Torkzaban, 2017; Shaw et al., 2011). The threshold $SAR_e$ was set at 20 as soils in northern Qatar are sandy with a clay content below 15%. Due to the critical importance of the $SAR_e$ for soil structural stability, no scenario can be considered agro-environmentally sustainable if the simulated $SAR_e$ exceeds the ANZECC guidelines threshold value of 20.

As for the $SAR_e$, the electrical conductivity ($EC_e$) of the soil saturation extract is commonly used as an indicator of soil salinity in irrigation studies (Ezlit, Smith and Raine, 2010). Moreover, both indicators were also adopted in environmental assessments addressing the impacts of PW on soil, plants and groundwater (Biggs et al., 2013; Newell and Connor, 2006). The relative crop yield was estimated through its expected response to the $EC_e$ considering the FAO salt tolerance parameters given by Shaw et al (2011). For sugar beet, the exemplar crop in this study, the threshold $EC_e$ for a maximum potential yield is 7 dS/m, from this value, the crop productivity decreases by 5.9% per dS/m increase of the $EC_e$. Therefore, considering a minimum acceptable yield corresponding to 50% of the crop yield potential, the resulting threshold $EC_e$ used in this study is 15.5 dS/m.

The quality of drainage water (DW) can affect groundwater. In fact, DW carries dissolved salts and depending on the aquifer depth and quality, it may increase groundwater salinity and sodicity (Shannon, Cervinka and Daniel, 1997). The volume and quality of the DW leaving the soil were simulated at the maximum soil depth (1 m). The DW quality parameters $EC_d$ and $SAR_d$ were compared to the average maximum EC (30.6 dS/m) and SAR (48) values of Qatar's northern shallow aquifer to estimate the risks of groundwater salinisation and sodification.

### 6.2.3 Quantification of the agro-environmental sustainability indicators

The agro-environmental sustainability indicators were calculated using the soil-water model SALTIRSOIL_M (Visconti, 2013). The modelling approach was chosen primarily for minimising the time to obtain results compared to field trials. Moreover, multiple 'what-if' scenarios can be tested with models without the need for a huge number of field experiments. Finally, extreme scenarios can be simulated without any negative environmental impact (Graves et al., 2002).

The SALTIRSOIL_M model is a deterministic, transient-state, unidimensional model with a monthly time step. It has been successfully used to calculate the long-term ionic composition and $EC_e$ of the soil saturation extract of an irrigated field in semi-arid south-eastern Spain (Visconti et al., 2014).

The soil depth selected for the simulation was 0–60 cm as this is the depth where sugar beet root density is maximal (Draycott, 2006). All results of soil composition were expressed for a saturated extract (i.e. the standard soil-water ratio for salinity measurements) (Rhoades, 1996) and at chemical equilibrium (i.e. the long-term state of the soil).

### 6.2.4 Irrigation scenarios and site characteristics

Irrigation was considered agro-environmentally sustainable only if the root zone $EC_e$ and $SAR_e$ remained below their critical threshold levels of 15.5 dS/m and 20 respectively. This can be achieved by leaching salt out of the root zone through over-irrigation and/or by reducing the salt input to the soil through diluting PW with treated sewage effluent (TSE) or reverse osmosis-treated produced water (ROPW).

1. Although groundwater is the main source of irrigation water in Qatar, this resource cannot be used for blending PW. Indeed, the authorities restrict groundwater abstraction for irrigation to preserve the aquifers and to use them as strategic reserves in case of severe water shortage (Mohieldeen and Al-Marri, 2016). On the other hand, the use of non-conventional water resources, such as TSE, is developing particularly for substituting groundwater in irrigation (Ali et al., 2016).
2. TSE can be used to dilute PW, thus improving irrigation water quality.
3. PW can be partially desalinated through reverse osmosis (RO) and the ROPW can be used to dilute raw PW and improve irrigation water quality. RO has been successfully used for adapting PW to irrigation (Brown et al., 2010) and remains the cheapest commercial technology for PW desalination (Jiménez et al., 2018).

In this paper, 39,999 simulations were performed to simulate irrigation with raw PW (1), PW blended with TSE (99 blends) and PW blended with ROPW (99 blends) with 201 irrigation amounts varying from 100–300% of the crop water needs for each water quality. The blend composition varied from 99% PW-1% TSE up to 1% PW-99% TSE and the same with ROPW from 99% PW-1% ROPW up to or 1% PW-99% ROPW (Figure 6-1). The long-term $EC_e$ and $SAR_e$ of the soil saturation extract and the $EC_d$ and $SAR_d$ of the DW resulting from these irrigation simulations were calculated using the SALTIRSOIL_M model.

### 6.2.5 Water quality

Three types of effluents were used to simulate irrigation: raw PW, PW blended with TSE (PW-TSE) and PW blended with ROPW (PW-ROPW). Irrigation waters of decreasing salinity were created by blending PW with TSE or with ROPW.

Data on PW quality were sourced from Al-Kaabi (2016), Janson et al (2015) and Ahan (2014). The quality parameters values of TSE from Doha municipal wastewater treatment plant were sourced from Ahmad (1989) except for nitrate content which was sourced from a similar type of effluent produced in Abu Dhabi, UAE (Dalahmeh and Baresel, 2014). The quality of ROPW was estimated according to the performance of a pilot treatment train which successfully treated oil-field PW generating 70% ROPW and 30% brine from the inflow PW (Ersahin et al., 2018) (Table 6-2).

### 6.2.6 Climate

Qatar's monthly climatic averages of temperature, relative humidity, precipitation, wind speed, downward solar radiation, and number of rainy days for the period 1975–1992, were obtained from the World Meteorological Organisation Standard Normals (UN Statistics Division, 2010). The number of sunshine hours was estimated using the adapted equation of Ångström-Prescott (Viswanadham and Ramanadham, 1969) and the reference evapotranspiration (ETo) estimated using the Penman-Monteith equation integrated into the CROPWAT 8.0 model (FAO, 2018b) (Table 6-1).

### 6.2.7 Soil

Soil parameters were sourced from the Harmonised World Soil Database (FAO, 2009b). The soil volumetric water contents at saturation and at field capacity were estimated from the soil texture and organic matter content (Saxton and Rawls, 2006). The soil organic matter content (SOM) was estimated from the total organic carbon content using the Van Bemmelen factor of 1.72 (Soil Survey Staff, 1996). The soil $CO_2$ partial pressure ($pCO_2$) was estimated from the soil pH (Thomas, 1996) (Table 6-3).

### 6.2.8 Crop growth and irrigation requirements

The planting date of sugar beet was set on the first of August, a typical planting date in Egypt, a major sugar beet producer in the Middle East which has a hyper-arid climate and sandy calcic soils as in Qatar (Tate and Hamza, 2017). The shaded area values of sugar beet were obtained from Webb et al (1997). Crop coefficients, growth stages length and root depth values were obtained from the FAO (2018) (Table 6-1).

The CROPWAT 8.0 model, a decision support system for the planning and the management of irrigation (FAO, 2018b) was used to estimate the crop water needs and the irrigation requirements in the conditions of Qatar.

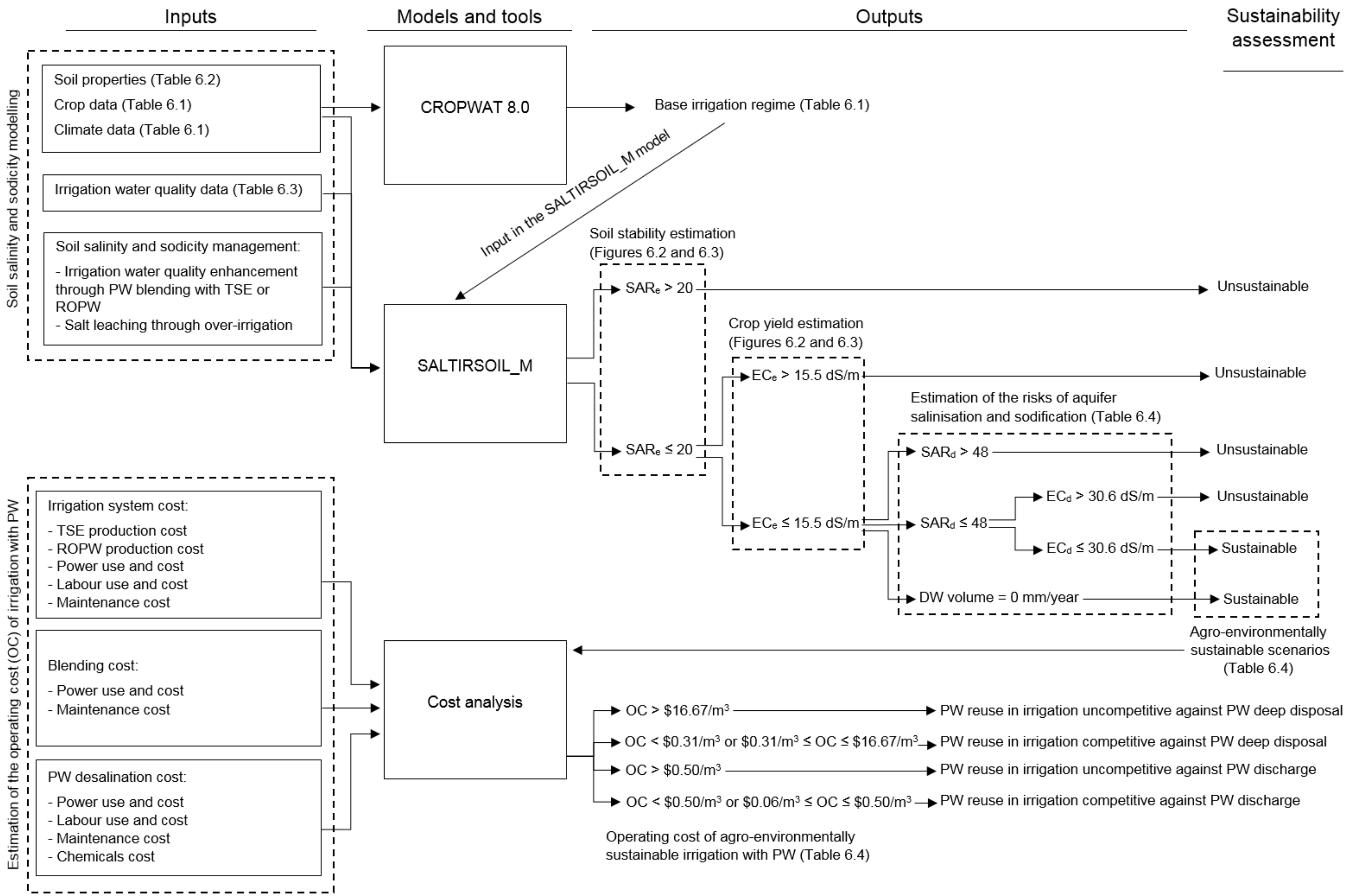



**Figure 6-1. Research methodology flowchart and decision tree for the sustainability assessment.**

### 6.2.9 Cost analysis

A cost analysis was developed to estimate the annualised operating costs of the identified agro-environmentally sustainable irrigation scenarios. The operating cost is defined as the cost of watering a hectare of sugar beet equipped with drip irrigation and calculated as the sum of the costs associated with PW blending, PW desalination, and with the irrigation system. The operating costs related to PW treatment (de-oiling) and to farming operations such as crop fertilisation, farm machinery, seasonal labour, pests and diseases control, etc., were not considered. Also, the capital cost related to investments in land and buildings as well as bank loans were not considered. These parameters would be dependent on the studied project size (e.g. infrastructure dimension) and local financial conditions (e.g. interest rates, subsidies, etc.) which are site-specific and could not be estimated.

The operating cost (OC) of irrigating a surface of one hectare with drip irrigation was estimated in Equation (6-1) as the sum of the irrigation system cost (IC), blending cost (BC) and PW desalination cost (DC), all terms are expressed in US$/ha/year:

$$OC = IC + BC + DC \qquad \textbf{(6-1)}$$

#### 6.2.9.1 Cost of the irrigation system

IC, in $US/ha/year, was estimated in Equation (6-2) as:

$$IC = WC + PC + MC + LC \qquad \textbf{(6-2)}$$

The water cost (WC), in US$/ha/year, was estimated in Equation (6-3) as:

$$WC = \sum_{i=1}^{k} (V_i \times C_i) \qquad \textbf{(6-3)}$$

where $V_i$ is the volume of PW, and/or TSE, and/or ROPW in $m^3$ and $C_i$ the production cost of PW, and/or TSE and/or ROPW in $/m^3$. The production cost of TSE for unrestricted irrigation was estimated at $0.45/m^3$ (Pistocchi et al., 2018), the production cost of ROPW was estimated at $0.89/m$^3$ (Ersahin et al., 2018) and de-oiled PW (i.e. raw PW) was assumed to be delivered at no cost.

PC is the power cost, in US$/ha/year, estimated in Equation (6-4) as:

$$PC = \frac{volume\ of\ water\ applied}{pump\ flow\ capacity} \times pump\ motor\ power\ \times electricity\ cost \quad \textbf{(6-4)}$$

PC is related to pumping irrigation water, with a pump of 48 m$^3$/h flow capacity powered by a 7.5 kW electric motor (Oosthuizen et al., 2007). The electricity cost without subsidies in Qatar was assumed to be $0.10/kWh (Krarti et al., 2017).

MC is the maintenance cost of the irrigation system, in $US/ha/year, estimated in Equation (6-5) as:

$$MC = \frac{annual\ maintenance\ cost\ of\ the\ irrigation\ system}{plot\ area} \quad \textbf{(6-5)}$$

The annual maintenance cost of a 25 ha plot equipped with drip irrigation, in $US/year, was obtained from Oosthuizen et al (2007). The currency used in this reference was the South African Rand, thus, this was converted to $US (January 2019 exchange rate) and inflation-adjusted (i.e. 2007 $1 = 2019 $1.24) using the US Bureau of Labour Statistics inflation calculator (US Bureau of Labour and Statistics, 2019).

LC is the labour cost, in $US/ha/year, estimated in Equation (6-6) as:

$$LC = \frac{annual\ amount\ of\ hours\ of\ labour\ required}{plot\ area} \times hourly\ mimimum\ wage \quad \textbf{(6-6)}$$

The annual amount of hours required (in hours/25 ha) was obtained from Oosthuizen et al (2007). The hourly minimum wage was estimated at US$1.98/hour for the generic profession ‘labour’ (Embassy of India in Qatar, 2014) and the maximum working hours of 47h per week allowed by the Qatari labour law (Qatar e-government, 2019b). The original labour cost in Qatari Rial was converted to $US using the January 2019 exchange rate.

#### 6.2.9.2 Cost of blending PW

Blending PW with TSE or with ROPW would require an infrastructure for mixing both effluents in the right proportion. It was assumed that PW and TSE or ROPW were pumped separately (using the same pump type as for the one used for irrigation) and mixed into a constructed reservoir. The blend was then pumped to irrigate the crop in the field plot.

BC, expressed in $US/ha/year, was estimated in Equation (6-7) as:

$$BC = PC + \frac{annual\ maintenance\ cost\ of\ the\ reservoir}{annual\ volume\ of\ water\ pumped\ into\ the\ reservoir} \tag{6-7}$$

The calculation of PC was described previously. The maintenance cost, in $US/m$^3$/year, was assumed for a lined reservoir of 75,000 m$^3$ of usable capacity suitable to irrigate an area of 30 ha (Weatherhead et al., 2014). The original cost in Pound Sterling was converted to $US according to the January 2019 exchange rate.

#### 6.2.9.3 Cost of PW desalination

DC, expressed in $US/ha/year, was estimated in Equation (6-8) as:

$$DC = WC_{ROPW} + PC + MC + LC + CC + other\ costs \tag{6-8}$$

where $WC_{ROPW}$ is the cost of the volume of ROPW applied in $US/ha/year and PC is the power cost, in $US/ha/year, estimated in Equation (6-9) as:

$$PC = Total\ power\ use\ of\ the\ desalination\ unit \times electricity\ cost \tag{6-9}$$

The estimations of the maintenance cost (MC), labour cost (LC), chemicals cost (CC) and other costs related to PW desalination, all expressed in $US/ha/year, were based on a pilot-scale treatment train (Ersahin et al., 2018).

## 6.3 Results

The impact of irrigation with PW on the long-term $EC_e$ and $SAR_e$ are presented in Figure 6-2 for the PW-TSE blends and in Figure 6-3 for the PW-ROPW blends. For clarity purpose, only some blends of interest are represented. The irrigation efficiency of the drip irrigation system was assumed to be 90% as per Brouwer, Prins and Heibloem (1989).

### 6.3.1 Irrigation with raw produced water

Irrigation with raw PW was unsustainable whatever the applied irrigation volume. In fact, Figure 6-2 shows that at a base irrigation regime (100% of the crop water needs), the use of raw PW led to a $SAR_e$ of 49, way above the ANZECC threshold level for maintaining the soil structural stability. Likewise, the $EC_e$ reached 45.8 dS/m which is much greater than 15.5 dS/m, the crop threshold value corresponding to 50% of the crop yield potential. Therefore, the soil structural stability and crop development cannot be preserved in these circumstances. The soil salinity and sodicity can be improved to a certain limit by increasing the irrigation amount. In fact, over-irrigation up to 300% of the crop water needs was effective to reduce the $SAR_e$ to 21 and the $EC_e$ down to 8.6 dS/m which would correspond to a yield of 90% of the crop yield potential. Despite that, irrigation with raw PW remained unsustainable as over-irrigation was unable to reduce the $SAR_e$ below the threshold level for soil structural stability conservation.

Consequently, no irrigation strategy could be found with raw PW without causing soil structural instability due to excessive $SAR_e$. As using raw PW cannot be considered, it was not necessary to further study its impact on groundwater and its cost of use in irrigation.

### 6.3.2 Irrigation with produced water blended with treated sewage effluent (PW-TSE)

#### 6.3.2.1 Impact on soil structural stability and on crop yield

There are multiple possibilities of irrigating sugar beet with PW-TSE while preserving the soil structural stability and a yield of at least 50% of the crop yield potential.

An extreme example is to use a low water quality combined with a high irrigation amount. Indeed, the minimum blending ratio and irrigation amount for preserving the soil structural stability and for having a yield of at least 50% of the crop yield potential was 96% PW-4% TSE with an irrigation amount of 272% of the crop water needs. In this scenario, the simulated $SAR_e$ and $EC_e$ reached 20 and 8.6 dS/m respectively (Figure 6-2).

The opposite extreme scenario is to use a higher water quality and a lower irrigation amount such as 26% PW-74% TSE with an irrigation amount covering 100% of the crop water needs. In this scenario, the simulated $SAR_e$ and $EC_e$ reached 13 and 12.9 dS/m respectively. Thus, the soil structural stability would be preserved, and the crop could yield at 65% of the crop yield potential (Figure 6-2).

#### 6.3.2.2 Impact on groundwater quality

Even if irrigation with PW-TSE could preserve the soil structural stability and a crop yield of at least 50% of the crop yield potential, it could represent a threat to groundwater quality. As an example, the irrigation scenario previously mentioned with 96% PW-4% TSE at 272% of the crop water needs, generated 1733 mm of annual drainage with an $EC_d$ of 43.1 dS/m, this is higher than the maximum aquifer EC value, and a $SAR_d$ of 45, which is below the maximum aquifer SAR value. Therefore, this irrigation scenario is unsustainable as DW would significantly increase groundwater EC.

Improving DW quality until it no longer constitutes a threat to groundwater was possible by increasing the dilution of PW and the irrigation amount. In fact, the minimum blending ratio for preserving soil fertility while preserving groundwater quality was 66% PW-34% TSE at 294% of the crop water needs. In this scenario, DW volume was higher (1988 mm/year), but its salinity and sodicity were both lower ($EC_d$ = 30.6 dS/m, $SAR_d$ = 27) compared to the previous scenario with 96% PW-4% TSE at 272% of the crop water needs (Table 6-4).

Alternatively, DW could be suppressed to avoid groundwater contamination. In fact, the excess irrigation water started to drain when the irrigation amount was greater than or equal to 109% of the crop water needs, the scenarios with an irrigation amount below 109% of the crop water needs which were sustainable from a soil point of view did not pose any risk to the aquifer neither. On the other hand, when the irrigation amount was greater than or equal to 109% of the crop water requirements, DW could potentially increase the groundwater EC and/or SAR even if the irrigation scenario was safe for the soil structural stability and for the crop yield. Thus, the groundwater could be preserved when the irrigation amount was minimised such as for the scenario using 26% PW-74% TSE with an irrigation amount covering 100% of the crop water needs (Table 6-4).

### 6.3.3 Irrigation with produced water blended with reverse osmosis-treated produced water (PW-ROPW)

#### 6.3.3.1 Impact on soil structural stability and on crop yield

When PW was blended with ROPW, the minimum PW dilution ratio for preserving the soil structural stability and a minimum yield of 50% of the crop yield potential was 89% PW-11% ROPW with an irrigation amount of 297% of the crop water needs. In this scenario, the $SAR_e$ reached 20 and the $EC_e$ was 8.3 dS/m enabling the crop to yield up to 90% of the crop yield potential (Figure 6-3).

On the other hand, a higher water quality and a lower irrigation amount could be used such as 15% PW-85% ROPW with an irrigation amount covering 100% of the crop water needs. In this scenario, the simulated $SAR_e$ and $EC_e$ reached 17 and 5.3 dS/m respectively. Thus, the soil structural stability would be preserved, and the crop could reach its full yield potential (Figure 6-3).

#### 6.3.3.2 Impact on groundwater quality

The same way as for the PW-TSE blends, a low PW dilution ratio had to be coupled to a high irrigation volume to maintain suitable $SAR_e$ and $EC_e$ values leading to high DW volumes. Although irrigating with 89% PW-11% ROPW at 297% of the crop water needs was sustainable from a soil point view, it generated 1999 mm of annual drainage with an $EC_d$ of 39.4 dS/m, which is higher than the maximum aquifer EC, and a $SAR_d$ of 45, which is below the maximum aquifer SAR. The minimum dilution ratio for preserving soil fertility and groundwater quality was 68% PW-32% ROPW at 290% of the crop water requirements. In this scenario, DW volume was higher (1924 mm/year) but its salinity and sodicity were both lower ($EC_d$ = 30.6 dS/m, $SAR_d$ = 40) compared to the previous scenario (Table 6-4).

Here again, a 'zero drainage' irrigation strategy with 15% PW-85% ROPW at 100% of the crop water needs was safe for the aquifer as the risk of altering groundwater quality was non-existent (Table 6-4).

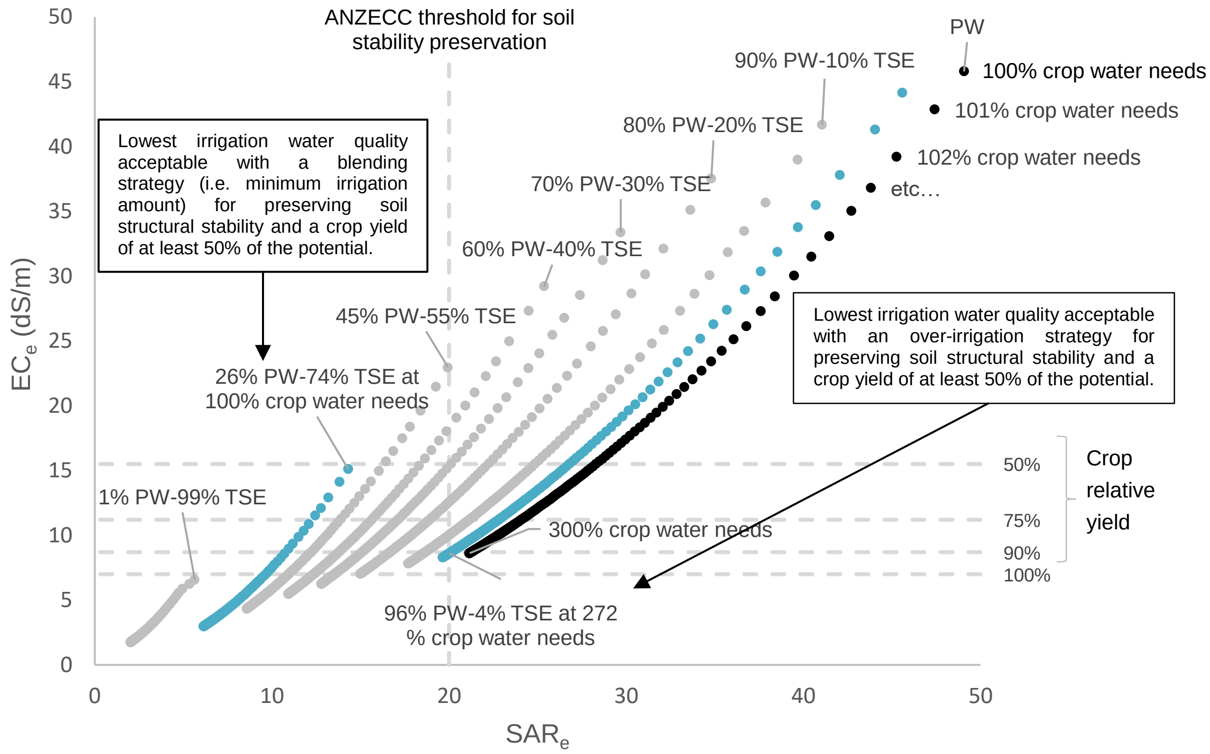



**Figure 6-2. Long-term $EC_e$ and $SAR_e$ following irrigation of sugar beet with different blends of PW diluted with TSE (from 100% PW down to 1% PW + 99% TSE) and with different irrigation amounts (from 100% up to 300% of the crop water needs).**

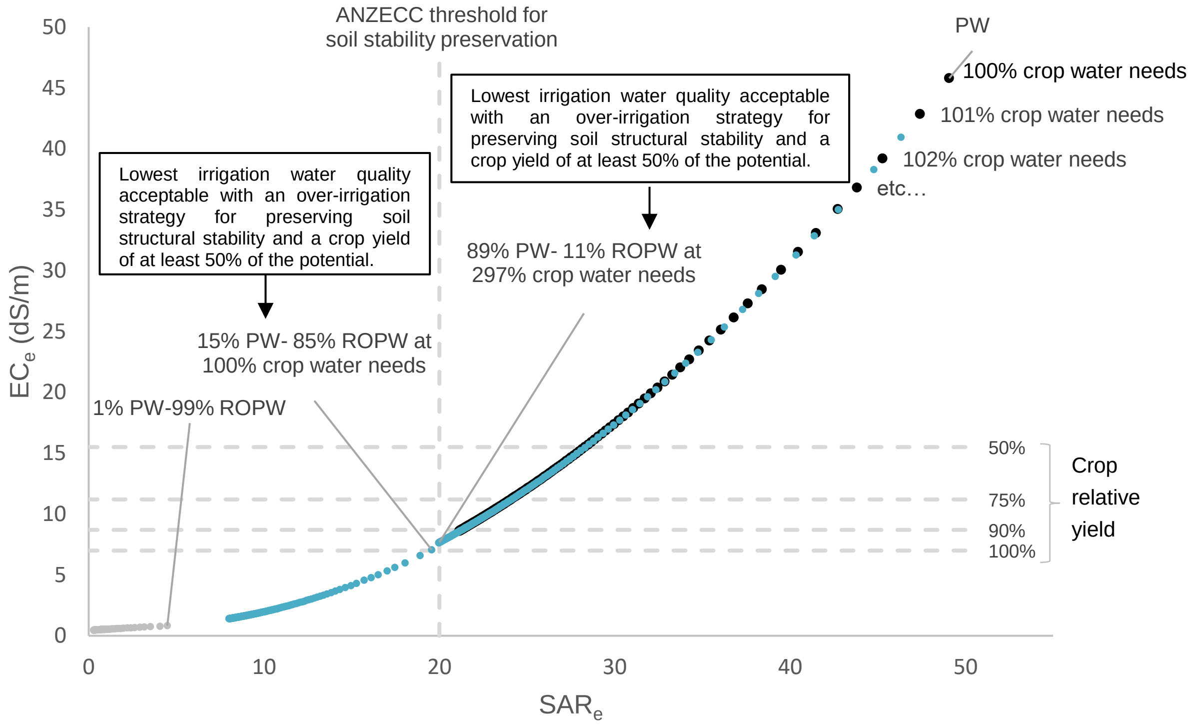



**Figure 6-3. Long-term $EC_e$ and $SAR_e$ following irrigation of sugar beet with different blends of PW diluted with ROPW (from 100% PW down to 1% PW + 99% ROPW) and with different irrigation amounts (from 100% up to 300% of the crop water needs).**

### 6.3.4 Operating cost of irrigation

The operating cost of irrigation was negatively correlated to the proportion of PW in the irrigation water and the irrigation amount. It also depended on the type of water used for blending PW (i.e. TSE or ROPW). This was explained by the fact that the operating cost is closely linked to the water and energy consumption of irrigation (Table 6-4).

The water consumption of irrigation depended on the volume of water applied but also on the volume of PW that had to be desalinated in the case of PW-ROPW blends. Indeed, desalinating PW led to a water loss (i.e. brine) representing 30% of the inflow PW volume. Thus, using ROPW to blend PW leads to a higher water consumption per hectare compared to using TSE to blend PW. Therefore, the higher the irrigation volume and the proportion of ROPW in the irrigation water, the higher the water consumption of irrigation.

The energy consumption was related to the pumping of water (from the gas field to the irrigated field and from the gas field to the constructed reservoir when PW was blended) and also to PW desalination. Thus, the water consumption and the energy consumption depended on the same parameters.

## 6.4 Discussion

### 6.4.1 Identified agro-environmentally sustainable irrigation scenarios

The potential agro-environmentally sustainable irrigation scenarios that have emerged from the simulations are summarised in Table 6-4. All these scenarios were preserving soil stability, enabling a minimum yield of 50% of the crop yield potential and were also preserving the aquifer from alteration by DW. These objectives were achieved in two ways; either through a combination of relatively low PW dilution along with a high irrigation amount or through a high dilution of PW along with a low irrigation amount.

Once the soil structural stability and a minimum yield of 50% of the crop yield potential were reached, groundwater preservation was the main factor limiting the irrigation water quality and the irrigation amount that could be used. Actually, DW minimisation is one way to prevent groundwater alteration, while the alternative was to increase the dilution of PW and the irrigation amount to decrease the $EC_d$ and the $SAR_d$ below the maximum aquifer EC and SAR values.

**Table 6-4. Selected agro-environmentally sustainable irrigation scenarios with PW blended with TSE (PW-TSE) and PW blended with ROPW (PW-ROPW), and their impacts on soil structural stability, crop yield, groundwater quality, water use (including losses through desalination brine), energy use and operating cost.**

| Scenarios | Irrigation water quality and amount | | | | | Impact on soil and crop | | | Impact of DW on Qatar's northern shallow aquifer | | | Water and power consumption | | Irrigation operating cost | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | PW (%) | TSE (%) | ROPW (%) | Volume (mm) | Crop needs (%) | $EC_e$ (dS/m) | $SAR_e$ | Crop yield (%) | $EC_d$ (dS/m) | $SAR_d$ | Volume (mm) | Water ($m^3$/ha) | Power (kWh/ha) | \$/ha | \$/$m^3$ |
| Lowest irrigation water quality acceptable | 66 | 34 | 0 | 3127 | 294 | 6.0 | 12 | 100 | 30.1 | 27 | 1967 | 31270 | 4886 | 5824 | 0.19 |
| | 68 | 0 | 32 | 3085 | 290 | 5.9 | 18 | 100 | 30.6 | 40 | 1924 | 33811 | 37888 | 18570 | 0.60 |
| Lowest water and energy use | 26 | 74 | 0 | 1064 | 100 | 12.9 | 13 | 65 | - | - | 0 | 10640 | 1662 | 3937 | 0.37 |
| | 15 | 0 | 85 | 1064 | 100 | 5.3 | 17 | 100 | - | - | 0 | 12523 | 22686 | 11548 | 1.09 |
| Least-cost scenarios | 50 | 50 | 0 | 1149 | 108 | 14.2 | 16 | 58 | - | - | 0 | 11490 | 1795 | 3006 | 0.26 |
| | 21 | 0 | 79 | 1106 | 104 | 6.0 | 18 | 100 | - | - | 0 | 14808 | 31005 | 5038 | 0.46 |

PW: produced water, TSE: treated sewage effluent, ROPW: reverse osmosis-treated produced water, $EC_e$: electrical conductivity of the soil saturation extract, $SAR_e$: sodium adsorption ratio of the soil saturation extract, DW: drainage water, $EC_d$: electrical conductivity of the drainage water, $SAR_d$: sodium adsorption ratio of the drainage water.

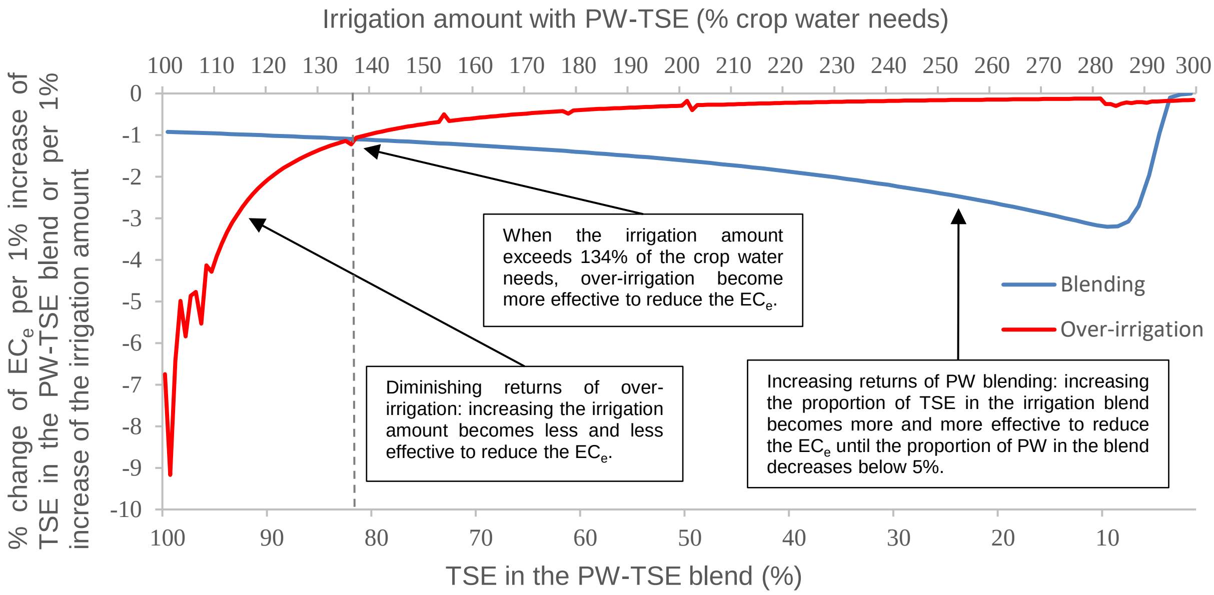
### 6.4.2 Understanding how agro-environmentally sustainable irrigation can be achieved

#### 6.4.2.1 Salt leaching through over-irrigation and salt dilution through produced water blending

Figure 6-4 shows how the marginal effect of over-irrigation and the marginal effect of PW blending on the $EC_e$ and $SAR_e$ differed in terms of dynamic and amplitude. Indeed, diminishing returns were observed regarding the marginal effect of over-irrigation on the reduction of the $EC_e$ and $SAR_e$. The average $EC_e$ decrease per percentage of increase of the irrigation amount (all PW-TSE blends considered) was higher than 4% for an irrigation amount up to 110% of the crop water needs. It then constantly decreased and was below 1% when the irrigation amount was higher than 142% of the crop water needs. The same was observed for the $SAR_e$, the average $SAR_e$ decrease per percentage of increase of the irrigation amount was superior to 2% for an irrigation amount up to 110% of the crop water needs. It continuously decreased and was below 0.5% when the irrigation amount was greater than 146% of the crop water requirements.

In contrast, increasing returns were observed regarding the marginal effect of PW blending to reduce the $EC_e$ and $SAR_e$. In fact, the average $EC_e$ decrease per increase of the TSE percentage in the PW-TSE blend was lower than 1% when the proportion of TSE in the blend was below 4%. It then increased and was over 2% when the percentage of TSE in the blend was between 63–95%. The $SAR_e$ reduction was quite steady, below 2% when the percentage of TSE in the blend was between 1–78%. It then drastically increased and was over 2% (up to 12%) when the proportion of TSE in the blend was between 79 to 99% (Figure 6-4).

These results show that the efficiency of over-irrigation in reducing the $EC_e$ and $SAR_e$ was very quickly limited. Actually, blending PW with TSE became more efficient than over-irrigation to reduce the $EC_e$ and $SAR_e$ when the irrigation amount was higher than 134% and 113% of the crop water needs respectively (dotted lines in Figure 6-4). This suggests that in the perspective of soil salinity management, it is more effective to leach excessive salt by over-irrigating first (until the irrigation amount reaches 134% of the crop water needs) before completing the soil salinity control strategy by diluting PW with TSE. However, if the soil sodicity is the main issue due to its negative impact on soil structural stability, over-irrigation should be at least practised until covering 113% of the crop water needs before considering to blend PW with TSE.



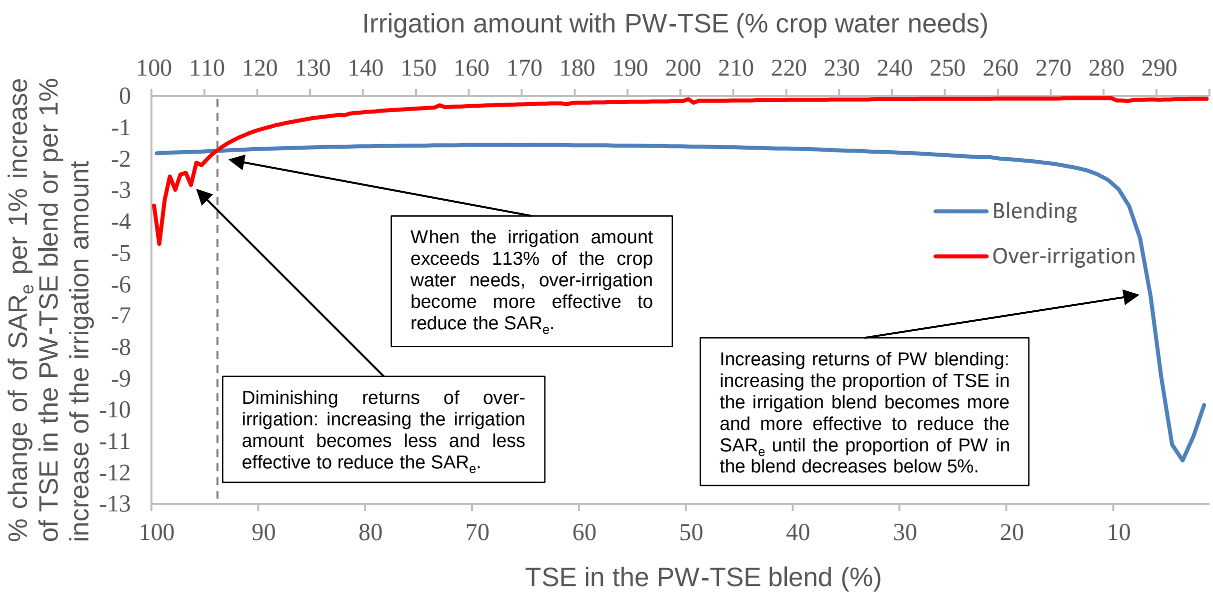




**Figure 6-4. The marginal effect of PW blending (upper horizontal axis) and over-irrigation (lower horizontal axis) on the average percentage reduction of $EC_e$ and $SAR_e$.**

### 6.4.2.2 Produced water blending with treated sewage effluent and with reverse osmosis-treated produced water

The type of effluent used to dilute PW influenced irrigation agro-environmental sustainability. In fact, blending PW with ROPW while irrigating at a base irrigation amount could have a similar impact on the $EC_e$ and $SAR_e$ than increasing the irrigation amount with raw PW (Figure 6-3). In contrast, blending PW with TSE at a base irrigation amount could result in similar $EC_e$ values but lower $SAR_e$ values compared to increasing the irrigation amount with raw PW (Figure 6-2). This is obviously explained by the lower salinity of ROPW compared to TSE (Table 6-1) which created blends of lower salinity ($EC_w$) compared to the PW-TSE blends. Nonetheless, ROPW has a higher $SAR_w$ compared to TSE as the latter has not been demineralised by the desalination process. Thus, at a comparable irrigation amount and PW dilution ratio, irrigation with PW-TSE was more sustainable (from a soil point of view) than irrigation with PW-TSE blends.

In practice, a remineralisation of ROPW could adjust the SAR of the irrigation water. The addition of gypsum or any other source of calcium and magnesium into the irrigation water would not be feasible for the blends with a high proportion of PW. Indeed, the solubility of gypsum is limited by the high total dissolved minerals in PW. Therefore, instead of amending water with gypsum, the application of gypsum to the soil could be efficient to reduce the $SAR_e$ and preserve the soil structural stability. However, as gypsum dissolves in the soil solution, it would increase the $EC_e$ and the crop osmotic stress and thus it would limit crop yield if the $EC_e$ exceeds the crop $EC_e$ threshold values after the addition of gypsum to the soil. Besides, as gypsum releases free $Ca^{2+}$ and $Mg^{2+}$ ions in the soil solution, it displaces $Na^+$ ions which are leached by DW (Ashworth, Keyes and Crépin, 1999). Thus, groundwater sodicity could be affected if the irrigation amount is high enough to generate DW.

### 6.4.3 Operating cost of agro-environmentally sustainable irrigation scenarios

#### 6.4.3.1 The drivers determining the operating cost of agro-environmentally sustainable irrigation

The water and energy consumptions of the irrigation system, the blending system, and of the RO unit were the main factors determining the operating cost of irrigation (Table 6-4).

The type of water used to blend PW largely influenced the operating costs of irrigation. In fact, using ROPW for blending PW was costlier compared to blending PW with TSE. Indeed, the production cost of ROPW (\$0.89/m$^3$) is about twice as much as the production cost of TSE (\$0.45/m$^3$). This difference of cost between both effluents is explained by the high costs of the inputs related to PW desalination (i.e. energy, chemicals, labour and maintenance costs of the RO unit). Moreover, in the least-cost scenario, the PW dilution ratio was higher and the irrigation amount was just slightly lower when ROPW was used rather than TSE for blending PW (i.e. 21% PW-79% ROPW at 1106 mm compared to 50% PW-50% TSE at 1149 mm). As a result, the volume of ROPW that had to be used per hectare (8737 m$^3$/ha) was significantly higher than the volume of TSE that had to be applied per hectare (5745 m$^3$/ha) for a comparable scenario objective (i.e. cost minimisation) (Table 6-4). The higher cost of blending PW with ROPW obviously discourages the use of this type of effluent to improve PW quality for irrigation.

Although the volume of water and the power consumed per hectare highly contributed to the operating cost of irrigation, the least-cost scenarios were not those which were consuming less water and power. In fact, the least-cost strategies were an equilibrium between using over-irrigation and PW blending. This could be explained by the marginal effect of these two techniques on the reduction of the $EC_e$ and of the $SAR_e$ as the most efficient way to reduce these agro-environmental sustainability indicators is to combine both over-irrigation (between 100 to 134% of the crop water needs) and PW blending. However, further investigation is needed to understand how the cost of over-irrigation and

PW blending influenced the equilibrium between these two techniques to obtain the least irrigation cost.

Lastly, it is interesting to notice that, in the perspective of preserving groundwater, avoiding generating DW through higher PW dilution rate was less costly than increasing the irrigation amount to improve DW quality. Indeed, the least-cost scenarios with TSE and ROPW were covering 108% and 104% of the crop water needs respectively (Table 6-4). These irrigation amounts were just below 109%, the amount of water from which excess irrigation water starts to drain.

#### 6.4.3.2 The cost of reusing produced water in irrigation compared to the cost of produced water disposal

Qatar has a favourable environment for developing the reuse of PW in irrigation including a hyper-arid climate, a pro-active wastewater reuse policy, a need for alternative irrigation water resources, and geographical proximity between the PW supply (i.e. North Field) and the farmlands (Shomar, Darwish and Rowell, 2014). Nonetheless, in order to be considered by O&G firms, the reuse of PW in irrigation must be competitive compared to current disposal practices (Table 6-5). Although the cost of PW disposal practices are site-specific, it was estimated that the cost of deep-well injection was between \$0.31–\$16.67/$m^3$ globally (Fakhru'l-Razi et al., 2009) and between \$1.57–\$15.72/$m^3$ in the USA, depending on PW quality and well ownership (Dolan, Cath and Hogue, 2018). If the deep disposal well is located at a long distance from the O&G field and if there is no pipeline to convey PW to the deep disposal well, PW needs to be hauled at a cost of \$0.20/$m^3$/km in the USA (Coday et al., 2015). The cost of surface discharge was estimated at \$0.06–\$0.50/$m^3$ globally (Fakhru'l-Razi et al., 2009) but this management is mainly for coastal locations with a discharge point into the sea. The estimated operating cost of irrigation in Qatar was between \$0.19–\$0.37/$m^3$ for PW-TSE blends and between \$0.46–\$1.09/$m^3$ if PW-ROPW was chosen. The operating cost of PW reuse in subsurface drip irrigation in the USA was estimated at \$0.98–\$1.48/$m^3$ while the capital cost was estimated at \$14,826/ha (Plappally and Lienhard, 2013). The total cost of other commercial-scale irrigation projects with PW in the USA was estimated at \$0.69–\$5.8/$m^3$ (Siagian et al., 2018).

**Table 6-5. Cost estimates of different PW management practices**

| PW management practice | Location | Cost | Reference |
|---|---|---|---|
| Surface discharge | Worldwide | $0.06–$0.50/m$^3$ | 1 |
| Shallow-well injection | Worldwide | $0.63–$8.37/m$^3$ | 1 |
| Deep-well injection | Worldwide | $0.31–$16.67/m$^3$ | 1 |
| | USA | $1.57–$15.72/m$^3$ | 2 |
| | Oman | $0.30/m$^3$ | 3 |
| Evaporation pond | Worldwide | $0.06–$5.03/m$^3$ | 1 |
| Freeze-thaw evaporation | Worldwide | $16.67–$31.45/m$^3$ | 1 |
| Electrodialysis | Worldwide | $0.13–$4.03/m$^3$ | 1 |
| Reuse in enhance oil recovery | Worldwide | $0.31–$7.86/m$^3$ | 1 |
| Reuse in irrigation | USA | $0.69–$5.80/m$^3$ | 4 |
| | Colorado (USA) | $1.76–$2.58/m$^3$ | 2 |
| | Colorado (USA) | $1.57–$6.29/m$^3$ | 5 |
| | Colorado (USA) | $2.36/m$^3$ | 6 |

[1](Fakhru'l-Razi et al., 2009), [2](Dolan, Cath and Hogue, 2018), [3](Hardisty, 2010) [4](Siagian et al., 2018), [5](Stewart and Takichi, 2005), [6](McGuire, 2007)

Although the total cost of the management of PW through irrigation in Qatar needs to be estimated, the estimated operating costs alone remain within the lower range of the cost of PW deep disposal (for PW-TSE and PW-ROPW blends) and within the cost range of PW surface discharge worldwide (for PW-

TSE blends only). This suggests that PW reuse in irrigation in Qatar is potentially competitive against traditional PW disposal practices.

### 6.4.4 Limitations

The simulations carried out and the cost analysis are exploratory, their limitations related to the model, the method, and the assumptions used in this study are acknowledged.

Although the SALTIRSOIL_M model has been calibrated and validated against field results in semi-arid environments with irrigation water of moderate salinity (Visconti et al., 2014), it has not yet been tested and validated in hyper-arid conditions with irrigation water of similar salinity as North Field PW. Therefore, the obtained results highlight possible agro-environmentally sustainable irrigation practices with PW in hyper-arid Qatar. These scenarios need then to be simulated and tested in field conditions including with soil amendments which were not considered in the simulations but their impact on soil salinity need to be evaluated.

In addition, the agro-environmental sustainability of irrigation with PW is principally, but not exclusively a salinity issue. The risks related to other constituents of concern present in PW, such as heavy metals (Al-Kaabi, 2016), production chemical compounds and radioelements (Alley et al., 2011) would need to be assessed. Notwithstanding, the high pH and low SOM content of soils in northern Qatar limit heavy metals bioavailability. On the contrary, the high $EC_e$ reached as a result of irrigation increases the risk of heavy metals absorption by plants (Singh et al., 2009). Although the risks associated with other components of PW are not as significant as those related to salts, they still deserve to be specifically addressed in a complete environmental impact assessment.

SALTIRSOIL_M is a soil-water model and does not apply to hydrogeological studies. Thus the evolution of groundwater quality as a result of irrigation is not precisely known. What can be suggested is that as Qatar’s northern shallow aquifer lies between 40 to 80 m deep (Shomar, 2015), DW would continue to migrate deeper and accumulate in successive geologic layers until field capacity is attained. Eventually, DW would reach the saturation zone corresponding to the aquifer. Although DW salinity is unlikely to significantly change after 1 m of depth as it is no longer affected by evaporation nor plant uptake, the volume of DW that would reach the aquifer and its impact on groundwater quality remains unknown and would need to be specifically quantified.

There are uncertainties regarding the estimated operating costs of PW reuse in irrigation. First, the cost of de-oiling PW was not considered due to lack of data in Qatar. Second, the cost of natural gas (the main fuel used for generating electricity in Qatar) fluctuates and would affect PW desalination cost (Darwish, Abdulrahim and Hassan, 2015). Third, the operating cost of RO-desalination has been decreasing and is as low as \$0.214/m$^3$ for recent large-scale plants treating seawater of 40 679 ppm of salinity (Bashitialshaaer, Persson and Aljaradin, 2011; Plappally and Lienhard, 2013). Assuming this lower production cost for ROPW, it would reduce the cost of the lowest cost irrigation scenario with PW-ROPW to \$4,306/ha, that is ~15% cost reduction compared to the simulated scenario. This cost reduction would improve the cost competitiveness of PW-ROPW blends compared to the use of PW-TSE. However, PW desalination facilities are smaller and do not benefit from scale economies compared to large seawater desalination facilities (Bernat et al., 2010). In fact, recent experiments have demonstrated that the total desalination cost of PW with a salinity of 50 000 ppm could be below \$1.5/m$^3$ (Osipi, Secchi and Borges, 2018). Despite that, PW desalination cost could be cheaper in Qatar thanks to the relatively low salinity of North Field PW (4502 ppm). A cost analysis based on numerical simulations estimated the total cost of desalinating water of 15 000 ppm of salinity to produce irrigation water (400 ppm of salinity) in a 24 000 m$^3$/day plant capacity at \$1.39/m$^3$ (Sarai Atab, Smallbone and Roskilly, 2016). Despite possible lower cost for producing ROPW, it is unlikely that it becomes more advantageous than TSE for blending PW as TSE does not need an energy-intensive desalination process to be produced.

The total cost of reusing PW in irrigation in Qatar still needs to be estimated by considering the exact cost of disposing and discharging PW locally. Finally, the income from crop production and the broader social and environmental benefits would need to be assessed through appropriate methods such as the cost-benefit analysis (CBA) to better estimate the overall sustainability of irrigation with PW in Qatar.

## 6.5 Conclusions

Being a water-scarce country, Qatar is developing unconventional water resources to irrigate its croplands and increase its food security. North Field PW is constantly generated independently of climatic conditions and could be an additional resource to develop. Reusing PW in irrigation could also benefit O&G firms as this practice is an alternative to conventional disposal techniques which are environmentally risky, increasingly regulated and costly. Unfortunately, PW is high in salt and sodium, thus its long-term use in irrigation can degrade soil fertility, crop productivity and contaminate groundwater. However, mitigation strategies such as over-irrigation and PW blending can be adopted to reduce these negative externalities.

Simulations results have shown that PW could be used in irrigation while preserving the soil and the aquifer from salinisation and sodification if PW was mixed with TSE in a 66:34 ratio and if the irrigation amount is at least equivalent to 294% of the crop water needs. However, the least cost option was to decrease the irrigation amount to 108% of the crop water requirements while decreasing the proportion of PW in the blend in a 50:50 ratio. Considering this irrigation scenario and the annual volume of PW generated by North Field, there would be enough PW to irrigate 121 ha of sugar beet with a yield of 58% of the crop yield potential. The crop value generated was estimated at ~$200,000/year, considering a potential yield of 50t/ha and an average crop value for the period 2000-2016 of $56/t (FAO, 2017).

The same way, blending 68% PW with 32% ROPW with an irrigation volume equivalent to 290% of the crop water needs was enough to preserve the soil, crop and groundwater. Still, the least-cost scenario with ROPW was to increase the dilution of PW (21% PW-79% ROPW) and to minimise the irrigation amount at

104% of the crop water requirements. In this case, there would be enough PW to irrigate 95 ha of sugar beet with the full crop yield potential.

In brief, the simulations and the cost analysis highlight that the quest for agro-environmentally sustainable irrigation implies trade-offs between the irrigation volume, the water quality and the crop yield potential.

The simulations put the stress on the fact that, although irrigation with blended PW can be sustainable from a soil-plant point of view, it could potentially affect groundwater even if the volume of DW that would reach the aquifer is uncertain. Thus, DW leaving the root zone must be properly managed to avoid transferring the salinity and sodicity hazards from the soil to the groundwater. This risk is relatively limited by the depth of Qatar's northern shallow aquifer and the absence or limited volume of DW when the excess irrigation amount is relatively low compared to the crop water needs (under 109% of the crop water needs). In case of a high risk of groundwater degradation, precautions such as DW capturing or eventually soilless agriculture could be imagined.

Further research is necessary to include other PW constituents such as heavy metals, organic compounds and radioelements in an environmental impact assessment. Additional work needs to be carried out to precisely quantify the total cost (including the capital cost) of reusing PW for irrigation in Qatar. Eventually, a CBA would be relevant to assess the social and economic sustainability of PW reuse in irrigation in Qatar where the economic (e.g. crop value) and the non-economic value generated (e.g. food and water security) would have to be considered.

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# 7 TOWARDS AGRO-ENVIRONMENTALLY SUSTAINABLE IRRIGATION WITH PRODUCED WATER IN HYPER-ARID ENVIRONMENT: THE CASE OF NIMR OIL FIELD IN OMAN[4]

## 7.1 Introduction

The extraction of oil and gas (O&G) is accompanied by massive volumes of 'produced water' (PW), which is composed of formation water initially present in the hydrocarbon reservoir and also, water that is injected during O&G operations and comes back to the surface such as water injected for enhanced oil recovery and for hydraulic fracturing (Engle, Cozzarelli and Smith, 2014). In volume, PW is the main by-product associated with the O&G industry (Veil, 2011) and its production is expected to increase dramatically by the next decade (Dal Ferro and Smith, 2007).

In Oman for instance, the volume of PW was estimated at 330 000 $m^3$/day in 1997, 650 000 $m^3$/day in 2005 (Al-Muscati, Huijskes and Parker, 1997), 910 000 $m^3$/day in 2018 and forecasts predict more than 1 million $m^3$/day in 2019 (Prabhu, 2018). Generally, PW management becomes increasingly costly due to the drastic growth of PW volume worldwide (Du, Guan and Liang, 2005). PW management cost is even more critical in mature production zones such as in Nimr oil field located in the southwest Omani desert where the water-to-oil ratio is as high as 10:1 (Stefanakis, Prigent and Breuer, 2018). In fact, as the water-to-oil ratio increases, the profitability of operating an O&G field could be compromised as it partly depends on the cost of PW management compared to the revenue obtained from O&G extraction.

In Nimr, as in most onshore O&G fields, the PW that is not reused for enhanced oil recovery is usually injected into deep disposal wells (Global Water Intelligence, 2014). This technique is the most common PW disposal practice in Oman (Van Den Hoek et al., 2000). However, this method is energy-intensive, carbon-

[4] This chapter is in the form of a paper to be submitted to Agricultural Water Management.

intensive and expensive (Al-Rawahi et al., 2014). Moreover, PW injection into deep disposal wells contains environmental risks such as groundwater contamination (Hagström et al., 2016) and induced seismicity (Walsh and Zoback, 2015). To prevent aquifer pollution and damages caused by induced seismicity to infrastructures and buildings, increasingly stringent regulations requiring extensive PW treatment prior deep-well injection have been developed in several parts of the world (Al-Sofi, 2014; Folger and Tiemann, 2016). For the same reason, since 2005, PW disposal into shallow geological formations is not permitted anymore in Oman. The Omani government also targets a gradual reduction of the PW volume managed through injection into deep geological formations (Stefanakis, Prigent and Breuer, 2018).

Reusing PW in irrigation comes as an alternative to deep-well injection to reduce the environmental and financial costs of PW management. In the meantime, the beneficial reuse of PW in agricultural irrigation is an opportunity to transform a waste stream into a valuable resource. This concept is particularly relevant in water-scarce regions where agricultural development is limited by water availability. Indeed, PW reuse in irrigation, is in alignment with Oman’s ambitious water management and agricultural development policies, aiming to increase crop production and food security through the reuse of marginal water resources (Jaffar Abdul Khaliq et al., 2017; McDonnell, 2016). In theory, the contribution of PW to irrigation in Oman could be significant. In fact, the 365 million $m^3$/year of PW represent the equivalent of 31% of the annual water volume withdrawn by the agricultural sector which is the largest water consumer in the Omani economy (FAO, 2009b).

Since 2015, PW has started to be reused in Nimr oil field as part of irrigation trials. On this site, the world’s largest constructed wetland treats 115 000 $m^3$/day of PW to reduce its oil content below 0.5 mg/L. A part of the de-oiled PW is reused to irrigate 22 ha of crops in a biosaline agriculture research project. This project aims to demonstrate the feasibility of achieving agro-environmentally sustainable irrigation with PW in hyper-arid environments and encourage similar initiatives to reuse larger volumes of PW in Oman and in drylands worldwide.

However, there are challenges related to the agro-environmental sustainability, water consumption and financial viability of PW reuse in irrigation. Firstly, PW quality constitutes the main barrier for its unrestrictive use in agriculture. Indeed, PW salinity, sodicity and heavy metals contents often exceed the values recommended in the FAO guidelines for irrigation water quality (Alley et al., 2011).

In fact, an irrigation experiment conducted in Oman showed that the electrical conductivity ($EC_e$) and the sodium adsorption ratio ($SAR_e$) of the soil saturation extract rose from 1.6 to 7.1 dS/m for the $EC_e$ and from 2.3 to 68.1 for the $SAR_e$ after 102 days of irrigation with de-oiled PW. Consequently, the soil saturated hydraulic conductivity declined from $1.42\times10^{-3}$ to $1.6\times10^{-6}$ m/s (Hirayama et al., 2002). These soil degradations after using PW in irrigation are not unique to Oman but have been observed in many dry areas. In fact, PW salinity and sodicity can be responsible for the decrease of the soil structural stability and crop productivity (Echchelh, Hess and Sakrabani, 2018).

To counter these negative impacts, techniques aiming at mitigating soil salinity such as over-irrigation to increase salt leaching (Norvell et al., 2009), PW blending (Atia, 2017; Martel-Valles, Benavides-Mendoza and Valdez-Aguilar, 2017; Mullins and Hajek, 1998; Sintim et al., 2017) and PW desalination (Sousa et al., 2017; Weber et al., 2017) to reduce salt inputs to the soil, as well as soil gypsum amendments to mitigate soil sodicity (Bennett et al., 2016; Johnston, Vance and Ganjegunte, 2008) have been tested in field experiments mostly in dry areas. Nonetheless, these techniques are costly and have specific drawbacks. Over-irrigation leads to water losses through drainage whereas PW desalination induces the production of brine which must be disposed of. Blending PW with another source of water depends on the availability and quality of other water resources. In many drylands such as in Nimr, surface water is non-existent and groundwater is usually fossil, deep and brackish. Therefore, there is no renewable water resource of suitable quality on-site that can be used to dilute PW. Finally, soil gypsum amendments reduce the soil SAR but also increase the soil EC due to the dissociation of gypsum into $Ca^{2+}$ and $SO_4^{2-}$ ions in the soil solution. Therefore, gypsum preserves soil structural stability but using a large amount of it negatively affects crop productivity (Hillel, 2000).

Furthermore, if irrigation with PW accompanied by soil salinity and sodicity mitigation techniques can be agro-environmentally sustainable, its financial cost is not precisely known. Indeed, there are few data about the economic and financial feasibility of reusing PW in irrigation (Plappally and Lienhard, 2013). Although a techno-economic analysis estimated the cost of reusing raw PW to irrigate crops in Colorado (USA), it did not include any technique to improve the quality of the PWs that were too saline-sodic to be used solely in irrigation (Dolan, Cath and Hogue, 2018). On the contrary, Meng et al (2016) estimated the cost of treating PW up to potable level using desalination in California (USA), but crops do not require such high water quality grade and costly treatments to be watered.

In this context, there is a need to quantify the impacts of irrigation with PW on soil salinity and sodicity to identify potential agro-environmentally sustainable irrigation strategies in Nimr. Indeed, even if halotolerant crops such as jojoba, eucalyptus, castor bean, seashore paspalum and buttonwood have been grown for three years, it is still unknown if these performances can be maintained in the long-term without being compromised by excessive soil salinity and sodicity. Also, if over-irrigation or PW quality improvement (i.e. PW blending and desalination) are necessary to preserve the soil from salinisation and sodification in the long-term, the costs of agro-environmentally sustainable irrigation scenarios need to be estimated to support stakeholders' decision in upscaling the irrigation project in Nimr. This study aims to address these knowledge gaps by identifying agro-environmentally sustainable irrigation strategies on jojoba with oil-field PW enabling soil structural stability preservation, a maximal crop yield potential, and a maximal irrigated area. The environmental performance and the operating cost of PW management through irrigation and through deep-well disposal are compared and discussed.

## 7.2 Methods

The integrated approach of this paper consists of modelling the impacts of irrigation with PW on soil salinity and sodicity using a soil-water model and estimating the operating costs of potentially agro-environmentally sustainable irrigation scenarios using a cost analysis (Figure 7-1).

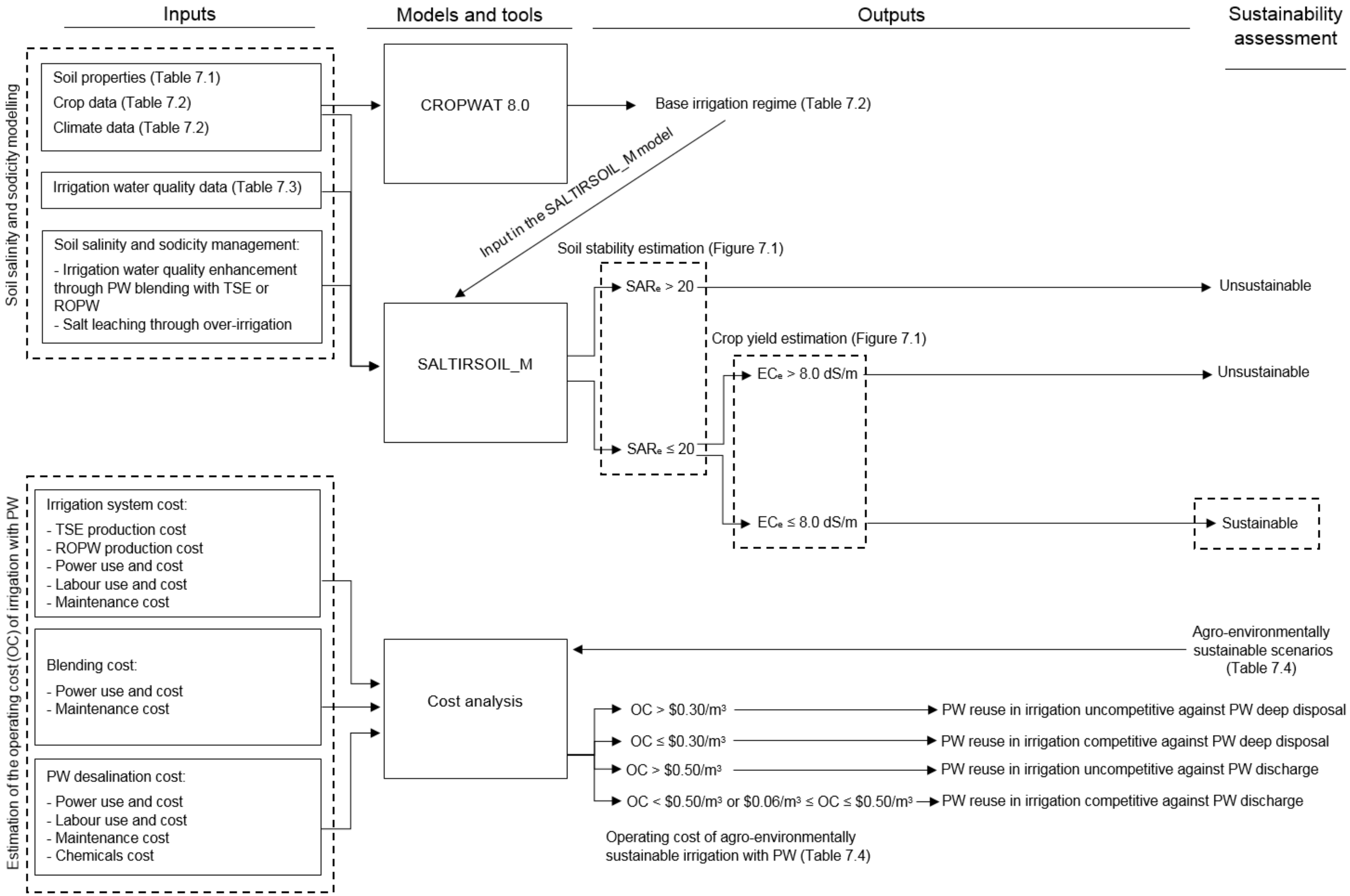


**Figure 7-1. Research methodology flowchart and decision tree for the sustainability assessment.**

### 7.2.1 Definition of agro-environmental sustainability

Sustainability is commonly defined as fulfilling current human needs without jeopardising the ability of future generations to meet their own needs (Held, 2001). In this paper, the agro-environmental sustainability is considered as conserving the soil capital to ensure the possibility for future generations to use the soil in Nimr to grow crops. Therefore, irrigation agro-environmental sustainability comprises the preservation of the soil structural stability and the maximum crop yield potential by maintaining safe soil salinity and sodicity levels. A large-scale irrigation project would include drainage pipes to capture and dispose of drainage water in the evaporation ponds that are already used for evaporating the excess PW volume which is not reused for irrigation. Thus, the impact of drainage water on the aquifer is not considered here as the aquifer would not be at risk of contamination.

### 7.2.2 Agro-environmental sustainability indicators and agro-environmental sustainability assessment

The risk of soil structure destabilisation was estimated by comparing the long-term $SAR_e$ estimated with the soil-water model to the threshold $SAR_e$ values obtained from the Australian and New Zealand Environment Conservation Council (ANZECC, 2000). The ANZECC guidelines inform about the risk of soil structural instability depending on the soil texture. These guidelines have been used as a reference to study the risks and feasibility of irrigating with PW under dry climates in Australia and in sub-Saharan Africa (Horner, Castle and Rodgers, 2011; Mallants, Šimůnek and Torkzaban, 2017; Shaw et al., 2011). The threshold $SAR_e$ was set at 20 as the soil is a sandy clay loam with a clay content below 24%. Due to the critical importance of $SAR_e$ for soil stability, no scenario can be considered agro-environmentally sustainable if the simulated soil $SAR_e$ exceeds the ANZECC guidelines threshold value of 20.

The relative crop yield was estimated through its expected response to $EC_e$. Jojoba has an estimated threshold $EC_e$ of 8 dS/m, but the slope of the yield decrease when the soil $EC_e$ exceeds this threshold remains unknown (Biosalinity Awareness Project, 2019).

Both the $SAR_e$ and $EC_e$ are commonly used as indicators of soil salinisation and sodification in irrigation studies (Ezlit, Smith and Raine, 2010). Moreover, these indicators were also adopted in environmental assessments addressing the impacts of PW on soil, plants and groundwater (Biggs et al., 2013; Newell and Connor, 2006).

### 7.2.3 Quantification of the agro-environmental sustainability indicators

The agro-environmental sustainability indicators were estimated using a soil-water model. The adoption of modelling in this study was justified because it minimises the time to obtain results compared to field trials. Indeed, the current field experiments in Nimr will take several years before providing results that could be interpreted in a sustainability assessment. This modelling exercise is an anticipation of the long-term agro-environmental sustainability level that could be expected from the field trials. Thus, it supports stakeholder's decision for future irrigation projects with PW by estimating the agro-environmental sustainability of the current practices that are being tested in the biosaline agriculture research project. Another advantage of modelling is that various 'what-if' scenarios can be tested whereas an unmanageable amount of trials would be necessary in an experimental approach. Finally, unlike field experiments, extreme scenarios can be simulated with a model without any negative environmental impact (Graves et al., 2002).

The agro-environmental sustainability indicators were calculated using the soil-water model SALTIRSOIL_M (Visconti, 2013). The SALTIRSOIL_M model is a deterministic, transient-state, unidimensional model with a monthly time step. It has been successfully used to calculate the long-term ionic composition and $EC_e$ of the soil saturation extract of an irrigated field in semi-arid south-eastern Spain (Visconti et al., 2014).

The soil depth selected for the simulation was 0–50 cm as this is the average soil depth in the biosaline agriculture research field plot. All results of soil composition were expressed for a saturated extract which is the standard soil-water ratio for salinity measurements (Rhoades, 1996) and at chemical equilibrium which is the state that would be reached in the long-term under constant irrigation water composition, irrigation management, climate features, soil physical properties, and crop cultivation.

#### 7.2.3.1 Irrigation scenarios and site characteristics

Irrigation agro-environmental sustainability can be achieved only if the root zone $EC_e$ and $SAR_e$ remain below threshold levels. This can be done by leaching salts out of the root zone through over-irrigation and/or by reducing salt inputs to the soil through the dilution of PW. Groundwater cannot be used for improving PW quality as the aquifer is very saline (EC = 39.1 dS/m). Alternatively, PW can be desalinated using reverse osmosis (RO), and RO-treated PW (ROPW) can be mixed with PW to improve irrigation water quality. RO has been successfully used for adapting PW to irrigation (Brown et al., 2010; Ersahin et al., 2018) and remains the cheapest commercial technology for PW desalination (Jiménez et al., 2018).

In the current paper, the SALTIRSOIL_M model simulated the long-term $EC_e$ and $SAR_e$ resulting from increasing irrigation amount (from 100% to 409% of the crop water needs) of raw PW and of 99 different blends of PW-ROPW. The blends composition varied from 99% PW-1% ROPW up to 1% PW-99% ROPW.

#### 7.2.3.2 Crop choice

Several halotolerant crops are currently grown in the biosaline agriculture research project in Nimr such as trees (jojoba, eucalyptus and acacia), shrubs (buttonwood, castor bean, glasswort and seashore paspalum) as well as a fibre crop (cotton). No food crops are grown at the moment due to public concerns regarding food safety. However, halotolerant food crops (e.g. barley, sorghum, oat, soya bean, date palm, Natal plum, sugar beet, courgettes and kale) could be of interest as they would increase Oman’s food self-sufficiency. In this study, Jojoba (*Simmondsia chinensis*) was selected as it is a salt-tolerant crop with low water requirements which can be irrigated with drip irrigation systems. This crop has shown promising growth results in the field trials and adapted well to the shallow desert soils and harsh climatic conditions in Nimr. Indeed, jojoba thrives in arid areas and on poor soils, it is also used to combat desertification and soil degradation in drylands. Jojoba oil has a myriad of applications in human consumption (e.g. food, pharmaceuticals and cosmetics) and also in industrial uses such as lubricants and biofuels (Al-Obaidi et al., 2017).

#### 7.2.3.3 Water quality

Table 7-1 presents the annual average quality of the two effluents (PW and ROPW) used in the irrigation simulations. These data were obtained from the on-site laboratory.

Prior to its use in irrigation, PW is de-oiled (oil in water < 0.5 mg/L) by the artificial wetland. PW is naturally poor in nutrients and thus, its contents in total nitrogen (~2.5 mg/L), phosphorus (~0.3 mg/L) and nitrate (~0.1 mg/L) are low. However, the salt concentration is not affected by the artificial wetland treatment process, thus PW remains highly saline (EC = 13.9 dS/m) and sodic (SAR = 65).

ROPW is not produced under normal operating conditions but it has been generated in Nimr oil field for testing the possibility of desalinating PW for irrigation purpose.

#### 7.2.3.4 Climate

The climate of the site is hyper-arid with no precipitation and 2790 mm average annual evapotranspiration (Table 7-2). Monthly averages of temperature, relative humidity, precipitation, number of days with precipitation, wind speed and downward solar radiation for the period 2013-2017 were obtained from an on-site Davis Vantage Pro 2 meteorological station. The reference evapotranspiration (ETo) was estimated using the Penman-Monteith equation integrated in SALTIRSOIL_M and the number of sunshine hours was estimated using the adapted equation of Ångström-Prescott (Viswanadham and Ramanadham, 1969).

#### 7.2.3.5 Soil

The soil is a shallow Gypsisol-Calcisol (Table 7-1) typical of this desert region (FAO, 2009a). The bedrock lies at 50–80 cm below the soil surface. The site is isolated and the soil has a poor fertility, thus the area has never been cultivated before the beginning of the PW reuse project.

A Dutch auger and a bulk density cylinder were used to collect 30 soil samples representative of two depth ranges 0-25 and 25-50 cm. Soil water retention properties, bulk density, texture (USDA), calcium carbonate equivalent and soil organic matter were determined according to standard methods (ISO 10693, 1995; ISO 10694, 1995; ISO 11272, 1998; ISO 11274, 1998; ISO 11277, 1998). The soil gypsum content was determined according to Soil Survey Staff (2014). A saturated paste was prepared for each sample, its $pH_e$ was measured, the saturation percentage was calculated, and the saturation extracts were obtained and analysed for $EC_e$ at 25°C, all according to Rhoades (1996). The alkalinity was determined by automatic titration (Eaton et al., 1998) and ionic contents ($Na^+$, $K^+$, $Mg^{2+}$, $Ca^{2+}$, $Cl^-$, $SO_4^{2-}$ and $NO_3^-$) by ion chromatography (Appendix 1). The ion contents and the $pH_e$ of the soil pastes were used to calculate the equilibrium $CO_2$ partial pressure for each soil layer using the ion speciation software SALSOLCHEMIS (Visconti, 2009).

#### 7.2.3.6 Crop growth and irrigation requirements

The crop coefficients values of jojoba were obtained from the Mallee Catchment Management Authority (2017), the shaded area was estimated on-site by measuring the surface area shaded by a mature jojoba tree. The irrigation requirements were provided by the company managing PW and estimated at 110 mm (Table 7-2). Although jojoba can grow with less than 120 mm of water, decent production of jojoba oil is unlikely below 250 mm of water. Jojoba water consumption can even exceed 450 mm (Ash, Albiston and Cother, 2005). Therefore, it was decided to vary the irrigation amount from 110 mm (100% of the crop water needs) up to 450 mm (409% of the estimated crop water needs).

**Table 7-1. Soil properties of the studied plot**

| Soil type (FAO's RSG) | Soil layer (cm) | Hydrophysical | | | USDA texture | | | Chemical | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | $\rho_b$ (g/cm$^3$) | $\theta_{fc}$ (%) | $\theta_{pwp}$ (%) | Sand (%) | Silt (%) | Clay (%) | pH | Gypsum (%) | CCE (%) | SOM (%) | log $pCO_2$ |
| Gypsisol-Calcisol | Topsoil 0–25 | 1.81 | 23.5 | 13.9 | 62 | 26 | 11 | 8.0 | 12 | 68 | 1.3 | -3 |
| | Subsoil 25–50 | 1.93 | 22.4 | 14.6 | 63 | 26 | 10 | 8.1 | 8 | 69 | 1.1 | -3 |

FAO's RSG: FAO's Reference Soil Groups, $\rho_b$: bulk density; $\theta_{fc}$: soil volumetric water content at field capacity; $\theta_{pwp}$: soil volumetric water content at permanent wilting point; CCE: calcium carbonate equivalent, SOM: soil organic matter, log $pCO_2$: log value of the $CO_2$ partial pressure.

**Table 7-2. Climatic, crop development and water quality data used in the simulations**

| | Parameter | January | February | March | April | May | June | July | August | September | October | November | December | Total |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Nimr on-site meteorological station | P (mm) | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| | ETo (mm) | 134 | 162 | 230 | 264 | 301 | 294 | 333 | 325 | 284 | 185 | 145 | 132 | 2790 |
| Jojoba | I (mm) | 6 | 6 | 8 | 9 | 11 | 12 | 12 | 12 | 11 | 9 | 8 | 6 | 110 |
| | $K_{cb}$ | 0.50 | 0.50 | 0.50 | 0.50 | 0.50 | 0.50 | 0.50 | 0.50 | 0.50 | 0.50 | 0.50 | 0.50 | 6 |
| | Root depth (cm) | 50 | 50 | 50 | 50 | 50 | 50 | 50 | 50 | 50 | 50 | 50 | 50 | - |

P: precipitation; ETo: reference evapotranspiration; I: base irrigation regime covering 100% of the crop water needs; $K_{cb}$: basal crop coefficient.

**Table 7-3. Quality of the different waters used for irrigation simulations (all ions contents are expressed in mmol/L or $mmol_c$/L of [$CaCO_3$] equivalent for the alkalinity, and the electrical conductivity in dS/m).**

| | [$Na^+$] | [$K^+$] | [$Ca^{2+}$] | [$Mg^{2+}$] | [$Cl^-$] | [$NO_3^-$] | [$SO_4^{2-}$] | $Alk_w$ | $EC_w$ | $SAR_w$ | $pH_w$ |
|---|---|---|---|---|---|---|---|---|---|---|---|
| PW | 123.591 | 1.082 | 2.409 | 1.209 | 124.524 | 0.002 | 5.782 | 4.753 | 13.89 | 65 | 8.0 |
| ROPW | 0.322 | 0.006 | 0.010 | 0.001 | 0.290 | 0.008 | 0.052 | 0.075 | 0.04 | 3 | 8.7 |

PW: produced water, ROPW: reverse osmosis-treated produced water, $EC_w$: water electrical conductivity, $SAR_w$: sodium adsorption ratio of the water, $Alk_w$: water alkalinity as [$CaCO_3$] equivalent.

### 7.2.4 Cost analysis

A cost analysis was used to estimate the annual operating costs of the identified agro-environmentally sustainable irrigation scenarios. The operating cost is defined as the cost of watering a hectare of jojoba equipped with drip irrigation and calculated as the sum of the costs of associated to PW desalination, PW blending with ROPW and to the irrigation system.

The cost of pumping the drainage water and RO-brine to the evaporation ponds as well as the cost of managing the crystallised salt (i.e. the end product of brine evaporation) were not considered. Also, the cost of de-oiling PW was not included in the cost analysis because PW needs to be de-oiled whether it is reused in irrigation or disposed of. Thus, it is not an additional cost when PW is reused in irrigation.

The capital and investment costs related to the irrigation and drainage system, water conveyance, water blending, PW desalination were not included in the cost analysis. These parameters would be dependent on the studied project size (e.g. infrastructure dimension) and local financial conditions (e.g. interest rates, subsidies, etc.) which could not be estimated.

#### 7.2.4.1 Cost of PW desalination

The desalination cost of PW (DC) was estimated at \$0.89/m$^3$ calculated as per Equation (7-1) and expressed in \$US/ha/year:

$$DC = WC_{ROPW} + PC + MC + LC + CC + other\ costs \tag{7-1}$$

where $WC_{ROPW}$ is the cost of the volume of ROPW applied in \$US/ha/year and PC is the power cost, in \$US/ha/year, estimated in Equation (7-2):

$$PC = Total\ power\ use\ of\ the\ desalination\ unit\ \times electricity\ cost \tag{7-2}$$

The estimations of the maintenance cost (MC), labour cost (LC), chemicals cost (CC) and other costs related to PW desalination, all expressed in \$US/ha/year, were based on a pilot-scale treatment train (Ersahin et al., 2018).

The volume of brine generated by RO was estimated at 30% of the inflow PW (Ersahin et al., 2018).

#### 7.2.4.2 Cost of blending PW

Blending PW with ROPW would necessitate a constructed reservoir for mixing both effluents in the right proportion. It was assumed that PW and ROPW were pumped separately (using the same pump type as for the one used for irrigation) and mixed into a constructed reservoir. The PW-ROPW blend was then pumped to irrigate jojoba.

The blending cost (BC) was estimated at \$0.10/m$^3$ and calculated as per Equation (7-3) and expressed in \$US/ha/year:

$$BC = PC + \frac{annual\ maintenance\ cost\ of\ the\ reservoir}{annual\ volume\ of\ water\ pumped\ into\ the\ reservoir} \qquad \textbf{(7-3)}$$

The calculation of PC was described previously. The maintenance cost, in \$US/m$^3$/year, was assumed for a lined reservoir of 75,000 m$^3$ of usable capacity suitable to irrigate an area of 30 ha (Weatherhead et al., 2014). The original cost in Pound Sterling was converted to \$US according to the January 2019 exchange rate.

#### 7.2.4.3 Cost of the irrigation system

The irrigation system cost (IC), in \$US/ha/year, was estimated in Equation (7-4):

$$IC = WC + PC + MC + LC \qquad \textbf{(7-4)}$$

As only ROPW was given a cost while PW was assumed to be delivered free of charge, the water cost (WC), in \$US/ha/year, was calculated as per Equation (7-5):

$$WC = DC \times Volume\ of\ ROPW\ applied \qquad \textbf{(7-5)}$$

PC is the power cost, in \$US/ha/year, estimated in Equation (7-6):

$$PC = \frac{volume\ of\ water\ applied}{pump\ flow\ capacity} \times pump\ motor\ power \times electricity\ cost \qquad \textbf{(7-6)}$$

PC is related to pumping irrigation water, with a pump of 48 $m^3$/h flow capacity powered by a 7.5 kW electric motor (Oosthuizen et al., 2007). The electricity cost in Oman was assumed to be $0.08/kWh (data provided by the company managing the site).

MC is the maintenance cost of the irrigation system, in $US/ha/year, estimated in Equation (7-7):

$$MC = \frac{annual\ maintenance\ cost\ of\ the\ irrigation\ system}{plot\ area} \tag{7-7}$$

The annual maintenance cost of a 25 ha plot equipped with drip irrigation was obtained from Oosthuizen et al (2007). The currency used in this reference was the South African Rand, thus, this was converted to $US (January 2019 exchange rate) and inflation-adjusted (i.e. 2007 $1 = 2019 $1.24) using the US Bureau of Labour Statistics inflation calculator (US Bureau of Labour and Statistics, 2019).

LC is the labour cost, in $US/ha/year, estimated in Equation (7-8):

$$LC = \frac{annual\ amount\ of\ hours\ of\ labour\ required}{plot\ area} \times hourly\ mimimum\ wage \tag{7-8}$$

The annual amount of hours required was obtained from Oosthuizen et al (2007). The hourly minimum wage was estimated at $15.58/hour for a monthly labour cost of $1039/month and a maximum working hours of 40h per week (data provided by the company managing the site). The original labour cost in Qatari Rial was converted to $US using the January 2019 exchange rate.

## 7.3 Results and discussion

### 7.3.1 Impact of irrigation on soil structural stability and on crop yield

The results of the simulations suggested that the irrigation of jojoba with raw PW would be unsustainable, and this, whatever the irrigation amount applied. Indeed, when 110 mm of PW was applied (100% of the crop water needs), the $SAR_e$ reached 87 which is far from the ANZECC threshold value of 20 for maintaining the soil structural stability. The $EC_e$ reached 28.5 dS/m which is also much higher than 8 dS/m, the threshold $EC_e$ value of jojoba. Increasing the irrigation amount up to 450 mm (409% of the crop water needs) enabled to partially leach excessive salt out of the root zone. Indeed, the $SAR_e$ and the $EC_e$ were reduced to 28 and 10.8 dS/m respectively (Figure 7-2). Therefore, the use of raw PW in Nimr is unlikely to be agro-environmentally sustainable due to the risk of damaging the soil structural stability and the crop yield in the long term.

On the other hand, it was possible to attain irrigation agro-environmental sustainability by mixing PW with ROPW. In fact, using irrigation water composed of 65% PW-35% ROPW with an irrigation amount of 425 mm (386% of the crop water needs) resulted in a long-term $SAR_e$ and $EC_e$ of 19 and 8.0 dS/m respectively. Approximately the same soil $SAR_e$ and $EC_e$ could be obtained with 110 mm of irrigation amount (100% of the crop water needs) but with a lower PW content in the irrigation water, that is 19% PW-81% ROPW (Figure 7-2). Both scenarios would be agro-environmentally sustainable by preserving the soil structural stability and the maximal crop yield potential.

Conserving irrigation water by minimising the irrigation amount required high irrigation water quality (i.e. a low PW content in the blend). In fact, when 110 mm of irrigation was applied to the soil to cover the crop water needs, almost all the water was used by the crop or evaporated (101 mm out of 110) leaving only 9 mm of drainage water. Thus, when irrigation waters with a PW content higher than 19% were used with the minimum irrigation amount, the soil solution became very concentrated as the amount of drainage water was not high enough to reduce the $SAR_e$ and the $EC_e$ below their respective threshold values for preserving the soil structural stability and the maximal crop yield potential. Consequently, for an irrigation strategy aiming to use the maximum proportion of PW in the irrigation water, such as 65% PW-35% ROPW, a higher irrigation amount (425 mm) had to be applied to leach excessive salt out of the root zone. However, this strategy reduced the water efficiency of irrigation as the volume of water that was either used by the crop or evaporated remained at 101 mm while the amount of water drained reached 324 mm. This lower water efficiency eventually limited the potential area that could be irrigated compared to a water conservation strategy (Table 7-4).

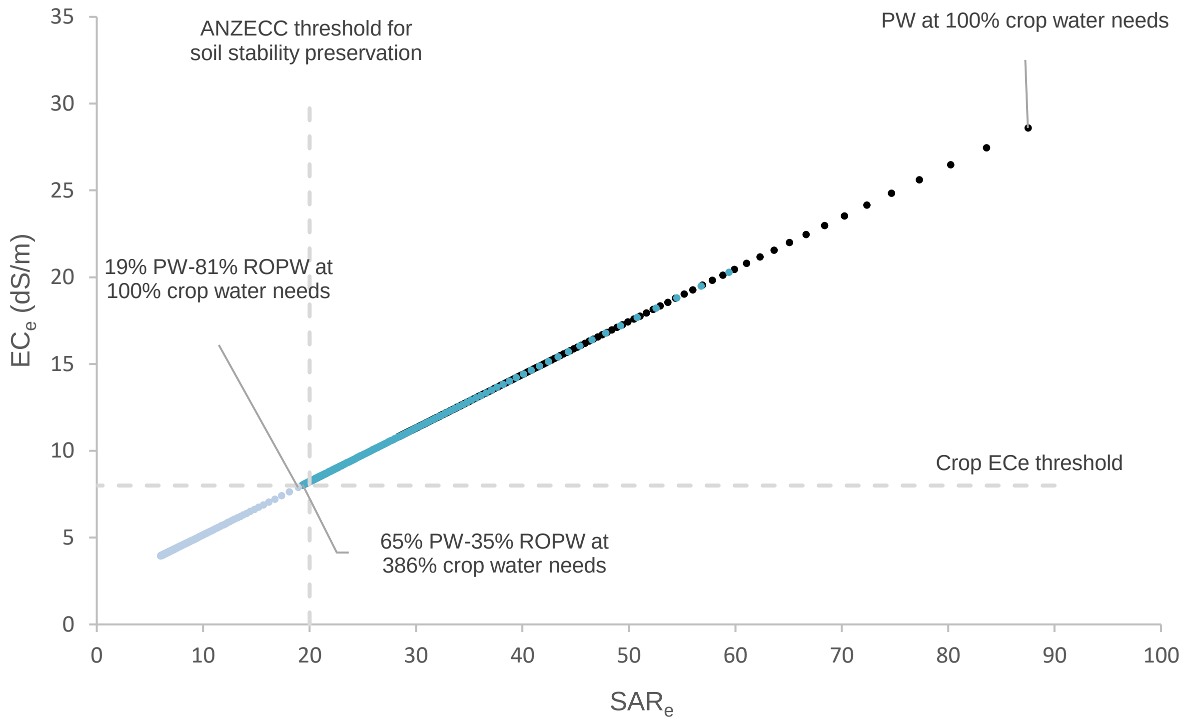


**Figure 7-2. Long-term $EC_e$ and $SAR_e$ following irrigation of jojoba with different blends of PW diluted with ROPW (from 100% PW down to 1% PW + 99% ROPW) and with different irrigation amounts (from 100% up to 300% of the crop water needs).**

### 7.3.2 The environmental performance of produced water reuse

Although both irrigation strategies achieved agro-environmentally sustainable $SAR_e$ and $EC_e$ values, they differ in terms of water use. Indeed, a water conservation approach with 110 mm of 19% PW-81% ROPW minimised the volume of water lost through drainage to 91 $m^3$/ha/year but it also generated 382 $m^3$/ha/year of RO-brine that would be lost in the evaporation ponds. In addition to have a lower irrigation efficiency, using 65% PW-35% ROPW with an irrigation amount of 449 mm/year generated more RO-brine (664 $m^3$/ha/year) compared to irrigation with 19% PW-81% ROPW. Actually, even if less desalinated PW had to be added in the 65% PW-35% ROPW blend than in the 19% PW-81% ROPW blend, more irrigation water had to be applied, therefore using 65% PW-35% ROPW led to a larger volume of ROPW used per hectare than using 19% PW-81% ROPW (Table 7-4).

Maintaining agro-environmentally sustainable $SAR_e$ and $EC_e$ levels involve a trade-off between wasting water through drainage for salt leaching and wasting water through RO-brine (generated by the desalination process) to reduce the irrigation water salinity and sodicity. In this study, targeting agro-environmentally sustainable $SAR_e$ and $EC_e$ values while minimising the water losses could be achieved by irrigating jojoba with 110 mm/year with a blend composed of 19% PW-81% ROPW (Table 7-4).

Reusing PW under the irrigation scenario 19% PW-81% ROPW with 110 mm/year of irrigation amount required about 0.2 kWh/m$^3$ of energy. This includes mixing PW and ROPW, pumping the irrigation water and the desalination of a part of the PW volume. The pre-treatment of the PW in the artificial wetland consumes less than 0.1 kWh/m$^3$. Most of this energy is not directly related to the treatment process but to the instrumentation, offices and accommodation facilities (Stefanakis, Prigent and Breuer, 2018). Thus, reusing PW in irrigation in Nimr would consume about 0.3 kWh/m$^3$ of energy (Table 7-4). In comparison, injecting PW into deep disposal wells requires between 3.6–5.5 kWh/m$^3$ of energy (Breuer and Al-Asmi, 2010). This comparison between the energy use of PW reuse compared to disposing of PW is corroborated by a detailed energy footprint assessment carried out in New Mexico (USA) which demonstrated that reusing PW was far more energy-efficient than transporting and disposing of PW into deep disposal wells (Zemlick et al., 2018). In Nimr, PW deep-injection is relatively cheap because the deep disposal wells are located nearby the oil field, whereas in several American O&G fields, PW needs to be hauled (sometimes over long distances) to the deep disposal wells, making PW disposal costly. For now, the reuse of PW in irrigation does not benefit from an environment which is as favourable as in the USA, but this situation could change due to the stricter regulation in Oman pushing towards the reduction of PW volumes to be disposed of.

In areas where there are no evaporation ponds to manage highly saline effluents (i.e. drainage water and RO-brine), these streams could still be managed through deep-well injection. Indeed, the reuse of PW in irrigation would at least reduce the volume of effluents that need to be injected, save energy, and reduce the cost of PW disposal compared to a situation where all PW is injected into deep disposal wells.

### 7.3.3 Cost of produced water reuse in irrigation

The cost of reusing PW in irrigation is very dependent on the cost of pumping water and desalinating PW. As these two factors are themselves dependant on the energy cost and on the irrigation amount, the least-cost irrigation scenario was the one minimising the energy use and the irrigation amount. For this reason, the less costly irrigation strategy estimated at \$821/ha was to use 19% PW-81% ROPW with 110 mm of irrigation amount (Table 7-4).

The operating cost of using PW in irrigation was estimated at \$0.75/m$^3$ (Table 7-4). In comparison, the cost of managing PW through deep-well injection in Nimr oil field was estimated at \$0.30/m$^3$ in 2010 (Hardisty, 2010). The main operating cost of deep-well injection is related to pumping and was estimated at \$0.11/m$^3$ in 2004 (Schrevel, Hellegers and Soppe, 2004). Jojoba starts to produce a significant volume of oilseeds four years after planting. Thus, based on world averages, the potential value of the crop generated was estimated at \$1,500/ha/year four years after planting and up to \$6,250/ha/year eight years after planting (Khan, Agarwal and Sharma, 2017).

Despite an estimated potential revenue greater than the operating cost, the larger economic and social benefits the financial justification of reusing PW in irrigation in Oman remains controversial. Indeed, Hardisty (2010), estimated that the agricultural revenues are not sufficient to cover the PW treatment cost, farm operating cost and project decommissioning cost. The decommissioning cost consist of excavating the salt-contaminated soil. Thus, this significant cost estimated between \$10,000–\$18,000/ha, would not be necessary if the irrigation scheme includes salinity and sodicity management strategies. On the other hand, Kojima et al (2015) estimated that using treated PW to irrigate tomatoes could be profitable in Oman. Indeed, with operating costs including PW treatment (i.e. de-oiling, desalination and boron removal) estimated at \$0.41/m$^3$ and a power cost related to the irrigation system at \$0.02/m$^3$, a water use of 6300 m$^3$/ha, a crop yield of 11,010 t/ha and a tomato value of $314.43/t, the theoretical profit of using PW to irrigate tomatoes was estimated at $1.48 per $m^3$ of PW used.

The financial cost of PW management is not the only criteria for selecting a PW management practice. In fact, the local O&G firm is determined to cut the amount of PW injected into deep disposal wells from 52% presently to 22% by 2025 (Prabhu, 2018). Therefore, the reuse of PW in irrigation should be compared to the alternatives proposed to reduce deep-well injection. Indeed, the range of PW reuse options differ in terms of environmental and social impacts, economic cost and benefits, PW treatment standards and potential volume that can be reused (Table 7-5).

**Table 7-4. Environmental and financial performance of selected agro-environmentally sustainable irrigation strategies**

| Scenarios | Irrigation water volume, quality, and water losses | | | | | | Impact on soil | | | Water and power consumption | | Irrigation operating cost | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | PW (%) | ROPW (%) | Crop needs (%) | Irrigation ($m^3$/ha) | Brine ($m^3$/ha) | Drainage ($m^3$/ha) | Potential area (ha) | $EC_e$ (dS/m) | $SAR_e$ | Total PW use ($m^3$/ha) | Power (kWh/ha) | $/ha | $/$m^3$ |
| A | 65 | 35 | 386 | 4246 | 637 | 3237 | 8596 | 8.0 | 19 | 4883 | 1327 | 1355 | 0.32 |
| B | 19 | 81 | 100 | 1100 | 382 | 91 | 28326 | 7.9 | 19 | 1482 | 344 | 821 | 0.75 |

A: scenario with the lowest irrigation water quality acceptable, B: scenario with the lowest irrigation amount, water losses, cost and largest potential irrigated area, PW: produced water, ROPW: reverse osmosis-treated produced water, $EC_e$: electrical conductivity of the soil saturation extract, $SAR_e$: sodium adsorption ratio of the soil saturation extract, RO: reverse osmosis.

**Table 7-5. Advantages and disadvantages of several beneficial reuses of PW compared to the reuse of PW in irrigation**

| Economic sector | Uses of PW | Advantages | Disadvantages | Reference |
|---|---|---|---|---|
| Agriculture and aquaculture | Irrigation | O&G fields are often surrounded by large farmland areas. | Risks of soil and aquifer contamination;<br>Seasonal variability in irrigation water demand;<br>Costly irrigation management and PW treatment is often needed;<br>Social acceptability remains challenging for food crops. | [1, 2, 3, 4] |
| | Livestock watering | O&G fields are sometimes surrounded by large dairy farms and feedlots.<br>No direct impacts on soil, crop and aquifer. | The water quality must be relatively high compared to irrigation standards to avoid livestock exposure to toxic contaminant levels (TDS < 10,000 mg/L). | [1, 2, 3, 4] |
| | Aquaculture | Some fish species can tolerate high water salinity (equivalent to seawater), thus the management of PW salinity is likely to be cheaper compared to PW reuse in irrigation. | Although salt-tolerant, fish are sensitive to a myriad of contaminants (organics, heavy metals, acidic or alkaline inputs, etc.);<br>Risk of food chain contamination (e.g. heavy metals) especially for fatty fish species. | [3] |
| Environmental restoration | Aquifer recharge | Restore aquifer for multiple groundwater users (agriculture, industry and services). PW of adequate quality can be injected into few wells reducing water conveyance cost. | Risk of aquifer contamination if PW is high in contaminants (i.e. dissolved minerals and organic pollutant). | [1] |
| | Stream flow augmentation | Can be a source of indirect PW reuse (e.g. irrigators pumping water into rivers);<br>Prevent low-flow surface streams from drying out, thus, maintaining ecosystems;<br>Limit water conveyance cost. | Risk of surface water contamination (water biological and chemical oxygen demand as well as salinity are critical);<br>Elevated flows accelerate erosion. | [1] |
| | Rangeland restoration | O&G fields are sometimes surrounded by extensive rangelands; | Risks of soil and aquifer contamination;<br>Lower crop value generated per $m^3$ of PW used compared to food crop irrigation. | [1] |

| | | | | |
|---|---|---|---|---|
| | | Restore rangelands damaged by over-grazing and drought;<br>Better social acceptability compared to food crop irrigation. | | |
| | Impoundment into natural or artificial wetland | Support biodiversity and prevent desertification;<br>Better social acceptability compared to food crop irrigation through the creation of leisure areas. | Risk of surface water, soil, aquifer and wildlife contamination (e.g. acute and chronic sodium bicarbonate toxicity to aquatic species). | 3 |
| Energy and industry | Reuse in O&G operations (enhanced oil recovery, hydraulic fracturing, well drilling, etc.) | PW is reused onsite limiting water conveyance cost;<br>Social acceptability is not a critical issue. | O&G operations may not be able to reuse the whole volume of PW generated (i.e. well saturation);<br>Hydraulic fracturing requires low-salinity PW to increase O&G reservoir permeability;<br>Risk of aquifer contamination when PW is reused in hydraulic fracturing. | 1, 3, 4 |
| | Power generation (steam) and cooling | Reduce freshwater abstraction for power plants and cooling units located in inland areas;<br>Social acceptability is not a critical issue. | PW must be of suitable quality to avoid equipment scaling. | 1, 3, 4, 7 |
| Mining | Metal recovery | Valuable metals (e.g. copper and lithium) can be recovered while treating PW whereas these are lost and can contaminated soils, plants and aquifers when PW is used in irrigation. | After metal recovery, PW still has to be managed somehow;<br>Metal recovery from PW remains costly;<br>Although PW contents in heavy metals get reduced, there are other contaminants of concern remaining in PW (e.g. salts). | 5 |
| | Dust control | Low quality PW can be used to control dust in coal mining.<br>Better social acceptability compared to food crop irrigation. | High PW conveyance cost unless PW is generated near the mine where PW is reused. | 1, 4 |
| Construction and infrastructure maintenance | Drilling | Reduce freshwater abstraction;<br>Better social acceptability compared to food crop irrigation. | Risk of contaminating soil layers and the aquifers crossed by the drill;<br>The volume that can be reused in drilling is limited compared to irrigation. | 1, 3 |

| | | | | |
|---|---|---|---|---|
| | Snow control and de-icing | Reduce grit salt consumption;<br>Better social acceptability compared to food crop irrigation. | Risks of soil and aquifer contamination;<br>Snow control and de-icing are seasonal activities;<br>High water hauling cost;<br>The volume that can be reused in snow and ice control is limited compared to irrigation. | 1, 3 |
| | Dust control | Suppress dust on unpaved roads used by heavy vehicles in dust-prone arid areas;<br>Better social acceptability compared to food crop irrigation. | Risks of soil and aquifer contamination, the volume that can be reused in dust control is limited compared to irrigation. | 1 |
| Safety | Fire control | Avoids the use of freshwater for firefighting,<br>The potential environmental degradation caused by PW quality is considered minimal compared to the damages caused by a wildfire.<br>Better social acceptability compared to food crop irrigation. | Risks of soil and aquifer contamination, large PW storage reservoirs have to be built. | 1 |
| Domestic | Potable water supply | Reduces freshwater abstraction;<br>Avoids environmental risks for the soil, plant and aquifer;<br>Potable water has a higher value compared to irrigation water. | The treatment of PW up to potable quality grade is costly compared to irrigation quality grade;<br>The social acceptability is likely to be even more challenging than for reusing PW to irrigate food crops. | 1, 5, 6 |

[1](Guerra, Dahm and Dundorf, 2011), [2](Horner, Castle and Rodgers, 2011), [3](Pichtel, 2016), [4](Nghiem et al., 2011), [5](Xu, Drewes and Heil, 2008), [6](Meng, Chen and Sanders, 2016), [7](Muraleedaaran et al., 2009)

### 7.3.4 Limitations

The simulations performed aim at anticipating the impacts of potential irrigation scenarios on soil salinity and sodicity. As the designers of a future irrigation scheme in Nimr might consider the results of these simulations, it is important to underline the limitations related to the model and to the method.

Firstly, the SALTIRSOIL_M model has been calibrated and validated against field results in semi-arid Spain with irrigation water of moderate salinity (Visconti et al., 2014) but it has not yet been tested and validated in hyper-arid conditions such as in Oman with water of equivalent quality as Nimr oil-field-PW. Therefore, the next step would be to continue the monitoring of the site by analysing soil and water samples on a regular basis to adjust the model assumptions. Also, the potential agro-environmentally sustainable scenario (19% PW-81% ROPW with 110 mm/year of irrigation amount) needs to be tested in field conditions and the measured $SAR_e$ and $EC_e$ values compared to the simulated $SAR_e$ and $EC_e$ values. In the long term, the measured values should tend to the values estimated with the soil-water model.

The addition of soil amendments was not considered in the simulations and their impact on soil salinity and sodicity need to be evaluated. The contribution of gypsum amendments to the soil $EC_e$ would need to be especially appraised. Indeed, gypsum has been added to the soil before planting jojoba to reduce soil sodicity. However, as gypsum dissolves into the soil solution it increases the $EC_e$. Thus, any gypsum amendment needs to be considered in irrigation modelling by monitoring the gypsum content of the soil and updating the gypsum content inputted in the simulations.

The environmental sustainability of irrigation with PW is principally, but not exclusively a salinity issue. The risks related to other constituents of concern present in PW, such as heavy metals (Al-Kaabi, 2016), organic compounds and radioelements (Alley et al., 2011) would need to be assessed. Notwithstanding, the soil in Nimr has high pH and low SOM content which limit heavy metals bioavailability. On the contrary, the high soil $EC_e$ reached due to irrigation increases the risk of heavy metals absorption by plants (Singh et al., 2009). Although the risks associated with other components of PW are not as significant as those related to salts, they still deserve to be specifically addressed in a global environmental impact assessment.

The total cost of reusing PW in irrigation in Nimr still needs to be estimated more precisely by considering agricultural inputs in the operating cost (e.g. fertilisers, labour, harvesting cost, etc.) and the capital cost related to PW treatment, irrigation water blending and investments related to the irrigation system and farm machinery. Finally, the income from crop production and the broader social and environmental benefits would need to be assessed through appropriate methods such as the cost-benefit analysis (CBA) to better estimate the overall sustainability of irrigation with PW in Nimr.

## 7.4 Conclusions

The increasing volume of PW is mostly managed through deep-well injection in Oman. This practice is environmentally controversial and more and more costly because of its energy consumption and the increasing regulatory pressure forcing the development of alternatives to current PW disposal practices. PW reuse in irrigation can reduce the negative impacts of deep-well disposal while providing a significant volume of water to irrigators. This concept is being tested in Nimr oil field in the Omani desert where a biosaline agriculture research project aims to select appropriate halotolerant crops and irrigation management for reusing large PW volumes. However PW quality is challenging and agro-environmentally sustainable irrigation can only be achieved by controlling soil salinity and sodicity. The SALTIRSOIL_M model was used to simulate the long-term impacts of irrigation with PW on soil $EC_e$ and $SAR_e$ and to test the ability of over-irrigation and PW blending to maintain these parameters to sustainable levels. The water efficiency and the operating costs of the different agro-environmentally sustainable irrigation strategies were also estimated.

Paradoxically, although water-efficiency is seen a priority in water-scarce drylands, preserving the soil from long-term salinisation and sodification impose high water losses. Indeed, increasing the irrigation amount to leach salt out of the root zone leads to a loss of water through drainage. On the other hand, improving the irrigation water quality by partially desalinating PW leads to a loss of water though RO-brine. Consequently, there are trade-offs between losing water and consuming energy by over-irrigating to leach the excessive salt load out of the root zone or by desalinating PW to reduce the salt input to the soil.

Amongst the identified agro-environmentally sustainable irrigation strategies, the scenario consisting of irrigating jojoba with 19% PW-81% ROPW at an irrigation amount corresponding to 100% of the crop water need (110 mm/year) provided the minimal operating cost, the lowest water use per hectare, and the largest potential irrigated area.

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# 8 SYNTHESIS

This thesis aimed at developing a framework to assess the agro-environmental sustainability of reusing O&G PW for agricultural irrigation in drylands. The specific objectives were to:

- assess the potential of reusing PW for agricultural irrigation in drylands by estimating the volume of PW generated in drylands, the volume of PW that can be potentially reused in irrigation, and by reviewing the experiences of irrigation with PW to identify the agro-environmental risks of this practice and the solutions to adapt PW to irrigation in drylands.
- estimate the long-term impacts of irrigation with PW on soil salinity and sodicity at the field level and its effect on soil structural stability, crop yield and groundwater quality in different climates and on different soils representative of drylands.
- identify agro-environmentally sustainable irrigation strategies with PW using techniques to prevent soil and aquifer degradation.
- estimate the costs of these agro-environmentally sustainable strategies in a regional and in an industrial context.

This chapter summarises the findings obtained in the preceding chapters, describes the research novelty and specifies the original contribution of this research. Finally, the practical implications and limitations of this research are discussed.

## 8.1 Research findings

### 8.1.1 Produced water volume, quality, and associated agro-environmental risks

The quantification of the PW volume generated worldwide and in several dry regions combined with the characterisation of the PW qualities originating from different dry areas and from different types of O&G productions provided information to estimate the potential of PW as a resource for irrigation in drylands.

In Chapter 3, the volumes of PW generated in major O&G-producing regions with dry climates were estimated based on a compilation of recent volume estimates and by calculating the volume of PW generated in some O&G basins using the total hydrocarbon production and the water-to-oil and water-to-gas ratios of these O&G basins. As more than half of the global PW volume is already beneficially reused mainly within the O&G industry for enhancing oil recovery, it was estimated that nearly 45% of the global PW volume is not beneficially reused and is thus potentially available for agricultural irrigation. This volume of PW which is currently considered as a waste and injected into deep disposal wells or discharged on the surface was estimated in 2010 at about 8.5 $km^3$ globally. If current PW management practices do not drastically change, it is estimated that between 13.1–24.3 $km^3$ of PW would be potentially available to be beneficially reused in irrigation in 2020. As a comparison, these volumes represent the equivalent of 28%, 44% and 81% respectively, of the volume of irrigation water used in the O&G-rich and water-scarce Arabian Peninsula in 2010 (Frenken and Gillet, 2012).

The experimental results that were reviewed in Chapter 2 have highlighted the prominence of soil salinisation and sodification as the main agro-environmental threats when PW is reused in irrigation. These soil degradations happen because most PWs are high in salt and sodium. Consequently, although there is 45% of global PW volume which is potentially available, most of PWs cannot be applied to the soil without jeopardising the agro-environmental sustainability of irrigation. Indeed, out of 474 PW samples from different types of O&G productions (i.e. conventional O&G, shale gas, tight oil, and coalbed methane) collected in the USA, Australia, South Africa and Qatar, only 8.4% were suitable for being reused in irrigation with no or limited restriction (Chapter 3). As most PWs do not meet the salinity and sodicity requirements of the FAO guidelines to be used raw in irrigation (Ayers and Westcot, 1985), it needs to be used in excess to leach excessive salt out of the root zone or PW needs to be blended or desalinated to improve irrigation water quality. Therefore, PW should not be seen as a primary water resource for irrigation but more as a secondary resource that supplement the existing irrigation water resources such as conventional (e.g. surface fresh water and groundwater) and unconventional water resources (e.g. TSE) in water-scarce areas.

### 8.1.2 Long-term agro-environmental impacts of irrigation with produced water in drylands

The main agro-environmental risks that were identified through field experiments in Chapter 2 were quantified using the SALTRISOIL_M model in Chapter 5 to assess the long-term impacts of irrigation with PW on the soil structural stability, crop yield and groundwater quality.

The simulations carried out in Chapter 5 brought further understanding of the salt dynamics (or the 'salt cycle') in the soil-water system when PW is used in irrigation. The PW has a role of 'salt supplier' by introducing a continuous flux of salt into the soil-water system. Thus, it greatly determines the long-term $EC_e$ and $SAR_e$ of the system. The soil 'regulates' the $EC_e$ through its water retention properties and buffers the $SAR_e$ through the dissolution of gypsum ($CaSO_4$ $2H_2O$ $\rightarrow Ca^{2+} + SO_4^{2-} + 2H_2O$) and calcite ($CaCO_3 + H_2O + CO_2 \rightarrow Ca^{2+} + 2HCO_3^-$). The soil has also a 'salt distribution' role, it retains a part of the salt load depending on the soil's water retention properties ($\theta_{(m)s}$, $\theta_{fc}$ and $\theta_{pwp}$). The water which cannot be retained by the soil is drained, exporting a part of the salt load from the root zone to the deeper soil layers and eventually to the groundwater. Finally, the crop and the climate aridity, further increase the long-term $EC_e$ and $SAR_e$ by water abstraction through evapotranspiration. These processes determine the long-term soil structural stability and crop yield which depend on the $SAR_e$ and $EC_e$ respectively as well as the long-term groundwater quality which depends on the $EC_d$ and $SAR_d$.

While the high alkalinity and sodicity of PW were expected to impact the long-term $pH_e$ and $Alk_e$ (Beletse et al., 2008), the simulation of irrigation with PW showed that PW does not significantly affect these soil parameters for most dryland soils (Chapter 5). Indeed, although the accumulation of carbonate ions and sodium in the soil is a common origin of soil alkalinisation in drylands (Merry, 2009), the major soil types in drylands such as Calcisols, Gypsisols and Vertisols are relatively rich in $CaCO_3$ which already gives a high pH and alkalinity to these soils by dissolving into $HCO_3^-$ (Koohafkan and Stewart, 2008). Therefore, the quantity of alkaline ions brought by PW is insignificant compared to the huge natural ‘stock’ of $CaCO_3$ contained in these soils which largely determines the soil pH and alkalinity in the long term (Mancino, 2003). In fact, the OFAT sensitivity analysis carried out in Chapter 4 underlined the role of the soil carbonic system (i.e.$CaCO_3$-$CO_2$) in the determination of the long-term $Alk_e$ and $pH_e$. The alkalinity added to the soil through irrigation with PW slightly reinforces the natural alkalinity and pH buffer capacity of alkaline soils in drylands. In contrast, irrigation with PW affects the long-term $pH_e$ of acidic soils such as Planosols as these soils have a low $CaCO_3$ content and thus, a limited pH buffering capacity (Chapter 5). As a practical illustration, Tarchouna et al (2010) demonstrated in a field experiment that long-term irrigation (26 years) with alkaline TSE on acidic Arenosols in Mediterranean climate increased the $pH_e$ by 0.8 units compared to a non-irrigated soil. Nonetheless, alkaline soils are more common than acidic soils in drylands as the latter generally occur in humid zones (Osipov and Minin, 2009). Consequently, soil alkalinisation as a result of irrigation with PW is not a critical risk in most dryland soils.

The simulations in Chapter 5 also question the relevance of referring to guidelines that are only based on irrigation water quality when assessing the risk of soil salinisation and sodification as result of irrigation. In fact, the Ayers and Westcot (1985) guidelines, which have been widely used in irrigation sustainability assessments, do not consider the difference between soils and climates for determining the risks of soil salinisation and soil structural destabilisation. The ANZECC guidelines used in the Chapters 5, 6 and 7 are more sophisticated by including the clay content to discriminate soil types as their vulnerability to sodification differ. Still, there is need to also include aridity as a determinant parameter for the sustainable use of irrigation water as a decreasing AI was shown to exacerbate soil salinisation and sodification. The results of this research underline the need for the development of new water quality guidelines for irrigation which are primarily soil-based and include aridity to determine threshold irrigation water EC and SAR values.

### 8.1.3 Irrigation strategies for agro-environmentally sustainable irrigation with produced water in drylands

The identification of key parameters to the agro-environmental sustainability of irrigation with PW in Chapters 4 and 5 was essential to identify the environmental conditions promoting or disadvantaging the reuse of PW in an agro-environmentally sustainable manner. It was also useful for discriminating parameters between those related to the natural conditions which can be hardly changed such as climate and soil, and those on which we have a relative control such as the irrigation amount and the PW quality.

The simulations in Chapters 6 and 7 have shown that agro-environmentally sustainable irrigation could be achieved by:

- Over-irrigation: leaching excessive salt out of the root zone through drainage water by increasing the irrigation amount beyond the crop water needs and the field capacity.
- PW blending: diluting the salt dissolved in PW by blending with water of lower salinity (e.g. fresh water, TSE or desalinated PW) to decrease irrigation water salinity.
- PW desalination: reducing the salt load introduced into the soil-water system by (partially) desalinating PW prior to its application to the soil.

Short-term field experiments have shown the positive impacts of over-irrigation (Norvell et al., 2009) and PW blending (Atia, 2017; Martel-Valles, Benavides-Mendoza and Valdez-Aguilar, 2017; Mullins and Hajek, 1998; Sintim et al., 2017) on soil salinity and sodicity as well as on crop yield (Chapter 2). However, the studies that have used over-irrigation or PW blending to mitigate the negative impacts of PW on soil salinity and sodicity have used these techniques individually but not in combination. In Chapters 6 and 7, over-irrigation and PW blending were simulated conjunctly, this showed that multiple combinations could be used to achieve agro-environmentally sustainable irrigation with PW.

Irrigation managers might prefer over-irrigation as this practice allows the use of low-quality irrigation water (i.e. a higher proportion of PW in the irrigation water blend). In Qatar for instance, the simulated irrigation of sugar beet showed that using an irrigation volume up to ~300% of the crop water needs with a blend composed of two-thirds PW and one-third TSE could preserve the soil stability, crop yield and groundwater quality (Chapter 6). The same way, simulations of irrigation in Oman with a blend composed of two-thirds PW and one-third ROPW used on jojoba at ~400% of the crop water was shown to be agro-environmentally sustainable (Chapter 7). On the contrary, irrigation managers might be concerned about water efficiency in the field to minimise the cost of adding ROPW or TSE in the irrigation water, minimise pumping cost, and maximise farmer's revenue through irrigating the largest possible area. In this case, higher irrigation water quality is required. For example, irrigation of sugar beet at a little over the crop water needs was shown to be agro-environmentally sustainable in Qatar if PW was mixed with an equivalent volume of TSE or four equivalent volumes of ROPW respectively. The irrigation of jojoba in Oman at 100% of the crop water needs required an irrigation water blend composed of one-fourth PW and three-fourths ROPW.

Although water-efficiency is seen as a priority in water-scarce drylands, both over-irrigation and PW desalination paradoxically lead to higher water losses compared to a standard irrigation scenario covering the crop water needs with fresh water. Increasing the irrigation amount to leach salt out of the root zone leads to a loss of water through drainage. On the other hand, improving the irrigation water quality by partially desalinating PW leads to a loss of water though RO-brine. When the simulated scenarios using over-irrigation and those using PW blended with ROPW were compared in terms of water consumption, over-irrigation was shown to be more water-intensive (Chapters 6 and 7). Consequently, an irrigation strategy consisting of partially desalinating PW to improve irrigation quality is less water-intensive, less energy-intensive, and is thus less costly compared to an irrigation strategy consisting of minimising water quality and increasing the irrigation amount to achieve agro-environmental suitability.

The performance of an irrigation strategy in maintaining the long-term $EC_e$ and $SAR_e$ below target threshold values depends on the right combination of over-irrigation and PW blending. The simulations of irrigation in Chapter 6 have highlighted that the marginal effect of over-irrigation and PW blending on the $EC_e$ and the $SAR_e$ differed in terms of dynamic and amplitude. Diminishing returns were observed regarding the marginal effect of over-irrigation on the reduction of the $EC_e$ and $SAR_e$. In contrast, increasing returns were observed regarding the marginal effect of PW blending to reduce the $EC_e$ and $SAR_e$. Consequently, the most efficient manner to preserve the soil structural stability and the crop yield was to find the optimum balance between salt leaching through over-irrigation and salt dilution through PW blending (Chapters 6 and 7).

### 8.1.4 Operating cost of agro-environmentally sustainable irrigation strategies with produced water in drylands

Despite the existence of effective technologies (e.g. PW desalination) and irrigation management techniques (e.g. over-irrigation) to achieve agro-environmentally sustainable irrigation with PW in drylands, the challenge remains financial. Indeed, as Hillel (2000, p.61) states, "*irrigation is sustainable but at a cost"*. The operating costs of agro-environmentally sustainable irrigation strategies were estimated in a regional (Chapter 6) and in an industrial context (Chapter 7).

The cost of over-irrigation is mainly driven by pumping cost, whereas the cost of blending PW depends on the production cost of the effluent which is mixed with PW to enhance the irrigation water quality. The preservation of the soil and the aquifer through salt leaching required high irrigation amounts and thus the cost of pumping and blending PW increased proportionally to the leaching requirement. In fact, the least-cost agro-environmentally sustainable irrigation scenarios minimised pumping by applying a low irrigation amount while increasing the PW dilution as much as necessary to preserve the soil structural stability and crop yield. The estimated operating costs of agro-environmentally sustainable irrigation schemes with PW are between \$0.19–\$0.37/$m^3$ when PW is blended with TSE (Chapter 6). The estimated operating cost range increases up to \$0.46–\$1.09/$m^3$ in Qatar and up to \$0.75/$m^3$ in Oman if PW needs to be partially desalinated, these estimates fit within the operating cost range reported by Burn et al (2015) regarding the use of RO to desalinate brackish water for irrigation purpose.

The cost of traditional PW disposal practices greatly depends on locations varying between \$0.31–\$16.67/$m^3$ for deep-well injection and \$0.06–\$0.50/$m^3$ for surface discharging (Fakhru'l-Razi et al., 2009). In Oman, the injection of PW into deep disposal wells costs \$0.30/$m^3$ (Hardisty, 2010) which is nearly one third of the cost of reusing partially desalinated PW in irrigation. The cost of disposing of PW in Qatar remains unknown. However, if the cost of injecting PW into deep disposal well in Qatar would be as costly as in Oman, blending PW with TSE prior reusing it in irrigation could be competitive in Qatar.

## 8.2 Research novelty and original contribution

This research has contributed to advance the empirical knowledge regarding the sustainable reuse of O&G PW for agricultural irrigation in drylands through:

- The creation of a dataset collating quantitative data on PW volume generated in several dry countries and regions, quality of PW from different regions and type of O&G productions, and PW management practices.
- The testing and application of the SALTIRSOIL_M model to a new context: modelling the long-term impacts of irrigation with PW on the soil structural stability, crop yield and groundwater quality in dry environments.
- The estimation of the long-term impacts of irrigation with PW of different qualities on the soil structural stability, sugar beet relative yield and drainage water quality using the SALTIRSOIL_M model on soil types (Arenosol, Gyspisol, Planosol and Vertisol) and climates (hyper-arid, arid, semi-arid and dry sub-humid) typical of dryland.
- The development of a framework integrating agriculture, hydrology and economics to assess the agro-environmental sustainability of irrigation with PW in drylands, which could be applied more widely to saline-sodic irrigation waters.

- The testing and application of the framework in two situations: a regional-scale study in Qatar and an industrial case study in Nimr oil field (Oman).
- The identification of agro-environmentally sustainable irrigation strategies with PW using over-irrigation, PW blending and PW desalination, and the estimation of their operating costs in two hyper-arid locations.

## 8.3 Practical implications of the research findings

### 8.3.1 The potential for reusing produced water in agricultural irrigation: current situation and future projections

The PW volume estimates in Chapter 3 gave an order of magnitude of the volume of PW that is potentially reclaimable to irrigate crops in drylands. In a given area, the available PW volume estimates can be compared to the volume of water used in irrigation or to the total volume of renewable water (Chapters 6 and 7) to inform about the significance of the PW resource. These volume estimates may be used by professionals involved in PW management such as O&G firms, farmers and local authorities to support the consideration of PW reuse in irrigation. Furthermore, the data provided in Chapter 3 can be used to compare PW and other unconventional water resources that are potentially available in a specific location (e.g. TSE, industrial wastewaters, brackish groundwater and seawater) in terms of volume, availability and quality. This way, decision makers can set priorities between reusing PW and/or different types of unconventional water resources.

The potential and relevance of reusing PW in agricultural irrigation in drylands are likely to be strengthened in the future. Indeed, the FAO projections to 2050 forecast increasing global demand for food and irrigation water while arable land and water resources per capita are expected to plateau or decrease depending on the evolution of agricultural productivity, land and water management (FAO, 2009d). In the Middle East and North Africa region, for example, one of the driest area in the world and a major O&G producing region, the irrigation water demand is projected to increase by 24–33% between the periods 2000-2009 and 2040-2050. This increase in irrigation water demand is fuelled by a rising food demand due to population growth and by an increasingly drier climate (Sewilam and Nasr, 2017). The pressure on freshwater resources combined with higher food demand and climate change are likely to make PW reuse in irrigation more and more significant even for the most challenging PW qualities. The increasing demand for irrigation water is unlikely to be met with depleting fossil groundwater resources nor with expensive desalinated seawater. Indeed, groundwater over-abstraction is unsustainable and desalinated seawater remains costly and almost exclusively used for domestic and industrial purposes. Therefore, municipal and industrial treated effluents, including PW are in a good position for partially covering the future demand in irrigation water. In parallel to the need of the agricultural sector to secure water resources for irrigation in dry areas, the O&G sector needs to develop alternatives to traditional PW disposal techniques which are getting costlier due to the volatility of the multiple inputs and services determining PW disposal cost (e.g. electricity to operate pumps, fuel for PW hauling, commercial disposal fees, chemicals for scale and corrosion control of deep disposal wells, etc.) (Boysen, Boysen and Boysen, 2002) and also because of the increasingly stricter environmental regulations (Stanic, 2014). As a consequence, O&G firms and irrigators will have increasing motivations for deepening their cooperation regarding the reuse of PW for agricultural irrigation in drylands.

### 8.3.2 Agro-environmentally sustainable irrigation with produced water in drylands: environmental and technical conditions

In certain ancient societies which heavily developed irrigated agriculture in dry areas (e.g. Mesopotamia, Indus Valley and Viru Valley), unsustainable irrigation practices leading to soil salinisation and sodification were identified as a major element causing the decline and eventually the collapse of these civilisations (Altaweel, 2013; Shahid, Zaman and Heng, 2018). These issues are still threatening irrigated agriculture nowadays. Indeed, 30% of irrigated land is affected by soil salinisation and sodification and about 1.5 Mha are becoming unproductive every year (Cherlet et al., 2018). The cost of land degradation caused by salinisation is estimated at $27.3 billion annually (Qadir et al., 2014). To avoid worsening these problems with PW reuse in irrigation, it is necessary to consider the lessons learnt from the current study which has improved our understanding of the agro-environmental sustainability of the reuse of PW in irrigation.

The soil texture, water retention properties, bulk density, gypsum and calcium carbonate contents, as well as climate aridity, should be carefully characterised and considered prior establishing a project of PW reuse in irrigation as these parameters are critical and cannot be significantly changed through human interventions. Environments combining sandy soils rich in gypsum and calcite with dry sub-humid climates are more likely to host agro-environmentally sustainable irrigation schemes with PW. In contrast, hyper-arid locations where soils are clayey with low gypsum and calcite contents should be avoided unless drastic mitigation measures are taken such as treating PW up to unrestricted irrigation quality grade or reusing PW only for soilless agriculture.

The environment also determines the irrigation strategies that can be used to preserve the soil structural stability, crop yield and groundwater quality. Over-irrigation should be used depending on irrigation water availability and groundwater contamination risk. Actually, even if over-irrigation could preserve the soil structural stability and an acceptable crop yield, it could threaten groundwater quality by transferring the risk of salinisation and sodification from the soil to the aquifer (Chapter 6). The possibility of blending PW to achieve agro-environmentally sustainable irrigation depends on the availability and quality of a second water resource. Ideally, fresh water could be used to dilute PW, however, fresh water is scarce in drylands and there might be restrictions on surface water or groundwater abstraction as it is the case in Qatar (Chapter 5). Therefore, unconventional water resources such as TSE and ROPW could be used to improve PW quality. Moreover, by using unconventional water resources, irrigators would not compete with other economic sectors to access these water resources.

In the simulated scenarios, the crop was assumed to be irrigated with irrigation water of constant quality from the early to late growth stages. In practice, particular attention has to be paid to the germination and early growth stages of the crop. Indeed, crops have different sensitivities to soil salinity throughout their development. A germination test has shown significant negative correlations between PW salinity and the germination rate of rapeseed and switchgrass (Pica et al., 2017). Consequently, besides selecting halotolerant crops, the compatibility of PW quality to the crop $EC_e$ thresholds values for each development stage should be investigated prior using PW in irrigation. At the crop establishment and early growth stages, it might be necessary to avoid irrigation with saline and very saline PWs and prefer the use of fresh water before starting the incorporation of PW in the irrigation water blend.

In summary, successful irrigation with PW necessitates essential criteria:

- Selection of a site where the soil is well-drained, with a low water retention capacity, a low clay content as well as high gypsum and calcium carbonate contents to limit the soil vulnerability to salinisation and sodification.
- Selection of a site where the aridity index is high, to maximise salt leaching and dilution by rainfall and to minimise salt concentration into the topsoil as a result of water evaporation.
- Selection of halotolerant crops with low water requirements to maximise crop yield and limit the salt load brought to the soil by the irrigation water.
- Adapted irrigation management to preserve the soil structural stability and crop yield (i.e. over-irrigation, PW blending, PW desalination, soil and irrigation water SAR adjustment).
- The soil fertilisation must be adapted to preserve the soil and groundwater salinity and sodicity. Thus, fertilisers containing sodium and chloride such as $NaNO_3$ and KCl must be avoided.
- Adapted irrigation management to preserve groundwater quality. The generated drainage water must at least be of similar quality as the groundwater. Alternatively, drainage water quality must be improved by improving the irrigation water quality, or drainage must be prevented by minimising the irrigation amount to avoid reaching field capacity. Lastly, drainage water can be intercepted and disposed of in deep injection wells, evaporation ponds, or discharged into the sea.

### 8.3.3 Financial implications of agricultural irrigation with produced water in drylands: cost sharing and minimisation

Developing the affordability of PW reuse in agricultural irrigation might be possible through an equitable share of cost between the O&G industry and irrigators. The share of the cost paid by irrigators should be low enough to compete with the cost of irrigation using other unconventional water resources while the share of the cost paid by O&G firms should be low enough to compete with the cost of traditional PW disposal practices. Due to the site-specificity of each PW reuse project, the cost of reusing PW in agricultural irrigation should be compared to the cost of traditional PW disposal practices and to the cost of using other unconventional water resources in irrigation in a case-by-case approach. Considering that O&G firms pay between \$0.31–\$16.67/m$^3$ for injecting PW into deep disposal wells and around \$0.06–\$0.50/m$^3$ for discharging PW on the surface (Fakhru'l-Razi et al., 2009) and that farmers growing high-value fruits and vegetables in dry zones of high-income countries are willing to pay for irrigation water up to \$0.85/m$^3$ (Barron et al., 2015); there is a chance that O&G firms and irrigators could together cover the whole operating cost (\$0.19–\$1.09/m$^3$) of reusing PW in agricultural irrigation in drylands.

The need for specific soil salinity and sodicity mitigation techniques such as over-irrigation, PW blending and desalination may represent a financial barrier for stakeholders. Minimising the financial cost of irrigation with PW requires:

- Minimising the energy use for treating PW and pumping irrigation water.
- Minimising the volume of irrigation water applied per hectare.
- Selection of a low-cost water resource to blend PW and improve the irrigation water quality.
- Selection of a low-cost source of energy to power pumps (for blending PW and irrigating the field) and the PW treatment process.

Using partially desalinated PW to irrigate crops remains challenging principally because of the high production cost of ROPW. However, the continuous decrease of desalination cost could be a game changer as using desalinated brackish water was proven to be profitable for high-value crops which require costly investments (e.g. vegetables in greenhouses) in places where the cost of water is relatively high (Barron, Campos-pozuelo and Fernandez, 2016; Burn et al., 2015). Subsidised desalinated seawater is also used to irrigate high-value crops in the water-scarce coastal regions of Israel and Spain (Martínez-Alvarez, Martin-Gorriz and Soto-García, 2016). Although PW desalination in coastal areas would be uneconomic for the PWs that are more saline than seawater (≥ 10–60 dS/m). This is, however, less likely to be the case in subtropical and tropical zones (i.e. between -40° and +40° of latitude) where seawater is the most saline in the world with an EC ranging from 40–60 dS/m (Tyler et al., 2017). In such cases and if O&G PW is generated nearby, PW can be an economic alternative if PW salinity is lower than seawater salinity. The same way, in inland regions, if brackish surface water or groundwater resources are available near irrigated lands, only the PWs that have a lower salinity compared to these alternatives resources would be economically interesting to be desalinated for being used in irrigation.

### 8.3.4 Produced water reuse in irrigation and other beneficial reuse practices

Reusing PW in irrigation has several environmental benefits and could be financially profitable compared to disposal practices. Nonetheless, agricultural irrigation is not the only reuse option, thus, other PW reuse practices might, in some cases, be more advantageous in terms of risks and benefits (Table 7-5).

The risks of environmental contamination can be reduced by:

- Not applying PW to the soil.
- Treating PW up to a high quality standard.
- Minimising the volume of PW applied to the soil or the frequencies of the applications.

Avoiding the application of PW to the soil suppresses the risk of contaminating soils, plants and aquifers. Reuse practices which do not involve contact between PW and soil include livestock watering, power generation, cooling and metal recovery. Also, preventing environmental pollution as a result of reusing PW in irrigation can be achieved by adopting stringent water quality standards. Indeed, reuse practices such as aquaculture, aquifer recharge, stream flow augmentation, and potable water supply require drastic reductions of PW contaminants. Therefore the risk of releasing high loads of contaminants into the environment is very much reduced. Finally, environmental pollution can be prevented by minimising the volume of PW that is used and the frequencies of the applications. The reuse of PW in the construction sector (e.g. dust control and drilling) generally involves limited volumes while seasonal activities such as snow control, de-icing and fire control limit the frequency of PW application.

Beneficial PW reuse practices also differ in terms of financial cost and benefits as these mainly depend on:

- The type of contaminants that need to be controlled and the cost of removing these contaminants from PW.
- The financial value of the treated PW.
- The market size for the reuse option.
- The government support, regulation and administrative barriers.

The type of contaminants that need to be removed from PW highly impact the PW treatment cost. As an example, the removal of salt through PW desalination is energy-intensive (2.90–4.22 kWh/$m^3$) and thus expensive costing \$0.38–\$1.23/$m^3$ (Chapter 2) whereas the removal of residual hydrocarbons and organics by hydrocyclone and chemical oxidation technologies are cheap to operate (\$0.06/ $m^3$) mainly because they are not energy-intensive (Igunnu and Chen, 2014). While PW salinity and sodicity are critical when PW is reused in irrigation, PW salinity is actually an asset for controlling snow and de-icing roads. PW salinity is also of a minor concern when PW is reused in aquaculture or to control dust in the construction industry. Consequently, the PW treatment cost for these applications is expected to be reduced compared to the reuse of PW in irrigation.

In addition to the cost of treating PW, its value needs to be considered to appraise the benefits of a reuse practice. Although the treatment of PW up to potable standards is expensive, the value of potable water is higher compared to irrigation water (Meng, Chen and Sanders, 2016). On the other hand, when PW is reused in rangeland restoration, impoundment, drilling, fire control, dust control, snow control and de-icing, its value remains lower than for agricultural irrigation. Besides the value of PW (in $\$/m^3$), the volume that can be absorbed by a beneficial reuse practice determines their profitability. The advantage of agricultural irrigation, aquifer recharge, stream flow augmentation and surface impoundment remain in their large absorption capacities compared to other uses such as drilling, fire control, dust control, snow control and de-icing. In fact, the beneficial PW reuse practices absorbing a high PW volume enable scale economies in PW treatment and so, enhances PW reuse profitability.

The regulation and administrative barriers have also a role in the development of PW reuse practices. For instance, the reuse of PW for O&G operations (EOR, drilling and hydraulic fracturing) does not require specific governmental permits in New Mexico (USA) whereas other reuse practices including irrigation are subjected to permit regulation. Consequently, reusing PW in O&G operations is an easier option for firms which are more reluctant in developing other PW reuse practices (Sullivan Graham, Jakle and Martin, 2015). On the contrary, Wyoming and Colorado have adopted regulations encouraging the reuse of PW in irrigation (Hagström et al., 2016). Additionally, governmental subsidies and taxes were identified as important elements supporting the development of PW reuse practices (Mudgal et al., 2015).

Furthermore, beyond its technical and financial aspects, the reuse of PW has social implications. Communities raise concerns over food crops irrigated with PW which is an usual apprehension affecting irrigation projects involving wastewater reuse (Bloomfield and Doolin, 2017). Thus, certain categories of PW reuse practices are likely to be challenging due to negative public perception (i.e. agricultural irrigation, aquifer recharge and potable water supply). Improving public perception regarding PW reuse practices require stakeholders to communicate with transparency about the technical aspects, the justifications, the risks and the benefits of PW reuse projects.

Lastly, high-level policies have a significant impact on the development of PW reuse practices. As an example, the environmental policy conducted by the State of Oman promoting the reuse of PW for environmental conservation, enabled the creation of an artificial wetland for treating PW in Nimr oilfield. This case has been recognised as a successful environmental enhancement project supporting biodiversity (e.g. local flora, migratory birds, amphibians and small mammals) (Stefanakis, Prigent and Breuer, 2018). Also, the agricultural policies aiming at increasing local crop production and food security in the Middle-East countries are certainly positive for the reuse of PW in irrigation. It is important to note that in drylands, the value and the sensitivity of food security issues could outfit the financial cost of a PW reuse practices (Qadir et al., 2007; Darwish et al., 2015). Although the end-use objective of the different PW reuse practices diverge, they all fit into a circular economy approach. The O&G industry and governments are adopting this economic paradigm which constitutes an encouraging environment for the development of beneficial PW reuse practices.

### 8.3.5 Uses of the integrated framework

The framework is applicable to multiple contexts such as prospective studies aiming to identify agro-environmental conditions that are potentially suitable for implementing irrigation projects with PW (e.g. Chapter 5), feasibility studies to assess the agro-environmental sustainability and the operating cost of irrigation with PW (e.g. Chapter 6), and projection studies to support stakeholders' decision in managing an irrigation scheme with PW (e.g. Chapter 7). The developed framework can also be used as a screening tool to identify a threshold PW quality for maintaining irrigation at a sustainable level knowing local climatic and soil parameters. Finally, the framework could inspire guidelines specifically developed to support good management practices for agro-environmentally sustainable irrigation with PW.

## 8.4 Research limitations

### 8.4.1 Framework limitations

The framework can help to anticipate the long-term impacts and the operating cost of irrigation with PW in drylands but the modelling results should not be evaluated for the absolute values. The complexity of predicting the long-term soil salinity and sodicity as well as the operating costs of a potential irrigation project with PW makes it difficult to develop a modelling framework that has accurate predictive capability. This is due to the interaction of physical (e.g. soil response to irrigation) and human processes (e.g. cost of PW management and irrigation management) and the large variability over space and time of the different characteristics of PW and of an irrigation system (e.g. crop rotation, irrigation technologies, irrigation management, etc.). Therefore, it is better to focus on the overall approach of the framework, which deepens the understanding of the agro-environmental sustainability and the operating cost of irrigation with PW in dry areas, than on expecting accurate predictions.

The hierarchy of agro-environmental risk proposed in Chapter 2 gives priorities to the different risks related to the reuse of PW in irrigation. It was found that the agro-environmental risks represented by PW salinity and sodicity were the principal concerns when PW is reused in irrigation. This ranking of risks does not differ much from the one suggested by Elgallal et al (2016) which applies to the reuse of sewage in arid areas. The notable exception relates to the risks posed by nutrients which is an issue for irrigation with sewage effluents but not for irrigation with PW. Although the priority must be given to the risks related to salt and sodium dissolved in PW, the agro-environmental risks related to heavy metals, organic compounds and radioelements deserve to be assessed in a case-by-case approach as PW quality and environmental conditions vary greatly between locations.

The simulated soil salinity and sodicity mitigation strategies included salt leaching, salt dilution and salt removal. Other strategies could also be used, such as altering the soil chemistry, extracting salt from the soil, and mitigating climate aridity. First, the soil chemistry can be altered by, for example, amending the soil with gypsum to adjust the $SAR_e$. This possibility was not considered because (1) soil gypsum amendments could not be simulated with the SALTIRSOIL_M model and (2) amending PW with gypsum requires PW of low salinity to maximise gypsum dissolution to significantly reduce the irrigation water SAR (Silveira et al., 2008). Nonetheless, the long-term impacts of amending the soil with gypsum on the soil and groundwater salinity and sodicity need to be studied as short-term field experiments have shown that gypsum could potentially preserve the soil structural stability (Bennett et al., 2016; Ganjegunte, Vance and King, 2005; Johnston, Vance and Ganjegunte, 2008; Vance, King and Ganjegunte, 2008). Second, although salt can be removed from PW prior to irrigation by desalination, it can also be extracted from the soil after irrigation. In fact, including a specific crop in the rotation such as *Suaeda salsa*, *Allenrolfea occidentalis* and *Salicornia bigelovii* can extract 3-5 t/ha of salt from the soil annually (Ke-Fu, 1991; Morteau, 2016), that represents between the double and the triple of the salt load extracted from the soil by sugar beet (Chapter 5). Thus, phytoextraction can contribute to maintaining the soil salinity to safe levels for the main commercial crop (Hasanuzzaman et al., 2014). Finally, although climate aridity cannot be changed at the regional scale, reducing the ETo at the farm scale is possible by growing crops under closed environments (i.e. greenhouse or container farming). These strategies were not considered but if data are available, they could be simulated using the SALTIRSOIL_M model to estimate how these strategies would eventually mitigate the long-term soil salinity and sodicity.

The current cost analysis only reflects the operating cost of reusing PW in irrigation. Although this practice generally appears to be more expensive compared to traditional PW disposal practices, the costs of the negative externalities of current PW disposal practices (e.g. aquifer contamination, damages caused by induced seismicity on buildings and infrastructures, and surface water contamination) need to be also considered in the cost disposing of PW. As reusing PW in irrigation prevent these problems, the cost of this management option might be more competitive than estimated. Besides, the environmental benefits of reusing PW in irrigation need to be financially quantified. Indeed, the stakeholders in Nimr PW reuse project have observed the benefits brought by the vegetation on wildlife in what used to be a bare desert soil (Stefanakis, Prigent and Breuer, 2018). In order to better reflect the hidden costs of traditional disposal practices and the environmental benefits of PW reuse in irrigation, a cost-benefit analysis (CBA) could be integrated into the modelling framework. Furthermore, a CBA would better quantify the total cost (including the capital cost), but also the economic, social and environmental costs and benefits of reusing PW in irrigation.

### 8.4.2 Limitations of the modelling approach

As any modelling approach, this study has limitations related to the model itself. In the SALTIRSOIL_M model, the simplifications and assumptions may eventually affect the simulated long-term soil and drainage water salinity and sodicity resulting from irrigation with PW.

In the SALTIRSOIL_M model, the crop has only a role of ‘salt concentrator’ through water uptake whereas the crop is also a ‘salt extractor’. When the crop is harvested, the salt load accumulated in the crop is removed from the soil-water system. Depending on the crop, salt uptake can be significant making the crop a salt reservoir along with the soil and drainage water. Indeed, it was estimated in Chapter 5 that sugar beet could export 1.2 t/ha of salt annually. In the case of Qatar (Chapter 6), sugar beet would export between 12% and 38% of the annual salt load introduced by irrigation in the least-cost scenarios with PW-TSE and PW-ROPW respectively. Thus, depending on the crop’s ability to extract salt from the soil, the latter can positively contribute to enhance the agro-environmental sustainability of irrigation with PW by reducing the soil $EC_e$ and $SAR_e$.

The simulated scenarios would ideally need to be tested in field experiments to assess the validity of the predicted agro-environmental sustainability levels of the simulated irrigation scenarios. In fact, combining long-term field experiments with modelling would enhance the confidence in the model predictions. However, the diversity of soils, climates and PW qualities simulated as well as the temporal scope of the simulations (i.e. until $EC_e$, $SAR_e$, $pH_e$ and $Alk_e$ values remain steady meaning that the system equilibrium state is reached) make the testing of the model predictions in all the simulated environments very challenging. Indeed, to date, the only reported long-running and large-scale irrigation scheme with PW that could be used to test the model is located in arid California (USA) and has been operating since 1994 (Heberger and Donnelly, 2015; Myers, 2014). However, it is not certain that the input data necessary for running the model are being collected and monitored onsite. Moreover, data collection and use might be restricted by the site owners (i.e. O&G firm operating the site, farmers, government of California or US federal government), this would probably make the testing of the model challenging. Lastly, even if this site would be accessible, it only represents one particular environment. The lack of sites located in drylands where irrigation with PW has been tested for a long period of time in different climates and on different soils unable the testing and validation of the model predictions.

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sig=ACfU3U20W-CqGDSSU4QjirWOvDbPk_riPg&hl=en&sa=X&ved=2ahUKEwi327Tn-vLgAhUiVBUIHYyKAVgQ6AEwCHoECAUQAQ#v=onepage&q=deep w (Accessed: 8 March 2019).

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# 9 CONCLUSIONS

By identifying the agro-environmental risks posed by irrigation with PW, modelling the long-term impacts of irrigation with PW on soil structural stability, crop yield and groundwater quality, and by estimating the operating cost of irrigation with PW; this research demonstrates that; agricultural irrigation with PW can be agro-environmentally sustainable in drylands and financially competitive compared to traditional PW disposal practices.

First, this research reveals that although 45% of the global volume of PW is potentially available for being beneficially reused, the clear majority of PWs (~92%), cannot be used without adapted irrigation management practices such as salt leaching, PW blending and PW desalination to prevent soil structural destabilisation, low crop yield, and groundwater salinisation and sodification.

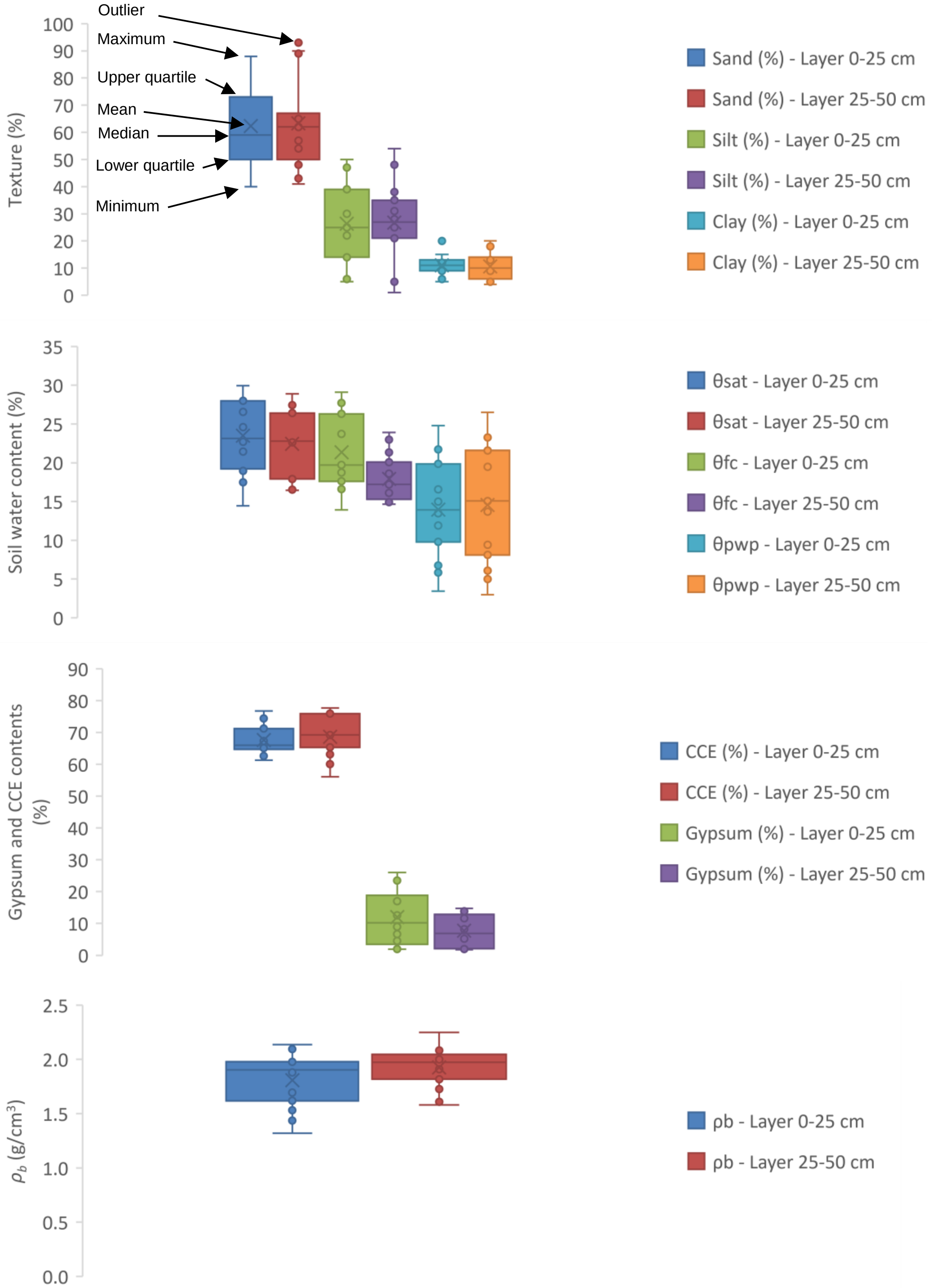
Second, the modelling of the long-term impacts of irrigation with PW on soil salinity ($EC_e$) and sodicity ($SAR_e$) has revealed the crucial role of the soil clay content and the soil water content at field capacity ($\theta_{fc}$) in salt retention and leaching processes and thus, in the long-term determination of the $EC_e$ and $SAR_e$. Additionally, the long-term $SAR_e$ is buffered by the gypsum naturally present in the soil. The simulation of irrigation with PW under different climates has also highlighted that as climate aridity intensifies, the long-term $EC_e$ and $SAR_e$ further increase. On the other hand, irrigation with PW is unlikely to significantly affect the long-term soil pH and alkalinity in most dryland soils as these are alkaline and rich in $CaCO_3$ which buffers soil pH.

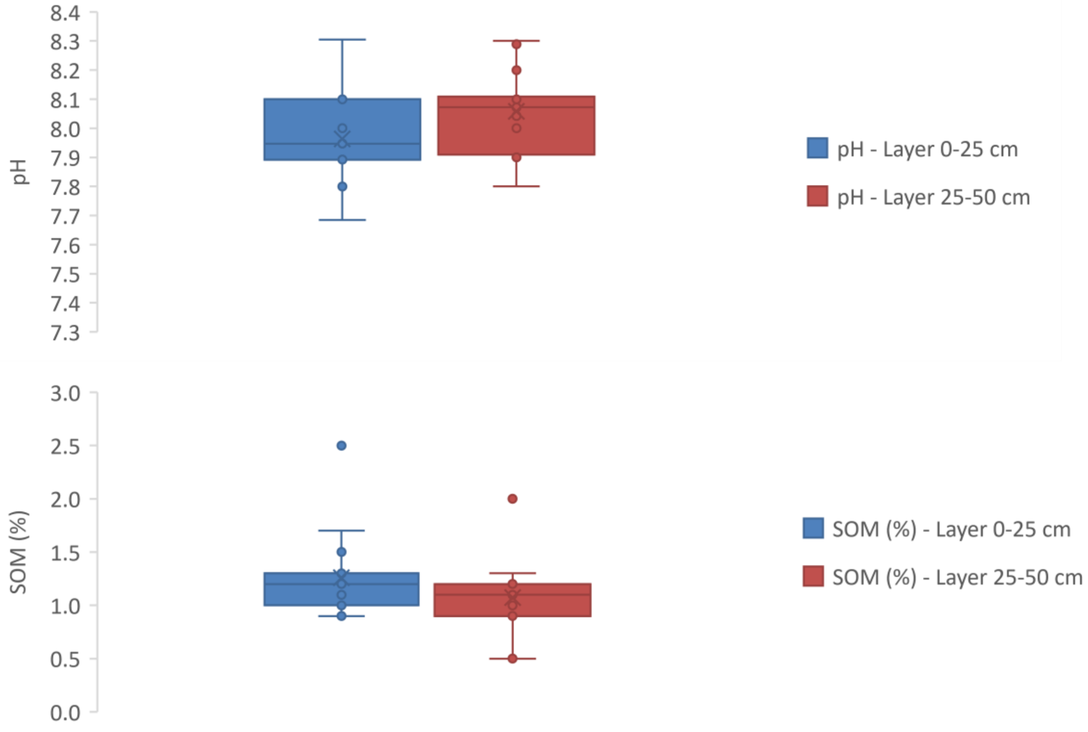
Third, irrigation strategies combining over-irrigation and blending PW with TSE or desalinated PW were proven to be effective to preserve the soil structural stability, the crop yield and the groundwater quality. The financial appraisal of a project aiming at reusing PW in irrigation should be undertaken from a regional or site viewpoint as many financial parameters are site-specific. From the point of view of the O&G industry, managing PW in agro-environmentally sustainable irrigation is generally more expensive than managing PW through traditional disposal practices. Nonetheless, PW reuse in irrigation is likely to be more competitive if the O&G industry and irrigators both contribute to cover its cost.

The research aim was achieved by developing a framework to assess the agro-environmental sustainability of reusing O&G PW for agricultural irrigation in drylands. However, the framework needs further development by integrating the agro-environmental risks posed by heavy metals, organic compounds and radioelements present in PW. Besides, the cost analysis could be broadened by conducting a cost-benefit analysis (CBA) in a case-by-case approach. The framework is meant to be applicable to the large diversity of environments (soils and climates) encountered in drylands and to the wide range of PW qualities. However, a thorough assessment of the sustainability of irrigation with PW in drylands is only feasible at a site or local scale.

# APPENDICES

## Appendix 1 Soil samples analyses results (Nimr oil field, Oman).






Soil layer 0–25 cm, n = 15, soil layer 25–50 cm, n = 15

Note: Unexpectedly, the measured $\theta_{sat}$ values were lower than the measured $\theta_{fc}$ values. The reasons explaining this abnormality were not identified with confidence although the principal cause could be due to a defaulting calibration of the electronic scale between the measurements of $\theta_{sat}$ and $\theta_{fc}$ values.

## Appendix 2 Soil sample collection in Nimr oil field (Oman) and laboratory analyses carried out in Cranfield University (Cranfield, Bedfordshire, UK) and in Milton Keynes Open University (Milton Keynes, Buckinghamshire, UK).

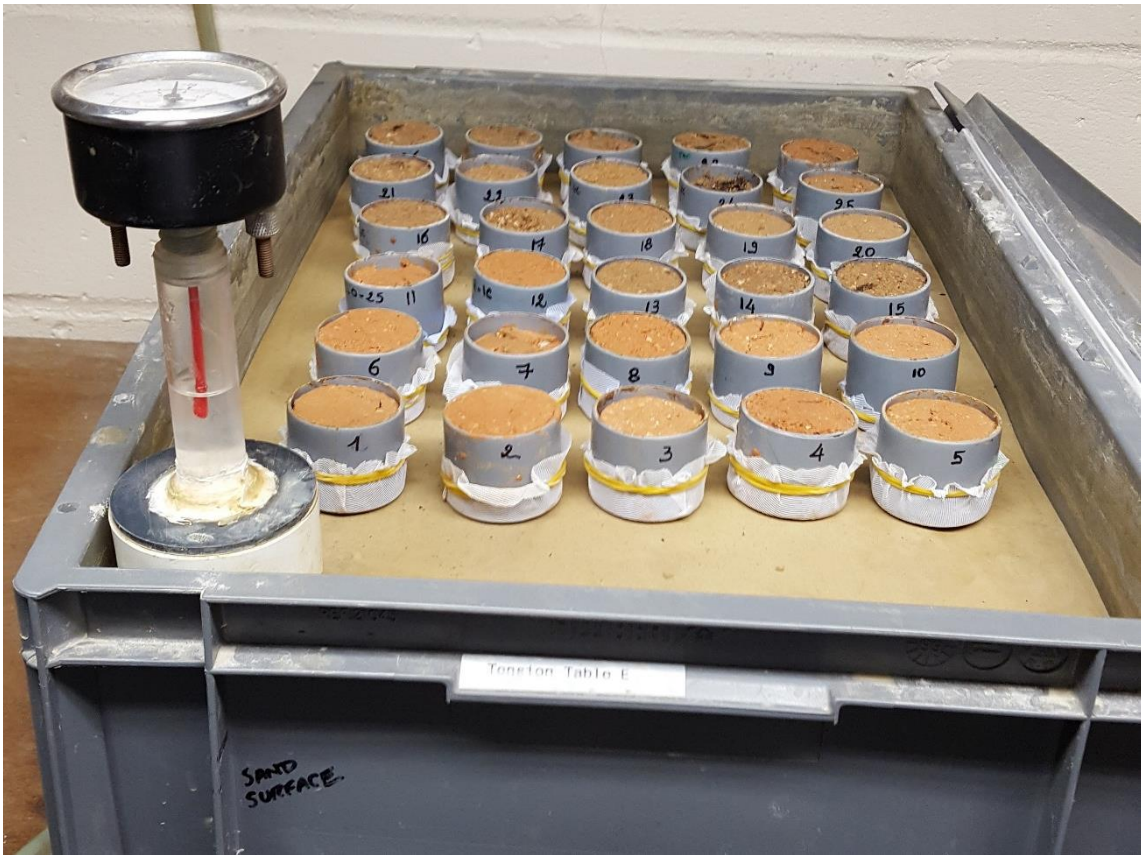
**Sampling of disturbed soil 25 cm of depth for bulk density measurement.**

**Sampling of disturbed soil at 25 cm and 50 cm of depth.**

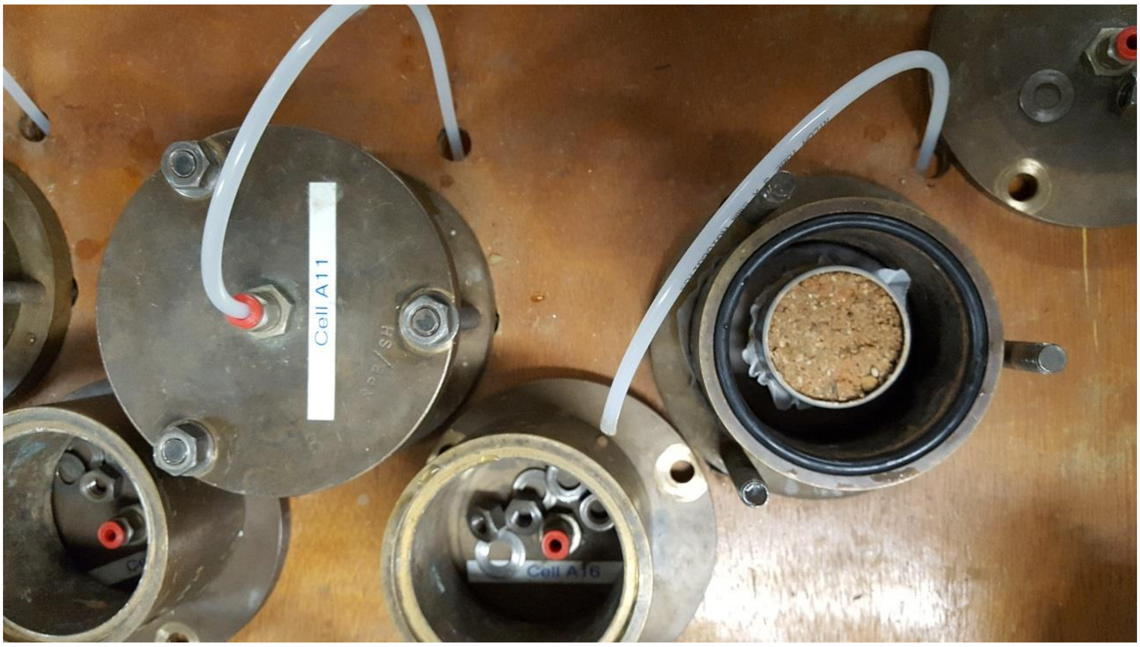
**Sampling of undisturbed soil at 25 cm and 50 cm of depth.**

**Undisturbed soil samples being water-saturated on foam wetting-up bath.**

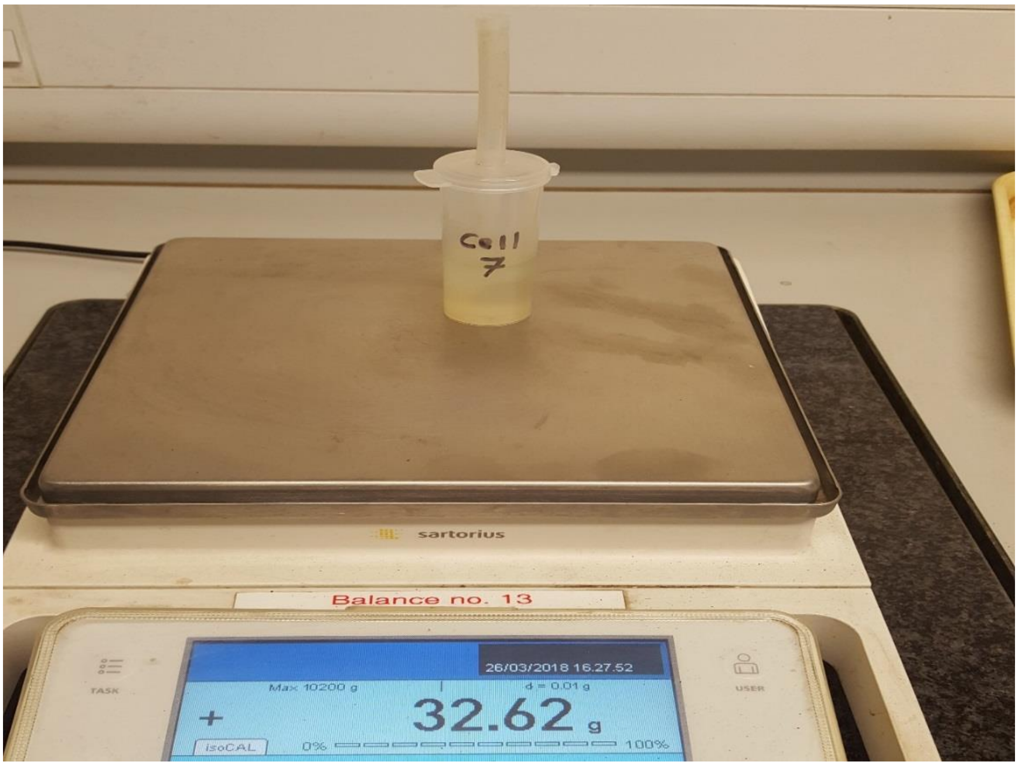

**Water-saturated undisturbed soil samples being weighed.**




**Undisturbed soil samples on a sand table transitioning from saturation to field capacity.**




**Undisturbed soil samples in a pressure membrane cell transitioning from field capacity to permanent wilting point.**

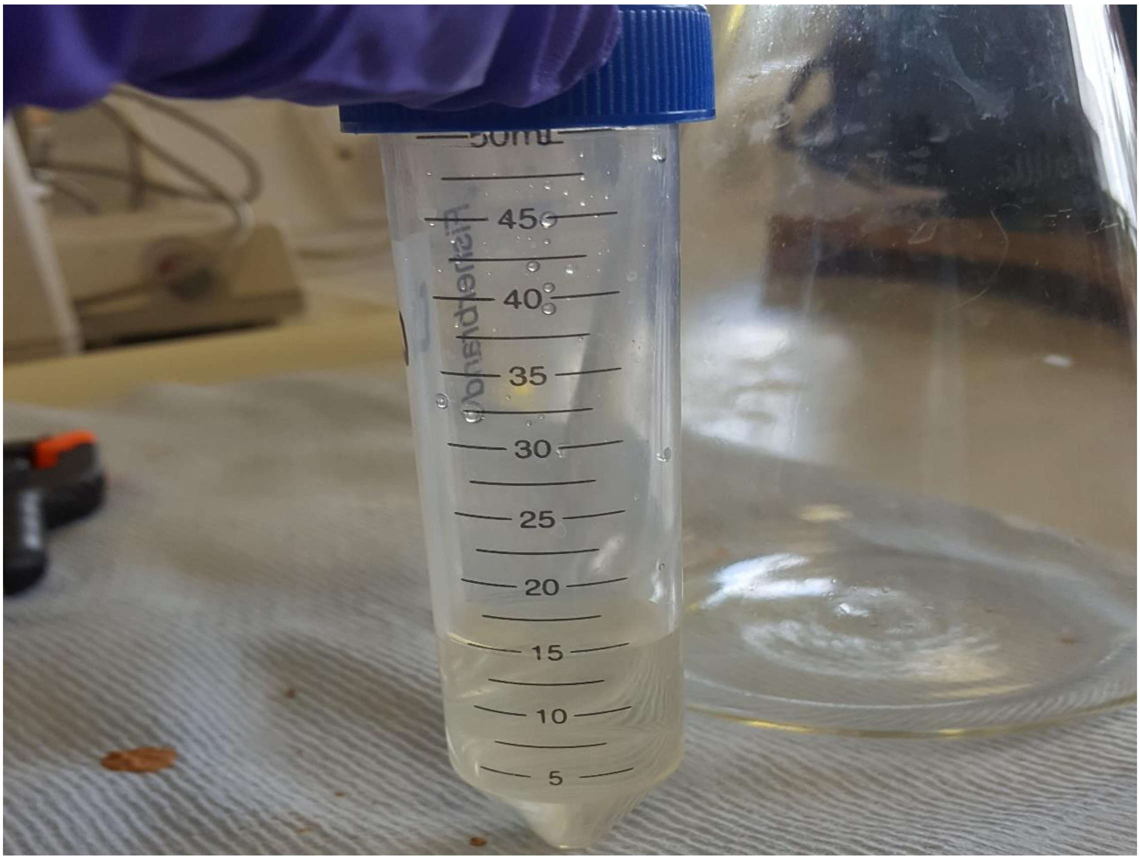



**Soil water extracted from an undisturbed soil sample at permanent wilting point being weighed.**


**Measurement of the electrical conductivity at 25°C of soil samples diluted in water for the determination of the soil gypsum content with a conductivity meter (Orion™ Versa Star Pro™, ThermoFisher).**

**Preparation of a saturated soil paste (left) from air-dried soil (right).**

**Extraction of the soil saturation extracts.**




**A soil saturation extract.**


**Measurement of the alkalinity (as $CaCO_3$ equivalent) of the soil saturation extracts with an automatic tritrator (TITRONIC® 96, SCHOTT).**

**Evidence of soil salinisation: crystallised salt deposits on the surface (alfalfa under drip irrigation).**

**Evidence of soil salinisation and sodification: Crystallised salt deposits on the furrow edges and sticky soil clogged by dispersed clay (eucalyptus under flood irrigation).**