This is a previous version of the article published in Environmental Earth Sciences. 2018, 77:197. doi:10.1007/s12665-018-7386-6

TITLE PAGE

Impacts of agricultural irrigation on groundwater salinity

A. Pulido-Bosch · J. P. Rigol-Sanchez · A. Vallejos · J. M. Andreu · J. C. Ceron · L. Molina-Sanchez · F. Sola

A. Pulido-Bosch · J. P. Rigol-Sanchez (e-mail: jprigol@ual.es) · A. Vallejos · L. Molina-Sanchez · F. Sola

Water Resources and Environmental Geology, Department of Biology and Geology, University of Almeria, E-04120 − Almeria, Spain

J. M. Andreu

Department of Earth Sciences and Environment, University of Alicante, E-03080 − Alicante, Spain

J. C. Ceron

Department of Earth Sciences, University of Huelva, E-21071 − Huelva, Spain

Abstract Agricultural irrigation represents the main use of global water resources. Irrigation has an impact on the environment, and scientific evidence suggests that it inevitably leads to salinization of both soil and aquifers. The effects are most pronounced under arid and semi-arid conditions. In considering the varied impacts of irrigation practices on groundwater quality, these can be classed as either direct – the direct result of applying water and accompanying agrochemicals to cropland – or indirect – the effects of irrigation abstractions on groundwater hydrogeochemistry. This paper summarizes and illustrates through paradigmatic case studies the main impacts of irrigation practices on groundwater salinity. Typically, a diverse range of groundwater salinization processes operating concomitantly at different time scales (from days to hundreds of years) is involved in agricultural irrigation. Case studies suggest that the existing paradigm for irrigated agriculture of focusing mainly on crop production increases has contributed to widespread salinization of groundwater resources.

Keywords Aquifer · Groundwater · Impacts · Irrigation · Over-exploitation · Salinization

Acknowledgements This study was supported by CEIMAR "Campus de Excelencia Internacional del Mar" and the projects CADESEM (Grant number P11-RNM-8115), Junta de Andalucia Regional Government, Spain, and KARSTCLIMA (Grant number CGL2015-71510-R) and ACUIDESAL (Grant number CGL2015-67273-R), Spanish Ministry of Science and Technology – FEDER.

MANUSCRIPT

Impacts of agricultural irrigation on groundwater salinity

Introduction

Significant changes on the terrestrial water cycle have occurred in many areas of the world as a result of the global expansion of agriculture in the last decades. Understanding the impacts of irrigated agriculture on hydrological systems is fundamental to implementing management programs that are effective in maintaining water resources. Agricultural practices can destruct natural vegetation and deteriorate soils, surface water bodies and aquifers. Agricultural irrigation poses a potential threat especially in arid and semi-arid areas, where evapotranspiration rates are typically high and precipitation is scarce and varies considerably both inter- and intra-annually (Oren et al. 2004). About 80% of the total cropped land in the world that is equipped for irrigation lies in arid and semi-arid subtropical zones (where more than 75 % of the global population live), and about 75% is located in developing countries (Morris et al. 2003; Han et al. 2011).

Since the 1950s water withdrawals for irrigation have almost doubled and, despite of improved irrigation management and practices, it is estimated that the amount of water used by agriculture will increase by 14% by 2030 (FAOSTAT 2016). About 43% of the global irrigated area is supplied from groundwater, and 57% from surface water (AQUASTAT 2016). Agriculture irrigation increases the amount of water applied to the soils typically enhancing groundwater recharge (Suarez 1989; Scanlon et al. 2005; Foster and Perry 2010). It is estimated that about 30% of global irrigation water withdrawals flows back to local hydrological systems by return flows and conveyance losses to groundwater and rivers (Scanlon et al. 2007). Changes in aquifer recharge in irrigated areas can have negative impacts on groundwater quality. The underlying aquifers can be impacted by several processes that lead to the contamination of groundwater (e.g. salt concentration by evapotranspiration, rising water-table and waterlogging, subsurface salt/chemicals leaching, seawater mobilization). Furthermore, intense groundwater abstraction for agricultural irrigation has resulted in the depletion and deterioration of aquifers all over the world (e.g. Ceron and Pulido-Bosch 1996; Konikow and Kendy 2005; Scanlon et al. 2005; Rodell et al. 2009; Wada et al. 2010; Vallejos et al. 2015; Faunt et al. 2016). Regarding water quality, salinization and nutrient and pesticide pollution have been identified as the main problems associated with agriculture worldwide (Mateo-Sagasta and Burke 2010). Salinization is the most widespread form of groundwater contamination (Richter and Kreitler 1993).

Groundwater is considered to be saline when the contents of dissolved solids in terms of the concentration level (i.e., salinity level) are above a predefined limit (usually 1,000 milligrams dissolved solids per litre of water, mgL^{-1}, or Total Dissolved Solids (TDS); Freeze and Cherry 1979). According to its origin, saline groundwater can be broadly classified into four genetic categories (Fig. 1; Van Weert et al. 2009): (A) marine, (B) terrestrial (natural), (C) terrestrial (anthropogenic), and (D) mixed origin. This study will primarily focus on category C, which can be further categorized into (C1) Produced by irrigation (input of concentrated residual water), and (C2) Anthropogenically polluted groundwater. While the former tends to occur in arid and semi-arid zones and at shallow depths (usually restricted to the first meters to tens of meters below the groundwater table), the latter occurs anywhere on earth, particularly in modern consumptive societies (Van Weert et al. 2009). Category D groundwater occurs anywhere on earth, although hydraulic gradients typically facilitate the mixing processes. Category D will also be considered but restricted to cases involving category C groundwater.

[Please Insert Fig. 1 about here]

Moreover, the spatial distribution of saline groundwater is subject to change. The genesis of saline groundwater and its migration and mixing are put into motion by certain drivers (Van Weert et al. 2009). These driving forces can be natural processes (geological or meteorological processes) or anthropogenic factors such as drainage, agricultural irrigation, groundwater abstraction, and waste or wastewater disposal (Fig. 2). Hence, irrigation is recognized as one of the main drivers affecting groundwater salinity, but as it is shown in sections below it is also often closely related to the other anthropogenic drivers.

[Please Insert Fig. 2 about here]

In consequence, salinization as a result of agricultural activities is found worldwide and is cited as the groundwater quality problem having the greatest environmental and economic impacts (Morris et al. 2003; FAO 2011). Globally, 11% to 30% of the irrigated area is estimated to be affected by some degree of salinity (Ghassemi et al. 1995; FAOSTAT 2016). This paper illustrates some of the main impacts of agricultural activities (especially application of irrigation water and intense irrigation groundwater abstraction) can cause on groundwater salinity. A compilation of studies conducted at a number of paradigmatic sites is included (Murray Basin, Australia; Souf Valley, Algeria; Costa de Hermosillo, Mexico; South East Spain). A distinction between *direct* and *indirect* impacts on groundwater quality is considered. Because of the magnitude of the overall subject this study limits discussion to inorganic salts, and omits nutrients (posing a wider problem) and other factors that may have a large impact on groundwater quality but normally do not contribute or contribute by a very small fraction to groundwater salinity (e.g. pesticides, herbicides, pathogens, heavy metals). Only some specific cases where nitrates reach very high concentrations (e.g. 200−400 mg/L) in groundwater are considered. Special attention is paid to hydrological systems located in arid and semi-arid areas because, a priori, they are more vulnerable than those located in humid and temperate climate zones.

The impacts of irrigation practices on groundwater quality

In considering the impacts of irrigation practices on groundwater quality, these can be classified as either direct – the direct consequences of applying water and accompanying agrochemicals such as fertilizers, herbicides and pesticides, to irrigated cropland – or indirect – the effects of irrigation abstractions on the chemistry of the aquifer water, which are typically evidenced by a continuous degradation of pumped groundwater quality (Fig. 3).

[Please Insert Fig. 3 about here]

Direct impacts

One of the main direct impacts is an increase in salinity of the irrigation return flow (IRF) (Fig. 3a). Irrigation water is regularly applied in excess to satisfy crop water requirements and to leach the salts from the soil (FAO 2011). The fraction of water eventually reaching the water table (recharge) will normally show an increase in salinity relative to the applied irrigation water due to concentration by crop transpiration and evaporation (almost pure water is evaporated and dissolved salts remain in the soil solution) or due to the mobilization of salts accumulated in soil and the unsaturated zone (Suarez 1989; Leaney et al. 2003; Scanlon et al. 2005, 2009). This can result in a one to tenfold increase in salinity levels in return flows relative to applied water (Aragues and Tanji 2003).

Other factors controlling the salinity level of IRFs include quality, volume and rate of applied water, climate, soils, water table depth, type of aquifer, and the specific agricultural, drainage and irrigation management practices (Tanji and Kielen 2002; Aragues and Tanji 2003; Kass et al. 2005; Scanlon et al. 2007, 2010; Garcia-Garizabal and Causape 2010; Merchan et al. 2015). Irrigation water quality will substantially influence the extent of the groundwater salinization process, ranging from fresh water to saline water depending on the source. Since groundwater usually has higher salinity than surface water (especially deep or old groundwater), irrigation effects on groundwater quality will also depend in part on whether groundwater or surface water is the main source of irrigation water (Bohlke 2002). In addition, solute recycling from irrigation can also contribute to aquifer salinization in groundwater-fed irrigation systems (Milnes and Renard 2004). A particular case is irrigation by means of application of wastewater, which is generally more saline than regional groundwater (Kass et al. 2005). In general, lower irrigation rates (e.g. drip irrigation) decrease negative impact of IRF on aquifer salinity but tend to increase the rate of salinization of soil and shallow groundwater because of reduced salt leaching (Scanlon et al. 2010). Thus, specific salt concentration factor in the crop root zone will be determined by irrigation application rates relative to crop evapotranspiration. IRFs pose serious problems in arid and semi-arid areas, where precipitation rates are low and where evapotranspiration rates and salt contents in soil are typically high. Large reservoirs of soluble salts occur naturally within soils and unsaturated zone in vast areas with arid and semi-arid climate around the world (Walvoord et al. 2003). These salts can be mobilized by increased groundwater recharge when such areas are converted to irrigated cropland (Suarez 1989; McMahon et al. 2006; Scanlon et al. 2007, 2009). Mobilization of stored salts can be the major source of salt in the discharge from irrigation regions (Smedema and Shiati 2002).

Soil types can also control salt accumulation. Soils with clayey to loamy-sandy textures tend to show larger salinity levels than coarser textures due to increased residence times allowing more time for evapotranspiration (Scanlon et al. 2010). Fertilizers are customarily applied in irrigation water to increase crop productivity (Fig. 3a). Normally, nitrogenous and phosphorous compounds including K, Cl, Ca, Mg, and S, are utilized. If excessive leaching of fertilizer in soil is produced, it can eventually reach the water table. This typically results in groundwater quality deterioration, particularly caused by high concentrations of nitrate in shallow aquifers (Bohlke 2002). On the other hand, inappropriate disposal of agricultural waste or wastewater can result in saline leachates that can be mobilized from surface downwards to the water table through the unsaturated zone. A particular case is reject brine, which is the major by-product waste of most inland desalination plants for irrigation, and typically contains high concentrations of inorganic salts (Mohamed et al. 2005).

In irrigated areas with shallow groundwater tables, the salinization process is typically more intense, especially in areas with high evaporation rates. Increased shallow groundwater evaporation can occur when the groundwater moves upward into the non-saturated part of the soil because of capillary rise (up to about 1.5 m; Van Weert et al. 2009). Upward capillary water flow has been identified as the main cause for soil and groundwater salinization in irrigated arid areas with shallow groundwater tables (Northey et al. 2006). In addition, in these areas the recharge is immediate and causes the water table to rise, eventually leading to waterlogging and non-beneficial evaporation directly from the water table (Tanji and Kielen 2002). In regions with larger depths to water, return flows have to pass through a thicker unsaturated zone to reach the water table. The application of large amounts of irrigation water to soils and the presence of salt-bearing sediments and evaporite formations (e.g. halite, gypsum) underlying the agricultural area can result in return flows showing increased salinity (Scanlon et al. 2005). Salinization by mobilized evaporite salts can reach severe levels in groundwater leading in some cases to the abandonment of wells and abstraction boreholes (Andreu et al. 2008). The vertical hydraulic conductivity of the unsaturated zone will be a key factor in the downward displacement of the dissolved salts. In the case of thick unsaturated zones (over 15 m), this can take years or decades or even centuries (e.g. 132–188 years for an unsaturated zone thickness of 33–47 m in a semi-arid area; McMahon et al. 2006). Nevertheless, the existence of preferential (and fast) flow paths linked to discontinuities should not be overlooked (e.g. McMahon et al. 2006; Kurtzman et al. 2016). Salts can remain in soil or, more frequently, pass through the unsaturated zone, where various hydrogeochemical processes can take place, including oxidation, reduction, ionic exchange, fixation and precipitation (Stigter et al. 1998; Kass et al. 2005; Lorite-Herrera et al. 2008).

Indirect impacts

Intense or excessive groundwater abstraction for irrigation can lead to groundwater quality deterioration in agricultural areas. The negative side-effects of such exploitation can be classed as indirect impacts of irrigation (Fig. 3b). Intense groundwater abstraction reduces the assimilative capacity of the aquifer and normally results in a decline in water levels and a new hydraulic head distribution that may lead to changes in the directions of groundwater flow (Freeze and Cherry 1979). If low quality water (e.g. saline) is part of the subsurface system, then it may encroach upon relatively fresh zones of the aquifer (Richter and Kreitler 1993). Saline waters, for instance, in aquitards adjacent to the aquifer under extensive pumping can be mobilized towards pumping boreholes. This usually causes a gradual increase in pumped groundwater salinity as aquifer depletion progresses. In coastal aquifers, over-abstraction typically results in seawater intrusion. Mixing with just 10% seawater renders fresh groundwater unfit for irrigation of most traditional crops (Maas 1986). Many productive coastal aquifers all over the world have been salinized due to seawater intrusion induced by intensive pumping of groundwater for agricultural use (e.g. Barlow and Reichard 2010; Shi and Jiao 2014). Saline intrusion is not restricted to coastal aquifers since old saline waters (e.g. connate water) may occur both in coastal and inland aquifers at depth (Morris et al. 2003). Thus, many aquifers worldwide have deteriorated due to connate water upconing from deeper aquifers (Molina et al. 2002; Szynkiewicz et al. 2008; Baghvand et al. 2010). Saline intrusion is consequently one of the most widespread causes of aquifer salinization (Barlow and Reichard 2010). In some detrital aquifers, irrigation over-abstraction can also have other negative consequences such as aquifer compaction and land subsidence (Faunt et al. 2016). These are not quality issues but in some cases can result in groundwater quality deterioration due to changes in physicochemical conditions. Substantial or fast water level declines can result in aquifer decompression and release of gasses, that in turn may salinize groundwater (Ceron and Pulido-Bosch 1996). Introduction of excess dissolved oxygen in aquifer pores may result in oxidation of the original immobile minerals, releasing metallic ions (e.g. arsenic; Morris et al. 2003).

Case studies

Although the aforementioned groundwater salinization processes may take place in isolation, they are more likely to occur concomitantly. In most irrigated areas affected by salinization, various of these impacts are typically identified. For example, IRF evapoconcentration, seawater intrusion and some other impacts are often superimposed in arid and semi-arid groundwater-fed coastal irrigated areas (i.e. category D groundwaters). In addition, local conditions (e.g. hydrogeology) can also play a significant role in determining the extent and rate of the salinization processes (Merchan et al. 2015). Consequently, a variety of combinations between salinization processes may exist. This variety complicates generic approaches to groundwater salinization studies, because the dominant salinization processes are usually site specific. Nevertheless, a number of salient features can be recognized in salinized aquifers in irrigated areas. This is illustrated with eight case studies in the sections below (Table 1). Cases have been selected to show the salinization mechanisms discussed above affecting different aquifer types and extents in irrigated areas around the world with over 50 years of agricultural development.

[Please Insert Table 1 about here]

Murray basin, SE Australia

The Murray Basin (MB; Fig. 4; Table 1) is the most important agricultural region of Australia, with about 475,000 ha of irrigated land (CSIRO 2008). Almost since its inception in the late 19^{th} century (water was pumped and conveyed from the Murray river to farms) salinity problems resulting from stored salts mobilization arose. It contains a sequence of Paleocene to recent sediments up to 600 m thick (Fig. 4a). Several aquifer units can be distinguished (Fig. 4b). The Pliocene Parilla Sands and, to the east, the Shepparton Formation are the upper-most units and are dominant in influencing salinity processes (Evans 2013). Groundwater flows mainly towards the basin central area where it discharges to numerous salt lakes and, to a lesser extent, to the Murray river (Cartwright et al. 2010). Land surface is nearly flat and surface waters and solutes are drained by the Murray river towards the Southern Ocean. While deep aquifer units tend to have (mostly fossil) freshwater, shallow units, as in most of arid/semi-arid Australia, are naturally saline due to salt concentration by evaporation and transpiration (e.g. Parilla unit 10,000–65,000 mg/L TDS; Evans 2013).

[Please Insert Fig. 4 about here]

The MB is a notorious example of dryland salinization and waterlogging due to enhanced recharge (from less than 0.3 mm $year^{-1}$ to 1 to 50 mm $year^{-1}$) resulting from clearance of native deep-rooted vegetation (able to remove over 99% of infiltration) and replacement with rain-fed shallow-rooted crops and pasture (Leaney et al. 2003). Salt bulges, accumulated naturally in soil and unsaturated zone in the last 20,000 years, are being mobilized by this human-induced recharge. In the irrigated areas, recharge can be further augmented one order of magnitude, and there is an additional risk from rising saline water tables due to irrigation accessions (Leaney et al. 2003; CSIRO 2008). Enhanced recharge through changed landuse and irrigation increases the hydraulic head (groundwater mounds associated with irrigation districts have been detected) on the underlying saline aquifers (Parilla, Shepparton) forcing flow and this salt towards discharge sites either inland or into the Murray River (CSIRO 2008). However, the presence of thick unsaturated zone or a shallow aquitard may result in slowed-down leakage from irrigation to the underlying saline aquifers, delaying also the impact of irrigation on the occurrence of river salinity (Evans 2013). In order to prevent saline groundwater from discharging to the river, salt/water trade was established and salt interception schemes (pumping boreholes) have been built along the southern parts of the Murray River.

Irrigation water is mainly sourced from surface water diversions. However, fresh groundwater is increasingly used for irrigation, especially in times of drought (CSIRO 2008). Groundwater abstractions vary considerably between areas of intensive extraction for irrigation (e.g. pastures, intensive horticulture, rice) to areas of broad scale stock and domestic use. Irrigation abstractions are mostly derived from the basal and the intermediate good-quality aquifer units (especially the Murray Limestone; Cartwright et al. 2010). In some cases, this has induced drainage from shallow saline aquifers groundwater to deeper good-quality aquifers, resulting in groundwater quality deterioration. However, where aquitards are thicker this saline drainage can be delayed for decades (Leaney et al. 2003).

Souf Valley, SE Algeria

The Souf Valley (SV) is located at the northern fringe of the Saharan Platform (Fig. 5a; Table 1). It extends over a plain area with no outlet in the North Western Sahara Aquifer System (NWSAS). The Quaternary phreatic aquifer of the Souf is mostly sandy (sand dunes). NWSAS is located in the large northern Sahara sedimentary basin and overlies two deeper confined aquifers: the Complexe Terminal (CT) and the underlying Continental Intercalaire (CI), one of the largest confined aquifers in the world, comparable in scale to the Great Artesian Basin of Australia (Djabri et al. 2010). Shallow groundwater in the Souf is salinized with a 2,000–10,000 mg/L TDS. Predominant chemical types are sodium sulfate to sodium chloride. Nitrate content is high due to agricultural activity and to untreated domestic sewage. Isotope signatures of this aquifer indicate evaporative enrichment and the presence of evaporite formations, and the tritium content indicates a recent recharge by precipitation (Guendouz et al. 2006). Salt contents of groundwater from the CI and CT range between 1,000 to 4,000 mg/L TDS.

The SV illustrates the combined effects of saline IRF and rising water tables on groundwater quality and crop yield (reduction by salinization and asphyxiation) in a groundwater-fed intensive agricultural area. The economy of the region is based mainly on the cultivation of date-palm trees planted in the traditional Ghout system (man-made craters, in between dunes, of about 10 m depth and 80–200 m diameter enclosing 20–100 trees), which allow the tree roots to tap the underlying water table (typically at 1 m depth; Remini 2006) (Fig. 5a). Between 1990 and 2000, these crops occupied an area of 9,500 ha, with around 10,000 Ghouts. This system of cultivation is well adapted to the erg (dune sea) environment but it is a fragile system because it is very dependent on the water level. Prior to the 1970s, water supply and irrigation relied on hand dug wells and springs. Early surveys in the 1950s indicated a water level decline trend (Guendouz et al. 2006). Since the 1970s, a number of deep boreholes tapping the underlying confined aquifers have been drilled. Water supply and irrigation abstractions soared fuelled by a strong population growth. This has resulted in increased saline IRFs, eventually leading to a rising water table and to the flooding of some Ghouts, which in turn has accelerated groundwater salinization rates (Fig. 5b). In 1994, the number of flooded Ghouts was about 500. This resulted in a loss of more than 150,000 date-palm trees by asphyxiation. In 2002, the number of flooded Ghouts rose to 950, and about 2,100 were wet (6,547 remained dry), with 231,540 date-palm trees affected out of 742,525 (Oeltzschner 2002). In addition, inappropriate disposal of untreated urban wastewater was contributing to groundwater salinization (approximately 100,000 cesspits existed in 2007; Meziani et al. 2009).

[Please Insert Fig. 5 about here]

Currently, dewatering is achieved by means of vertical drainage via a network of wells into the aquifer equipped with pumps. In addition, a series of drains comes into play when the water rises to their level. This water, along with wastewater from recently installed sewers, is fed into a lagoon system for purification. Infilling the flooded Ghouts is not a viable option for reversing the phenomenon, though this practice can limit proliferation of mosquitoes, prevent waste dumping in urban areas and reduce direct evaporation.

Costa de Hermosillo aquifer, NW Mexico

The Costa de Hermosillo (CH) coastal aquifer is located in the Gulf of California, towards which surface waters within the basin drain (Fig. 6a; Table 1). Exploitation of the aquifer began in 1945 (17 wells for irrigation). The peak abstraction volume was reached in 1965, with more than 900 wells pumping over

1,100 hm^3/year (Fig. 6b). The land under irrigation reached 130,000 ha, with a withdrawal around 500 hm^3/year, though in 2002 some 45,000 ha were irrigated (CNA 2003). Nowadays, groundwater abstractions have reduced to an estimated amount of 350 hm^3/year – a figure that is still far higher than the estimated mean aquifer recharge of about 100 hm^3/year.

[Please Insert Fig. 6 about here]

Decline in water levels (62 m in 2003) changed the original hydraulic head distribution, producing a drawdown cone (noticeable since 1949) which caused an inversion of hydraulic gradient and drew groundwater into the centre of the plain. This resulted in the mobilization of saline groundwater towards the central area of the aquifer. Evaporation of irrigation water prior to infiltration is substantial due to arid to semi-arid conditions. Thus, recharge from saline IRFs also contributes to aquifer salinization. In addition, widespread use of fertilizers has resulted in elevated concentrations of nitrates. Recent studies indicate that despite seawater lateral intrusion is detected in boreholes close to the coastline, upconing of basinal connate water (Miocene/Pliocene transgression) is probably the dominant salinization process of the aquifer (Szynkiewicz et al. 2008). Anyhow, aquifer salinization has led to the abandonment of many wells located in the strip of land that extends from the coast to some 25 km inland. Some of the pumping boreholes have been relocated to the northern part of the aquifer, transferring the problem there.

South East Spain aquifers

Many aquifers of the Mediterranean region show evidence of salinization due to agricultural abstractions and, to a lesser extent, to urban supply. Five paradigmatic case studies in SE Spain illustrate this issue (Fig. 7a). The climate of this region is semi-arid with Mediterranean characteristics (Table 1). Main source of irrigation water is groundwater. Surface water from reservoirs and limited transfers from other river basins (e.g. Tajo) are secondary water sources.

[Please Insert Fig. 7 about here]

The Campo de Dalias (CD) aquifer system (Almeria) supplies one of the main greenhouse crop areas of Europe (Fig. 7; Table 1). In the last decades, groundwater abstractions have fuelled an intensive agricultural boom in the area. This over-exploited multilayer complex aquifer includes an upper detrital unit and two deeper carbonate units (Fig. 7b). Early boreholes tapped the shallow aquifer, but progressively deeper units were tapped as phreatic groundwater quality gradually declined because of increasing recharge from saline IRFs. These resulted from extensive application of fertilizers and principally from salt concentration by evapotranspiration. Nowadays, the shallow aquifer water is saline (1,000–4,200 mg/L TDS) and nitrate levels are high (200–400 mg/L; Fig. 7c). Deep boreholes penetrating low-permeability formations in the superficial layers tap deep units and yield fresh water with low nitrate concentrations. However, intermediate nitrate concentrations are found in some boreholes at intermediate depth, suggesting drainage of the shallow aquifer downwards to deep over-exploited units (Pulido-Bosch et al. 2000) (Fig. 7c). Excessive irrigation abstractions have induced seawater intrusion in two sectors and have also mobilized connate water existing at the bottom of the detrital unit accelerating the salinization process. High salt contents of phreatic groundwater led to the abandonment of numerous wells (most of them replaced by deeper boreholes). As a result, a drastic decline in shallow groundwater abstractions occurred (from 45 to 10 hm^3/year). This reduction eventually resulted in a remarkable rise of the water levels, leading to local waterlogging and flooding of some lowlands (Daniele et al. 2008).

The Sierra de Crevillente (SC) carbonate aquifer (Alicante; Fig. 7a; Table 1) has been intensely exploited for irrigation since the 1960s (9,000 ha of table grapes, fruit trees and vegetables). In 1964, a gallery and twelve shallow boreholes were drilled, but these had to be extended to 300 m depth over the years. In 1980 abstractions reached 18 hm^3/year, but declined to 4 hm^3/year after 1997 (Andreu et al. 2008). Subsequently, exploitation extended to other sectors of the aquifer. Recharge of the aquifer has been estimated as 10 hm^3/year. About 40 hm^3/year of groundwater were pumped in some years. As a result, water levels declined drastically (300 m in the Tolomo sector) (Fig. 7d). Boreholes were deepened or abandoned due to reduced yields or salinization. The main impact on the groundwater quality was a gradual increase in salinity as the aquifer depletion progressed. Excessive irrigation abstraction mobilized saline water from bottom clay and gypsum strata (Triassic Keuper layers). Increased abstraction costs and poor groundwater quality have reduced crop profitability drastically since year 2000.

The Alto Guadalentin (AG) aquifer (Murcia; Fig. 7a; Table 1) has been intensively exploited for irrigation since the 1960s. Excessive irrigation abstractions resulted in a water level decline of 195 m from 1973 to 2005 (Rodriguez-Estrella 2014). As aquifer depletion progressed salinity raised, partially due to a marked increase in bicarbonate contents (from 300 mg/L in the 1960s, through 800 mg/L in 1986, to 1,800 mg/L in 1987). Pumped groundwater gas contents also increased (CO_2 with small proportions of N_2, O_2,

CH_2, H_2 and He), particularly since 1983 (decline of 140 m; Fig. 7e). This was attributed to the decrease in hydrostatic pressure resulting from the considerable water level decline. Thus, deep saline waters were propelled upwards by the action of geogenic CO_2 both through faults and boreholes penetrating the metamorphic substratum (Ceron and Pulido-Bosch 1996). Currently, bicarbonate contents range between 495–1,890 mg/L and salinity levels are between 900–3,200 mg/L TDS. Water type is calcium–magnesium sulfate–carbonate–chloride and often sodium. Recharge from saline IRFs resulting from evapotranspiration and mobilization of salts in soil and salt flats existing in the area (e.g. El Saladar) has also contributed to aquifer salinization. However, the mobilization of sulfate–chloride salts existing in deep basinal Miocene strata resulting from irrigation over-abstraction has been identified as the dominant quality deterioration process of the aquifer.

Groundwater quality in SE Spain have significantly deteriorated in the last three decades, so that pumped groundwater from some aquifers is currently no longer suitable for irrigation of most crops. This is the case of the Campo de Cartagena (CC; Murcia) and the Campo de Nijar (CN; Almeria) over-exploited coastal detrital aquifer systems (Fig. 7a; Table 1), where traditional and intensive agriculture are highly developed. Both aquifers are affected by recharge of evapoconcentrated saline IRF, seawater intrusion and old saline water upconing. In order to reduce salt in irrigation water, small private modular brackish water desalination plants were installed by farmers (over 1,000 in the Campo de Cartagena aquifer). Each of these early private plants pumped groundwater from nearby wells (30 m^3/h on average) and was normally used only for a particular farm. However, elimination of wastewater (approximately 25% of input flow) was not appropriately planned and reject brines were discharged into ditches cut into the ground or injected into nearby wells not used for abstraction, allowing saltwater to infiltrate back into the aquifer and increase groundwater salinity. In many farms, brines eventually reached the influence zone of nearby abstraction boreholes and wells. In the Campo de Nijar aquifer an “à la carte” approach to irrigation water use is being increasingly utilized to allow for augmented profits (Miguel et al. 2011). In this approach, minimum facilities are employed to continuously mix adequate proportions of raw pumped groundwater and desalinized groundwater according to demand in order to produce irrigation water with the desired salinity level. Inappropriate agricultural waste disposal has also negatively affected groundwater quality of many aquifers of the region. For instance, in the Almeria province, about one and a half million tonnes of plant waste, thirty thousand tonnes of plastic waste and six thousand tonnes of diverse wastes are generated each year (Callejon and Lopez-Martinez 2009). Prior to the 1990s, crop residues were not recycled but burnt or stored directly on land, often leading to percolation of saline leachates with high organic contents. Nowadays, a significant proportion of plastic waste is recycled and plant remnants are mostly recycled as compost.

Conclusions

Irrigation is indispensable for maintaining global food production at current rates. Population projections for the next decades indicate that more high-yielding irrigated land will be needed for crop production. Consequently, the amount of water used by agriculture will also increase, imposing further pressure on available water resources, especially in arid and semi-arid areas. However, without correct planning and management, irrigated agriculture can have adverse effects on water resources and lead to the depletion and deterioration of aquifers and consequential social and economic loss. In this regard, salinization is the main groundwater quality problem resulting from irrigation. Agricultural activities can affect aquifer salinity directly and indirectly. Thus, salinity alterations resulting from the application of irrigation water can be classed as direct impacts, and those from irrigation abstractions as indirect impacts. Although evidence of these impacts can be found in aquifers all over the world, two particularly vulnerable domains can be identified, namely arid and semi-arid areas (where salinization is an inherent part of irrigation) and coastal zones. Hence, arid/semi-arid coastal zones are particularly prone to serious problems of salinization.

Case studies illustrate how water quality of productive aquifers around the world can deteriorate by salinization resulting from poor irrigation planning and practices. Case studies describe a range of different impacts of irrigation on groundwater salinity (Table 1). In many cases, saline groundwater of marine or natural terrestrial origin is also involved (category C and D groundwaters). Typically, several salinization processes are superimposed. Irrigation induced groundwater salinization by evapoconcentration is ubiquitous in the arid and semi-arid zone and is detected in all case studies. This salinization process is typically reinforced by several other processes leading to increased groundwater salinization. In addition, as noted above, many aquifers in arid and semi-arid zones are grossly over-exploited. In groundwater-fed irrigated systems, groundwater salinization usually results from the combined side-effects of the intense abstraction of fresh irrigation groundwater from the aquifer and the recharge from saline IRF to the aquifer, i.e. the combination of direct and indirect impacts. These two salinization mechanisms usually strengthen each other (a reduced fresh groundwater volume receiving salt; Smedema and Shiati 2002) and usually lead to a progressive increase in groundwater salinity. Thus, the observed groundwater salinization is often due to a combination of increased saline recharge and aquifer depletion. In some cases, aquifer salinization rates are further increased by irrigation induced mobilization of stored salts. In the MB stored salts mobilized by IRF add to shallow naturally saline groundwater (river salinization by groundwater seepage is of great concern). Thus, good quality deep aquifer units are being affected by (delayed) downward leakage of shallow saline groundwater induced by head differences due to increasing irrigation abstractions. In the SV, both shallow and deep groundwater is saline. However, accelerated groundwater salinization was triggered by the change of the main irrigation water source from shallow phreatic (hand dug wells) to more productive deep confined aquifer units. This has resulted in a marked development of irrigation and has accelerated the ongoing phreatic groundwater salinization processes, thus leading to rising water tables and waterlogging. In the CD, conversely to the SV, IRF induced shallow groundwater salinization led to the exploitation of deeper good quality aquifer units. Since then, high economic return of intensive agriculture has led to over-exploitation of the deep aquifer units resulting in induced seawater intrusion, connate water upconing and shallow saline water downward leakage along with rising water levels and waterlogging. In the CH, SC and the AG aquifers, intense irrigation abstractions led to a dramatic decrease in water levels (62, 300 and 195 m, respectively) resulting in a significant reduction of the assimilative capacity of the aquifers and the mobilization of saline waters. In the case of the AG aquifer, this process was reinforced by gas ascent by aquifer decompression, evapoconcentration and mobilization of stored salts. In the CC and CN aquifers, intense agricultural development has led to groundwater salinization resulting from recharge of evapoconcentrated saline IRF, seawater intrusion and old saline water upconing induced by over-abstraction as well as inappropriate disposal of agricultural waste and wastewater from desalination plants. In many cases, fertilizer over-application has also contributed to groundwater salinity.

The existing paradigm for irrigated agriculture of focusing mainly on crop production increases has led to widespread salinization of groundwater resources. This is graphically illustrated by the cases involving multilayer aquifers, where salinization of shallow aquifer units resulting from irrigation practices is typically succeeded by intense (and often unregulated) exploitation of deeper good quality units and their consequential depletion and salinization, and a further worsening of shallow groundwater quality. In this regard, management strategies for controlling cropland salinization have typically concentrated on ensuring plant uptake and low salt levels within the root zone (e.g. by leaching, drainage, flooding), but have often paid less attention to adequately protect aquifers from salinity and sometimes this has been considered as an inevitable result of agricultural irrigation development. Consequently, a shift to a paradigm where sustainability of groundwater resources is an essential component must occur in order for it to support sustainable agricultural development. Basically, the mitigation of groundwater salinization may concentrate on minimizing recharging IRF volumes and/or salinity, and, in the case of groundwater-fed irrigated systems, balancing irrigation abstractions. A number of management measures for groundwater salinity control in irrigated areas have been proposed including appropriate design and planning (including careful consideration of suitable land uses in recharge areas), implementation of strict soil-surface-groundwater salinity monitoring programs, improved irrigation water use efficiency, interbasin water transfers, conjunctive use of groundwater and surface water, diverting of saline drainage out of the basin (e.g. construction of outfall drains), and strict regulation of groundwater withdrawals (in some cases, irrigated land and/or pumping wells may need to be abandoned) (Suarez 1989; Smedema and Shiati 2002; Scanlon et al. 2005, 2007; Duncan et al. 2008).

It is clear from the case studies that irrigation induced soil, surface, and groundwater salinization are inextricably linked and form a feedback loop. In this regard, despite the considerable progress achieved in recent years, further efforts are needed to improve our understanding of the variety of processes involved, their interactions, linkages and long-term consequences. Consequently, integrated and coordinated investigation of these impacts is essential for the correct implementation of appropriate mitigation measures.

References

Andreu JM, Pulido-Bosch A, Llamas MR, Bru C, Martinez-Santos P,Garcia-Sanchez E,. Villacampa L (2008) Overexploitation and water quality in the Crevillente aquifer (Alicante, SE Spain). In: Prats-Rico D, Brebbia CA, Villacampa-Esteve Y (eds.) Water Pollution IX, WIT Transactions on Ecology and the Environment 111, WIT Press, Southampton. pp 75–84

AQUASTAT (2016) FAO Information System on Water and Agriculture. Land and Water Division, Food and Agriculture Organization of the United Nations. http://www.fao.org/nr/water/aquastat/main/. Accessed 10 November 2016

Aragues R, Tanji KK (2003) Water quality of irrigation return flows. In: Stewart BA, Howell TA (eds) Encyclopaedia of Water Science. Marcel Dekker, New York, pp 502–506

Baghvand, A., T. Nasrabadi, N. G. Bidhendi, A. Vosoogh, A. Karbassi, and N. Mehrdadi. 2010. Groundwater quality degradation of an aquifer in Iran central desert. Desalination 260: 264–275.

Barlow PM, Reichard EG (2010) Saltwater intrusion in coastal regions of North America. Hydrogeology Journal 18:247–260

Bohlke JK (2002) Groundwater recharge and agricultural contamination. Hydrogeology Journal 10:153–179

Callejon AJ, Lopez-Martinez JA (2009) Briquettes of plants remains from the greenhouses of Almeria (Spain). Spanish Journal of Agricultural Research 7:525–534

Cartwright I, Weaver TR, Simmons CT, Fifield LK, Lawrence CR, Chisari R, Varley S (2010) Physical hydrogeology and environmental isotopes to constrain the age, origins, and stability of a low-salinity groundwater lens formed by periodic river recharge: Murray Basin, Australia. Journal of Hydrology 380:203–221

Ceron JC, Pulido-Bosch A (1996) Groundwater problems resulting from CO_2 pollution and overexploitation in Alto Guadalentin aquifer (Murcia, Spain). Environmental Geology 28:223–228

CNA (2003) Costa de Hermosillo Aquifer Management Plan. Gerencia Regional Noroeste, Comision Nacional del Agua (CNA), Mexico. Unpublished report (in Spanish)

CSIRO (2008) Water availability in the Murray. A report to the Australian Government from the CSIRO Murray-Darling Basin Sustainable Yields Project. CSIRO, Australia

Daniele L, Pulido-Bosch A, Vallejos A, Molina L (2008) Geostatistical analysis to identify hydrogeochemical processes in complex aquifers: a case study (Aguadulce unit, Almeria, SE Spain). Ambio 37: 249–53

Djabri, L, Hani A, Djouama MC, Bouhsina SJ, Mudry J, Pulido-Bosch A (2010) Transboundary Resources and Good Neighbourhood: Case of joint management of fossil water layer in the South. UNESCO-IAH-UNEP Conference, Paris

Duncan RA, Bethune MG, Thayalakumaran T, Christen EW, McMahon TA (2008) Management of salt mobilisation in the irrigated landscape – A review of selected irrigation regions. Journal of Hydrology 351:238–252

Evans R (2013) Geology and hydrogeology. Mallee Salinity Workshop 2012. Mallee Catchment Management Authority, Mildura

FAO (2011) The state of the world's land and water resources for food and agriculture – Managing systems at risk. Food and Agriculture Organization of the United Nations and Earthscan, Rome

FAOSTAT (2016) FAO Food and Agriculture Database. Statistics Division, Food and Agriculture Organization of the United Nations. http://faostat.fao.org/. Accessed 25 November 2016

Faunt CC, Sneed M, Traum J, Brandt JT (2016) Water availability and land subsidence in the Central Valley, California, USA. Hydrogeology Journal 24: 675–684

Foster SSD, Perry CJ (2010) Improving groundwater resource accounting in irrigated areas: a prerequisite for promoting sustainable use. Hydrogeology Journal 18: 291–294

Freeze RA, Cherry JA (1979) Groundwater. Prentice-Hall, Englewood Cliffs, New Jersey

Garcia-Garizabal I, Causape J (2010) Influence of irrigation water management on the quantity and quality of irrigation return flows. Journal of Hydrology 385:36–43

Ghassemi F, Jakeman AJ, Nix HA (1995) Salinisation of Land and Water Resources: Human Causes, Extent, Management and Case Studies. CAB International, Wallingford

Guendouz A, Moulla AS, Remini B, Michelot JL (2006) Hydrochemical and isotopic behaviour of a Saharan phreatic aquifer suffering severe natural and anthropic constraints (case of Oued-Souf region, Algeria). Hydrogeology Journal 14:955–968

Han D, Song X, Currell MJ, Cao G, Zhang Y, Kang Y (2011) A survey of groundwater levels and hydrogeochemistry in irrigated fields in the Karamay Agricultural Development Area, northwest China: implications for soil and groundwater salinity resulting from surface water transfer for irrigation. Journal of Hydrology 405:217–234

Januel Y (2010) In the context of a new agricultural dynamics, what advantages of the traditional system of Ghouts in relation to the evolved oasis system? Center for International Development Studies and Research. http://www.fao.org/3/a-bp888f.pdf (in French). Accessed 11 March 2016

Kass A, Gavrieli I, Yechieli Y, Vengosh YA, Starinsky A (2005) The impact of freshwater and wastewater irrigation on the chemistry of shallow groundwater: a case study from the Israeli Coastal Aquifer. Journal of Hydrology 300:314-331

Konikow LF, Kendy E (2005) Groundwater depletion: A global problem. Hydrogeology Journal 13:317–320

Kurtzman D, Baram S, Dahan O (2016) Soil–aquifer phenomena affecting groundwater under vertisols: a review. Hydrological Earth Systems Science 20:1–12

Leaney FW, Herczeg AL, Walker GR (2003) Salinization of a fresh palaeo-ground water resource by enhanced recharge. Ground Water 41:84–92

Lorite-Herrera M, Jimenez-Espinosa M, Jimenez-Millan J, Hiscock KM (2008) Integrated hydrochemical assessment of the Quaternary alluvial aquifer of the Guadalquivir River, southern Spain. Applied Geochemistry 23:2040–50

Maas EV (1986) Salt tolerance of plants. Applied Agriculture Research 1:12–26

Mateo-Sagasta J, Burke J (2010) Agriculture and water quality interactions: A global overview. SOLAW Background Thematic Report TR08, Food and Agriculture Organization of the United Nations, Rome. http://www.fao.org/nr/solaw/. Accessed 16 September 2016

McMahon PB, Dennehy KF, Bruce BW, Bohlke JK, Michel RL, Gurdak JJ, Hurlbut DB (2006) Storage and transit time of chemicals in thick unsaturated zones under rangeland and irrigated cropland, High Plains, United States. Water Resources Research 42:W03413

Merchan D, Auque LF, Acero P, Gimeno MJ, Causape J (2015) Geochemical processes controlling water salinization in an irrigated basin in Spain: Identification of natural and anthropogenic influence. Science of The Total Environment 502:330–343

Meziani A, Dridi H, Kalla M (2009) The rise of deep waters in the Souf – Algerian Sahara. Proceedings of the International Symposium Energy, Climate Change and Sustainable Development, Djerba. pp 5–6 (in French)

Miguel J, Lopez-Segura J, Baena R (2011) Use of desalinated water in agriculture. Experiences in Almeria (Spain). IDA World Congress on Desalination Solutions, Perth

Milnes E, Renard P (2004) The problem of salt recycling and seawater intrusion in coastal irrigated plains: an example from the Kiti aquifer (Southern Cyprus). Journal of Hydrology 288:327-343

Mohamed AMO, Maraqa M, Al Handhaly J (2005) Impact of land disposal of reject brine from desalination plants on soil and groundwater. Desalination 182:411–433

Molina L, Vallejos A, Pulido-Bosch A, Sanchez-Martos F (2002) Water temperature and conductivity variability as indicators of groundwater behaviour in complex systems in the south-east of Spain. Hydrological Processes 16:3365–3378

Morris BL, Lawrence ARL, Chilton PJC,Adams B, Calow RC, Klinck BA (2003) Groundwater and its susceptibility to degradation. Early Warning and Assessment Report Series, RS. 03-3. UNEP, Nairobi

Northey JE, Christen EW, Ayars JE, Jankowski J (2106) Occurrence and measurement of salinity stratification in shallow groundwater in the Murrumbidgee Irrigation Area, south-eastern Australia. Agricultural Water Management 81:23–40

Oeltzschner H (2002) Some remarks on the phenomenon of the rise of the waters in the Souf. Technical and Scientific Meeting on the Quality of Waters of the South, El-Oued, Algeria, pp 65–75 (in French)

Oren O, Yechieli Y, Bohlke JK, Dody A (2004) Contamination of groundwater under cultivated fields in an arid environment, central Arava Valley, Israel. Journal of Hydrology 290:312–328

Pulido-Bosch A, Bensi S, Molina L, Vallejos A, Calaforra JM, Pulido-Leboeuf P (2000) Nitrates as indicators of aquifer interconnection. Application to the Campo de Dalias (SE-Spain). Environmental Geology 39:791–799

Remini B (2006) The disappearance of ghouts in the region of el Oued (Algeria). Larhyss Journal 5:49–62 (in French)

Richter BC, Kreitler CW (1993) Geochemical techniques for identifying sources of ground-water salinization. CRC Press, Boca Raton

Rodell M, Velicogna I, Famiglietti JS (2009) Satellite-based estimates of groundwater depletion in India. Nature 460:999–1002

Rodriguez-Estrella T (2014) The problems of overexploitation of aquifers in semi-arid areas: characteristics and proposals for mitigation. Boletin Geologico y Minero 125:91–109

Scanlon BR, Reedy RC, Stonestrom DA, Prudic DE, Dennehy KF (2005) Impact of land use and land cover change on groundwater recharge and quality in the southwestern USA. Global Change Biology 11:1577–1593

Scanlon BR, Keese KE, Flint AL, Flint LE, Gaye CB, Edmunds WM, Simmers I (2006) Global synthesis of groundwater recharge in semiarid and arid regions. Hydrological Processes 20:3335–3370

Scanlon BR, Jolly I, Sophocleous M, Zhang L (2007) Global impacts of conversions from natural to agricultural ecosystems on water resources: quantity versus quality. Water Resources Research 43:W03437

Scanlon BR, Stonestrom DA, Reedy RC, Leaney FW, Gates J, Cresswell RG (2009) Inventories and mobilization of unsaturated zone sulfate, fluoride and chloride related to land use change in semiarid regions, southwestern United States and Australia. Water Resources Research 45:W00A18

Scanlon BR, Gates JB, Reedy RC, Jackson WA, Bordovsky JP (2010) Effects of irrigated agroecosystems: 2. Quality of soil water and groundwater in the southern High Plains, Texas. Water Resources Research 46:W09538

Shi L, Jiao JJ (2014) Seawater intrusion and coastal aquifer management in China: a review. Environmental Earth Sciences 72:2811–2819

Smedema LK, Shiati K (2002) Irrigation and salinity: a perspective review of the salinity hazards of irrigation development in the arid zone. Irrigation and Drainage Systems 16:161–174

Stigter TY, van Ooijen SPJ, Post VEA, Appelo CAJ, Carvalho Dill AMM (1998) A hydrogeological and hydrochemical explanation of the groundwater composition under irrigated land in a Mediterranean environment, Algarve, Portugal. Journal of Hydrology 208:262–279

Suarez DL (1989) Impact of agricultural practices on groundwater salinity. Agriculture, Ecosystems and Environment 26:215–227

Szynkiewicz A, Medina MR, Modelska M, Monreal R, Pratt LM (2008) Sulfur isotopic study of sulfate in the aquifer of Costa de Hermosillo (Sonora, Mexico) in relation to upward intrusion of saline groundwater, irrigation pumping and land cultivation. Applied Geochemistry 23:2539–2558

Tanji KK, Kielen NC (2002) Agricultural drainage water management in arid and semi-arid areas. FAO Irrigation and Drainage Paper No. 61, Food and Agriculture Organization of the United Nations, Rome

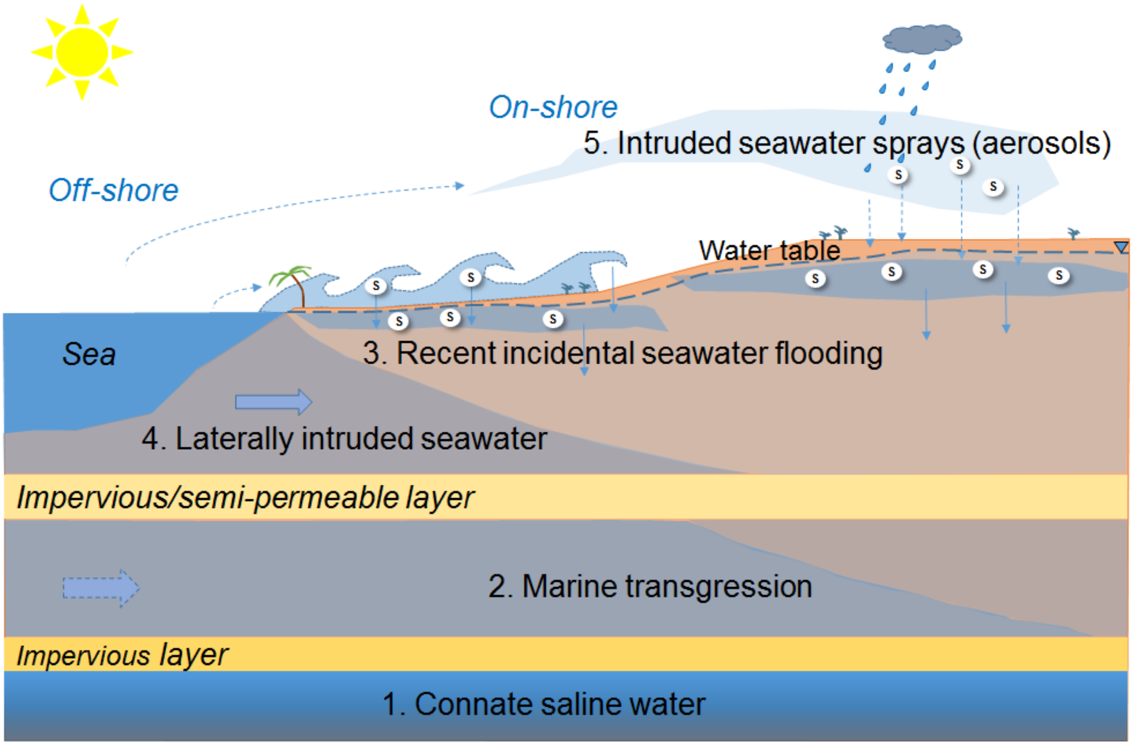
Vallejos A, Andreu JM, Sola F, Pulido-Bosch A (2015) The anthropogenic impact on Mediterranean karst aquifers: cases of some Spanish aquifers. Environmental Earth Sciences 74:185–198

Van Weert F, Van der Gun J, Reckman J (2009) Global overview of saline groundwater occurrence and genesis. Groundwater Resources Assessment Centre, Report GP-2009–1, Utrecht

Wada Y, van Beek LPH, van Kempen CM, Reckman JWTM, Vasak S, Bierkens MFP (2010) Global depletion of groundwater resources. Geophysical Research Letters 37:1–5

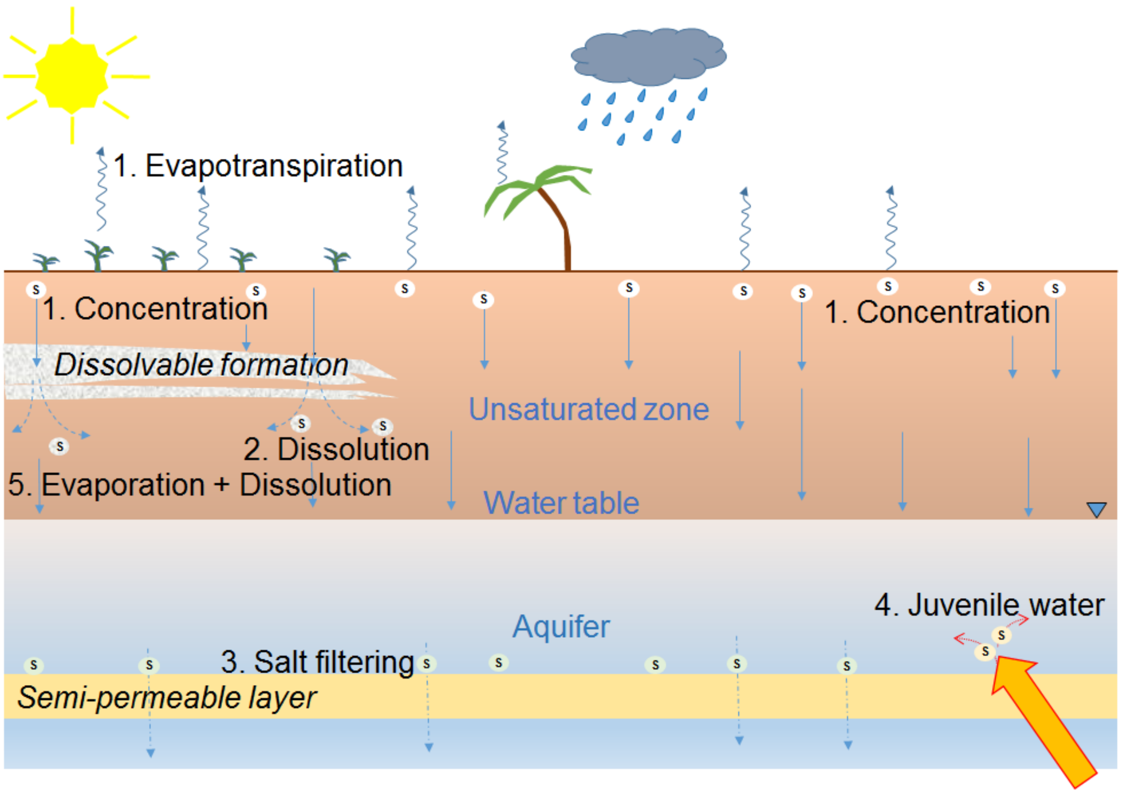
Walvoord MA, Phillips FM, Stonestrom DA, Evans RD, Hartsough PC, Newman BD, Streigl RG (2003) A reservoir of nitrate beneath desert soils. Science 302:1021–1024

FIGURES



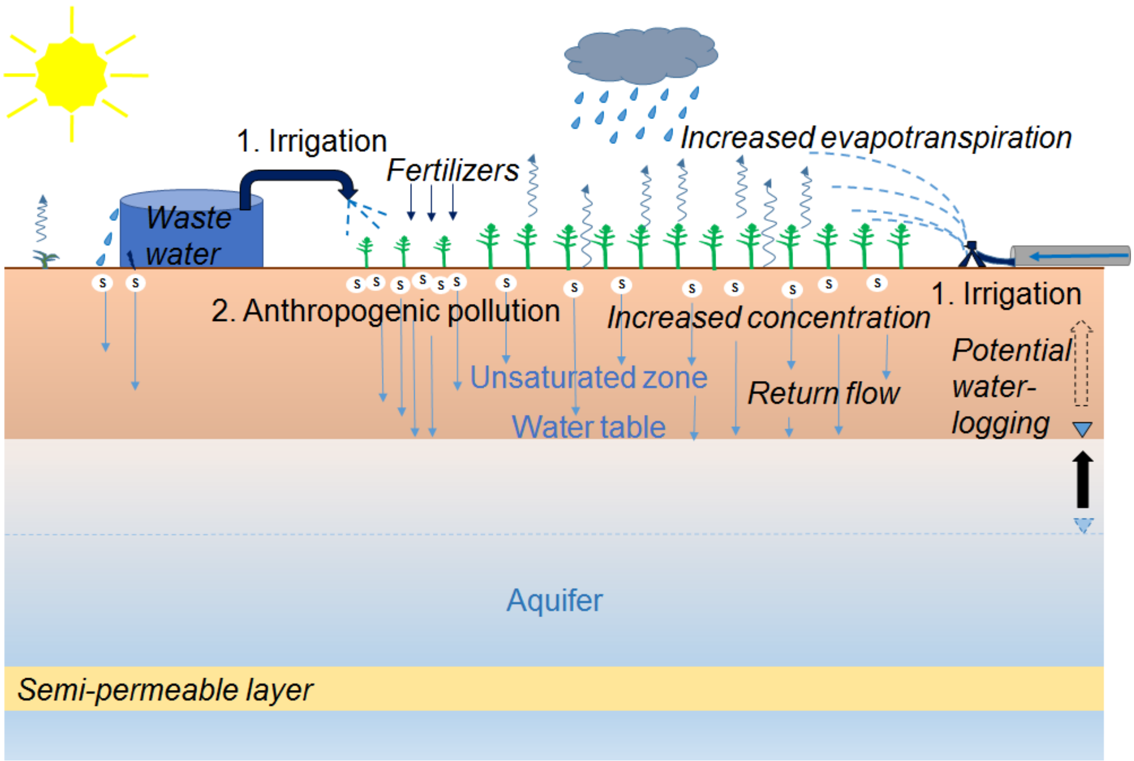
On-shore
5. Intruded seawater sprays (aerosols)
Off-shore
Water table
Sea
3. Recent incidental seawater flooding
4. Laterally intruded seawater
Impervious/semi-permeable layer
2. Marine transgression
Impervious layer
1. Connate saline water

a



1. Evapotranspiration
1. Concentration
Dissolvable formation
1. Concentration
Unsaturated zone
2. Dissolution
5. Evaporation + Dissolution
Water table
4. Juvenile water
Aquifer
3. Salt filtering
Semi-permeable layer

b



c

Fig. 1 Genetic categories of saline groundwater (according to Van Weert et al. 2009): **a** marine origin: (1) Connate saline water; (2) Intruded by marine transgression; (3) Intruded by recent incidental flooding by the sea (e.g. tsunami); (4) Laterally intruded seawater; (5) Intruded seawater sprays (aerosols). Neither category (6), consisting of mixture of (2) and (3), nor category (7), consisting of mixture (1), (2) and (3), are shown; **b** natural terrestrial origin: (1) Produced by evaporation (concentration); (2) Produced by dissolution of subsurface salts; (3) Produced by salt filtering membrane effects; (4) Emanated juvenile water and other products of igneous activity; (5) Mixture of 1 (evaporation) and 2 (dissolution); **c** anthropogenic terrestrial origin: (1) Produced by irrigation (input of concentrated residual water); (2) Anthropogenically polluted groundwater

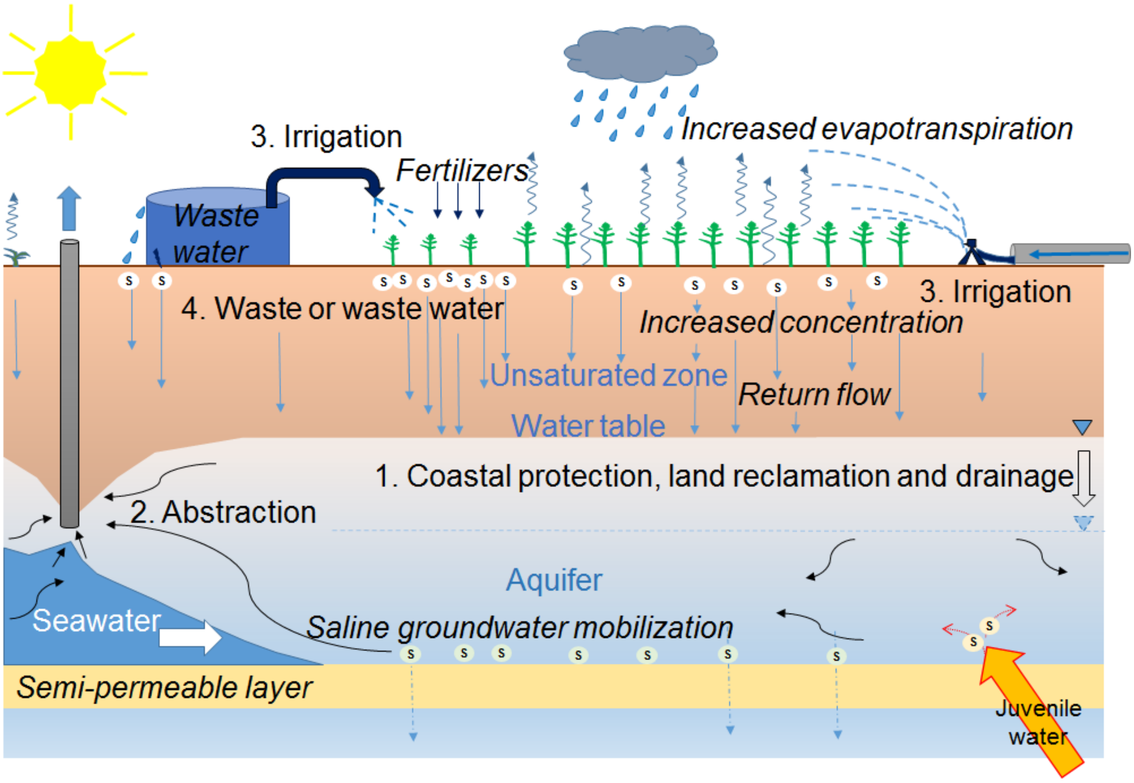


Fig. 2 Anthropogenic drivers affecting groundwater salinity (according to Van Weert et al. 2009): (1) Coastal protection, land reclamation and drainage; (2) Groundwater abstraction; (3) Irrigation; (4) Intentional and unintentional disposal of waste or wastewater.

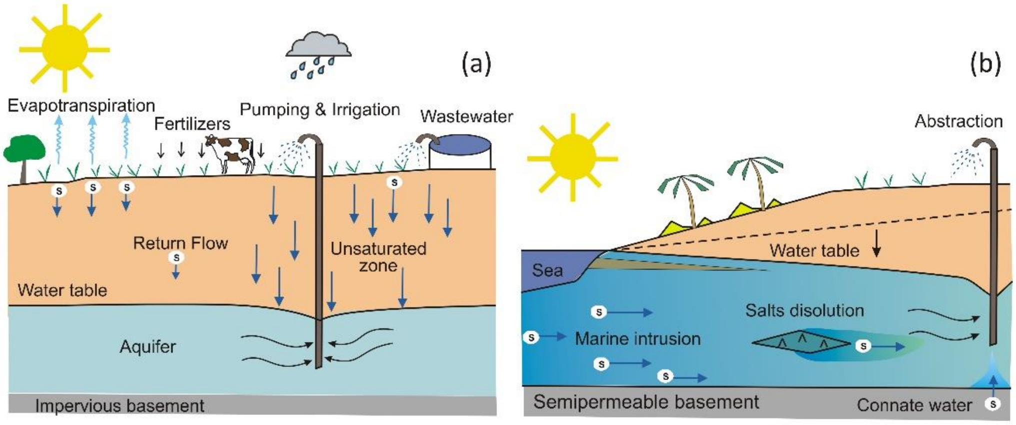


Fig. 3 Main **a** direct and **b** indirect impacts of irrigation on groundwater quality

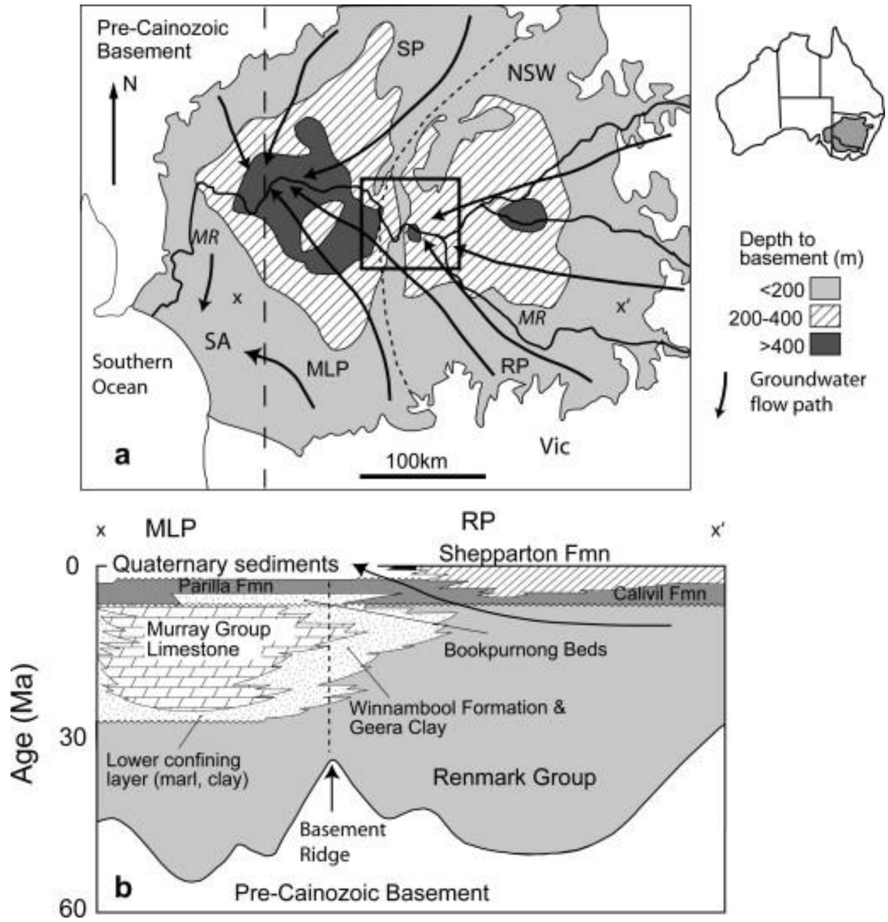
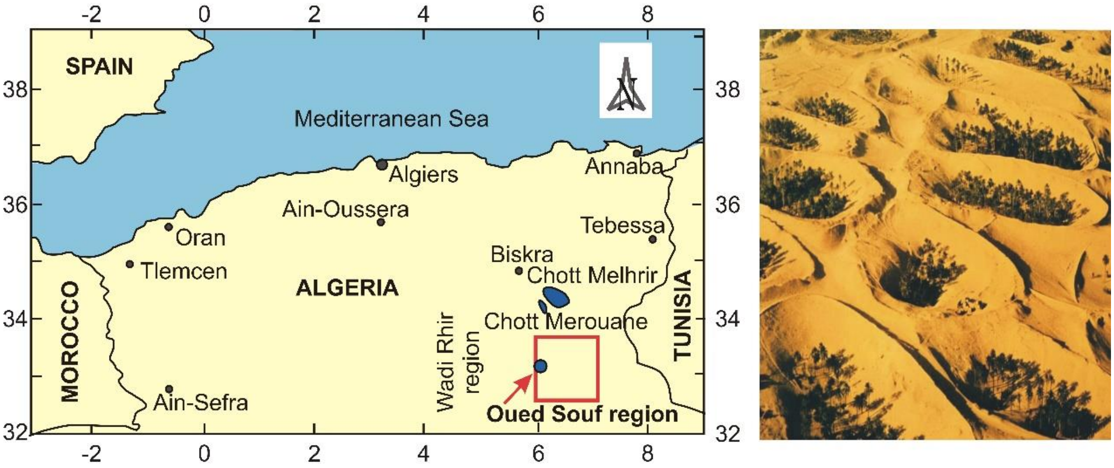
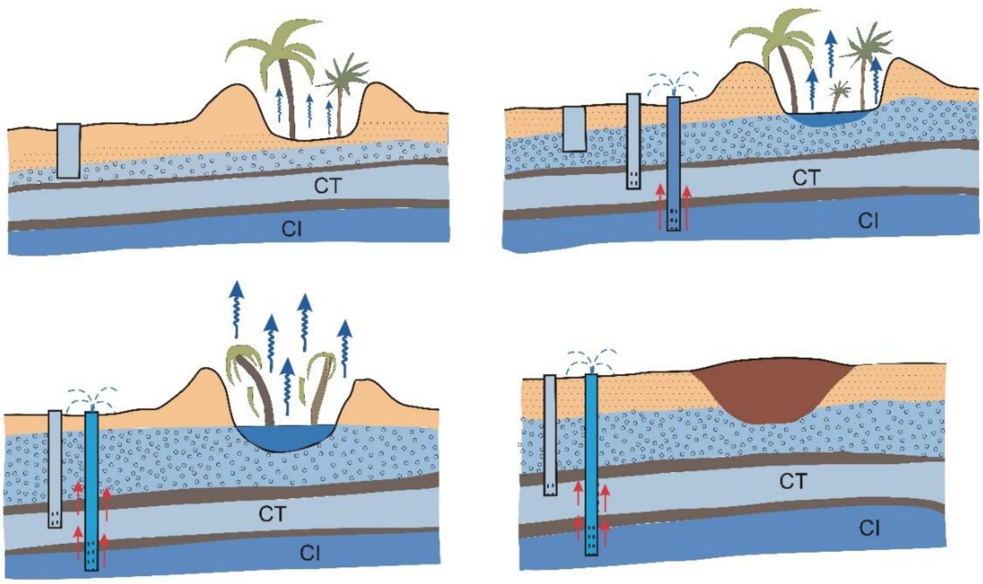


Fig. 4 a Map of the Murray Basin showing depth to basement and groundwater flow paths. NSW = New South Wales, SA = South Australia, Vic = Victoria. MR = Murray River, SP = Scotia Province, MLP = Mallee-Limestone Province, RP = Riverine Province (from Cartwright et al. 2010. Reproduced with permission from Elsevier). **b** Stratigraphic cross-section between x and x′ (Fig. 4a) showing major units in the Murray Basin. Main regional aquifers are: (1) Paleocene to Miocene Renmark Group (confined to semi-confined), (2) Oligocene-Miocene Murray Group Limestone and (3) Pliocene Parilla-Calivil and Shepparton Formations (unconfined to semi-unconfined with delayed drainage). Several Quaternary shallow discontinuous unconfined aquifers (e.g. river paleo-channels, sand dune fields) and a thin regional aquitard also exist (from Cartwright et al. 2010. Reproduced with permission from Elsevier)



a



b

Fig. 5 a Geographical location of the Souf Valley (Argelia) and panoramic image of Ghouts (Januel 2010). **b** Hydrologic evolution of the Ghouts in the area: from left-to-right and top-to-bottom (modified from Remini, 2006). Abbreviations as in text. The phreatic aquifer is shown as dots. Impervious or semi-permeable layers are shown in brown. Not at scale

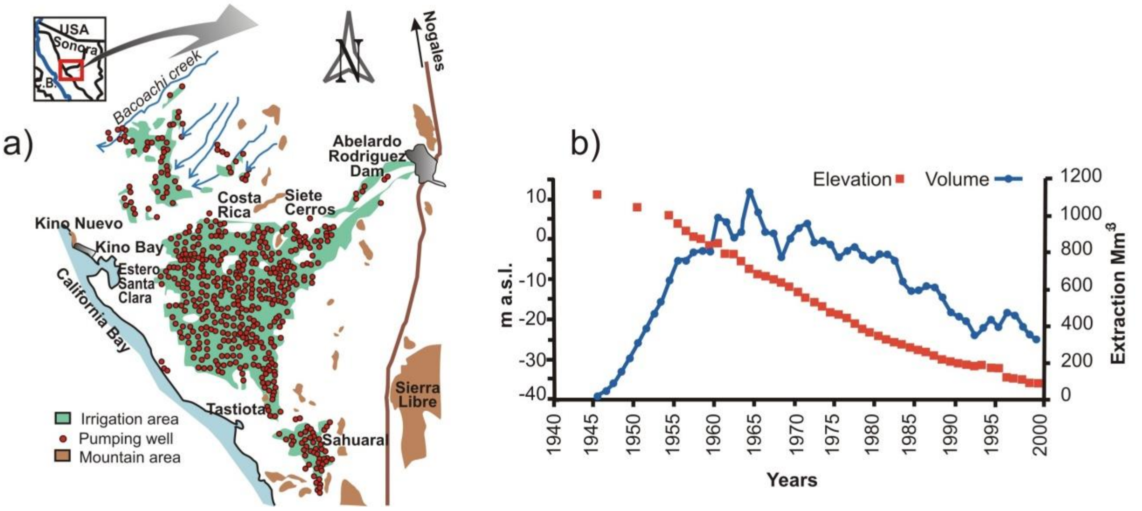


Fig. 6 a Geographical location, irrigated area and pumping wells of the Costa de Hermosillo aquifer. **b** Water levels evolution and water abstraction in the aquifer (CNA 2003)

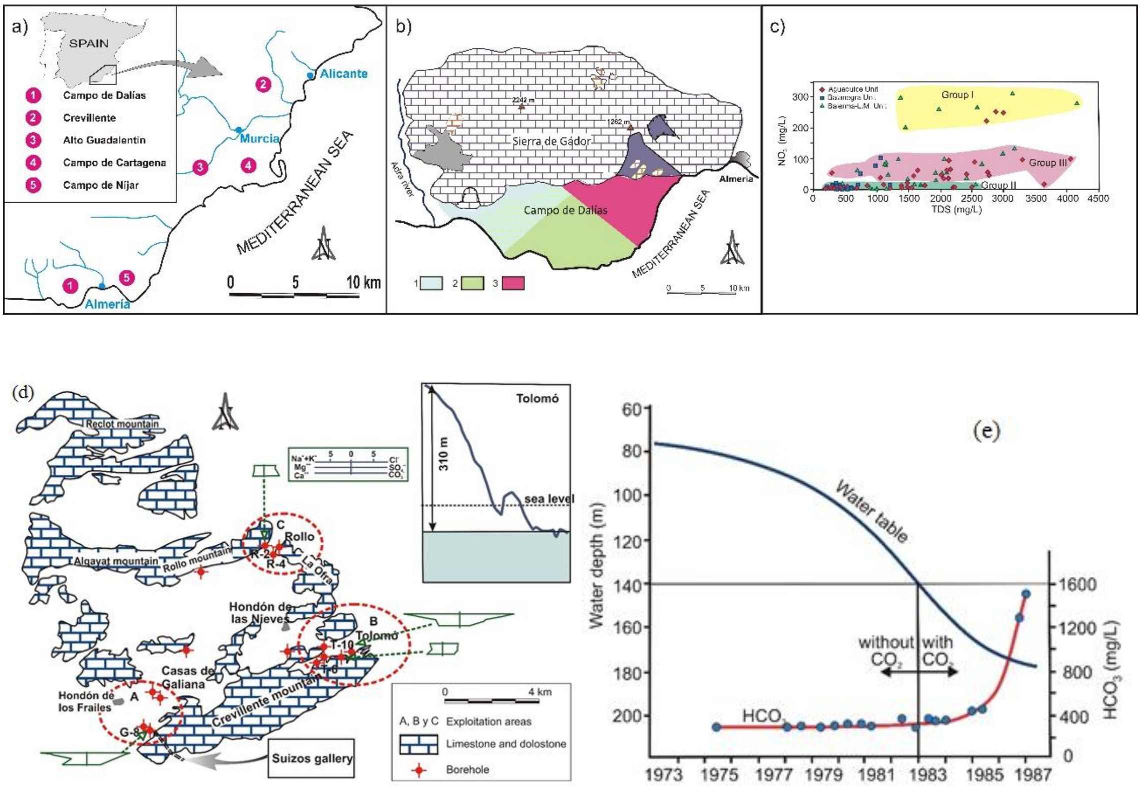


Fig. 7 **a** Location of case studies selected in Spain. **b** Hydrogeological scheme of the Campo de Dalias aquifer. 1: Balanegra Unit, 2: Aguadulce Unit, 3: Balerma-Las Marinas Unit. **c** Relationship between TDS and NO_3^- in the Campo de Dalias aquifer system. **d** Hydrogeological layout of the Sierra de Crevillente aquifer. Areas of largest abstraction are shown in red. Inset shows the rate of groundwater depletion in the Tolomo area (water level is currently below sea level). Stiff diagrams for some wells are also shown. **e** Evolution of water levels and HCO_3^- contents in the Alto Guadalentin aquifer (Rodriguez-Estrella 2014)

TABLES

Table 1 Basic descriptive characteristics and main irrigation-induced salinization processes of the case study aquifer systems. MB: Murray Basin; SV: Souf Valley; CH: Costa de Hermosillo; CD: Campo de Dalias; SC: Sierra de Crevillente; AG: Alto Guadalentin; CC: Campo de Cartagena; CN: Campo de Nijar. M: Multilayered; D: Detrital; K: Karstic. Average thicknesses shown. S: Surface water; G: Groundwater. Climate data as annual means

	MB	SV	CH	CD	SC	AG	CC	CN
Extent (km^2)	300,000	3,000	3,200	300	140	236	1,450	583
Aquifer type	M	M	D	M	K	D	M	D
Inland/Coastal	I	I	C	C	I	I	C	C
Thickness (m)	10	40	500	125	500	300	100	150
	40	300		100			50	
	100	300		800			125	
	300						175	
Main irrigation water source	S	G	G	G	G	G	G	G
Irrigated area (ha)	475,000	9,500	45,000	20,000	9,000	60,000	50,000	9,000
Climate:								
T (ºC)	23	23	23	19	17	18	18	19
P (mm)	400	70	130	300	330	280	270	250
E (mm)	1,700	2,400	1,700	1200	900	1200	1200	1200
Direct impacts								
Concentration by evapotranspiration	x	x	x	x	x	x	x	x
Rising water table and waterlogging	x	x		x				
Stored subsurface salt mobilization	x	x				x		
Fertilizer overapplication		x	x	x				
Agricultural waste and wastewater		x		x			x	x
Indirect impacts								
Seawater lateral intrusion			x	x			x	x
Old saline/connate water upconing			x	x	x		x	x
Leakage of shallower saline groundwater	x			x				