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Long-term artificial seawater irrigation as a sustainable environmental management strategy for abandoned solar salt works: The case study of Agua Amarga salt marsh (SE Spain)

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ABSTRACT

Groundwater abstraction is among the main anthropogenic causes of wetland desiccation worldwide, and corrective measures must be taken to avoid degradation of this valuable ecosystems. A case study is the Agua Amarga salt marsh (≈180 ha) (SE Spain). Agua Amarga includes a solar saltwork pond network in operation between 1925 and 1975, when it was abandoned, and the ponds were colonized by salt marsh vegetation. In 2008 two desalination plants were operating in the marsh vicinity, which were supplied with groundwater. To mitigate the possible negative impact on the salt marsh ecosystem due to groundwater drawdown, in 2009 a sea water irrigation program was implemented. This paper summarizes the results of a ten-year monitoring program (2010–2020) to evaluate the effects of the irrigation program on groundwater levels and quality, soil salinity and moisture, and vegetation cover. During this period, average groundwater level was 2.5 m below the surface and around 1 m deep near the irrigated ponds. Groundwater salinity was not affected outside the saltmarsh, but inside, where the saltworks caused values to rise above 300 mS/cm, it decreased more than 150 mS/cm in some 20 m deep piezometers. Between 2012 and 2020, vegetation cover increased between ≈10 and ≈25 %, with halophyte species such as *Arthrocnemum macrostachyum* and *Sarcocornia fruticosa* being the most favoured. In areas with prolonged flooding, *Ruppia maritima*, a plant species that lives submerged in saline water, was found. In the irrigated areas, soil electrical conductivity (1:5 soil:water extracts) decreased from ≈7-14 mS cm-1 to ≈2-6 mS cm-1. We present an example of sustainable actions in a coastal wetland, where the exploitation of water resources in semiarid areas is combined with promoting natural habitats.

1. Introduction

Wetlands provide many environmental benefits, including aquifer recharge, water purification, temperature control, biological control, flood mitigation, the retention and/or inactivation of harmful substances, the provision of food and shelter for many organisms, biodiversity conservation, and carbon (blue carbon) sequestration (Russi et al., 2013; UNEP, 2014; Neubauer and Verhoeven, 2019; Macreadie et al., 2017; Mitsch and Gosselink, 2015; Reddy and DeLaune, 2008; Owen, 2007; Beck et al., 2001). However, these ecosystems are among the world's most endangered (European Commission, 2002; Hefting et al., 2013).

According to the global assessment report on biodiversity and ecosystem services (Díaz et al, 2019), the world has lost up to 85% of the area occupied by wetlands between 1970 and the present. Thirty-five percent of them have disappeared, and this percentage rises to thirty-nine in the case of coastal wetlands (Darrah et al., 2019; Newton et al., (2020). Among the main human activities that exert direct pressure on coastal wetlands are water extraction, salt extraction, and land-use change, all of which appear in the case study we deal with here.

The value of wetlands and the negative impacts they suffer explain why the number of management actions implemented to preserve/restore these ecosystems increased by 233% between 1970 and 2014 (Darrah et al., 2019). Kaza and BenDor, 2013 and White and Kaplan, 2017 indicated that restoration programs in aquatic ecosystems should be made visible to the public. To guarantee the suitability of any

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intervention/activity developed in wetlands and their surroundings, sustainability principles should be followed (European Commission, 2020; European Environment Agency, 2021). This involves monitoring possible environmental impacts.

The actions carried out in the Agua Amarga salt marsh are an example of sustainable and suitable management practices because they make use of a natural resource (groundwater) compatible with the conservation and improvement of a wetland. Seawater is abstracted from the aquifer below the salt marsh to supply two desalination plants, while the salt marsh is irrigated with seawater to preserve it. This paper describes the wetland irrigation plan established according to groundwater abstraction rates and the results of the evolution of groundwater piezometry and salinity, together with changes in soils and vegetation as key ecosystem elements throughout the ten years of study (2010–2020).

2. Description of the Agua Amarga salt marsh, management of groundwater abstraction and the wetland irrigation plan

The Agua Amarga salt marsh (≈180 ha) is a coastal wetland located in southeastern Spain, near the city of Alicante (Fig. 1). The climate is Mediterranean semiarid with an annual average temperature of 18.2 °C, annual average precipitation of 292 mm, and an annual evapotranspiration rate of 850 mm/year. It occupies a topographically depressed area separated from the Mediterranean Sea by a sand bar ≈2 km long and 50–200 m wide. Based on lithology, the area was originally a natural brackish water lagoon formed by the mix of seawater from regular flooding and the inflow of freshwater during heavy rains. The geological formation is similar to most of the Neogene- Quaternary basins of the Cordilleras Béticas (Galindo-Zaldívar et al., 2019). According to the stratigraphy observed from boreholes, the substratum is composed of Tyrrhenian sandstone containing Quaternary marine fossils covered with fine, coarse-grained alluvial deposits 1 to 5 m thick (Alhama, 2011). The Quaternary-Neogene aquifer is connected to the sea and has an average hydraulic gradient towards the coastal edge of between 0.1 and 1.5%. It is conditioned by a depression cone to the northeast of the saltmarsh, with piezometric depths of between 15 and 5 m, depending on the abstraction rate of the desalination plants (see head contour lines on June 15, 2018; Figure S5). Most of the soils are basically silty clay Solonchak (IUSS, 2007) with an abundance of salt crystals in the matrix.

The wetland is included in the Valencia Community Wetlands Catalogue, subject to special environmental protection, and its coastline is part of the Tabarca Island Marine Reserve, which is a site of community interest (SCI) and a Special Protection Area for Birds (SPA), hosting EU protected habitats. The vegetation on the periphery of the salt marsh, which is not irrigated, mainly consists of halophilous shrubs of *Suaeda vera* and *Limonium* sp, and the steppe grass *Lygeum spartum*. The succulent halophytes *Sarcocornia fruticosa* and *Arthrocnemum macrostachyum* grow in most of the old saline ponds, with stands of *Phragmites australis* colonizing some areas (González-Alcaraz et al., 2014). In ponds with prolonged flooding, patches of the submerged macrophyte *Ruppia maritima* grows from late winter to summer.

Between 1925 and 1975, a solar salt works was installed over the saltmarsh (Box Amorós, 1985). Taking advantage of its flat and depressed geomorphology, the surface was compartmentalized into ponds used as evaporation pans, and a channel to the Mediterranean Sea to allow seawater inflow was built. The exploitation of salt resulted in the salinization of the groundwater (over 400 mS/cm in Abril 2009) and the presence of salt crystals in the first 5 m of soil (Alhama, 2011).

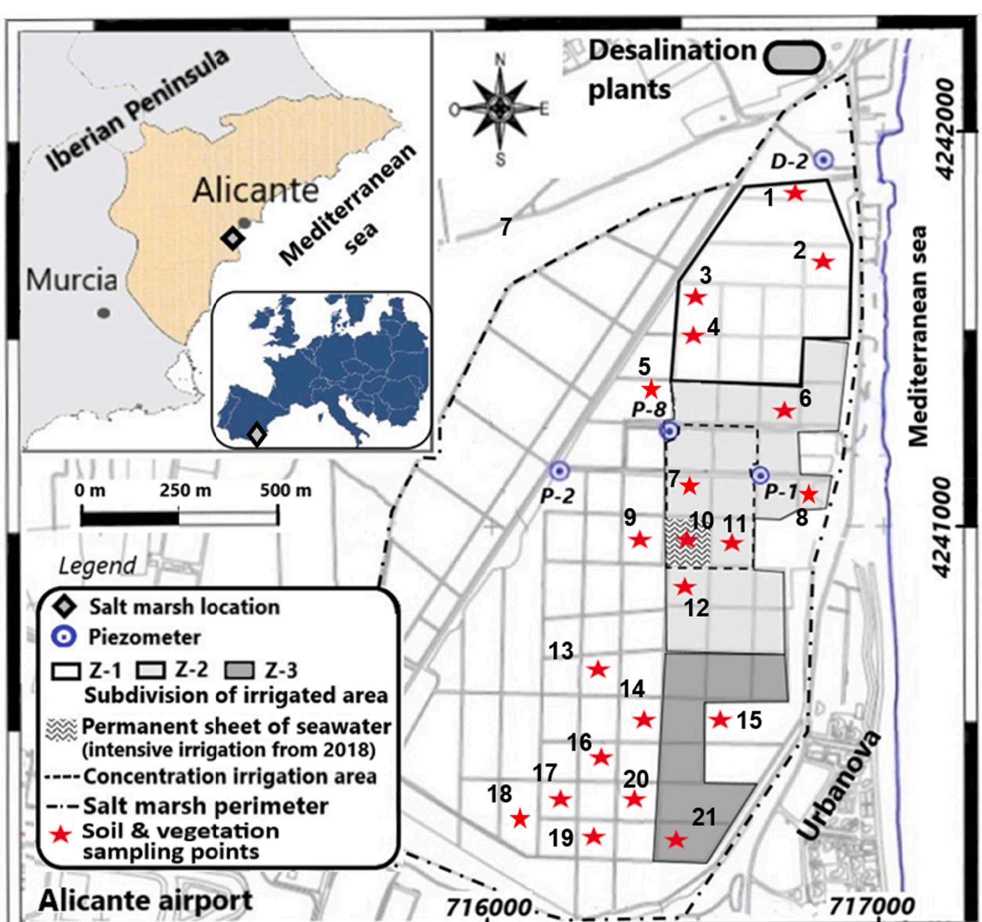

Fig. 1. Location of the Agua Amarga salt marsh and the ponds used for irrigation. Piezometers and sampling plots for soil and vegetation (1–21) are also depicted.

After the closure of the salt works, the connection with the Mediterranean Sea was closed off, and regular flooding with seawater and occasional flooding with freshwater and runoff during rainfall became the norm. In 2004, a national plan to alleviate water deficiency due to persistent drought in the Mediterranean basins (Jefatura del Estado, 2004) promoted the construction of several seawater desalination plants along the coastline (Pulido-Bosch et al., 2019). The Agua Amarga coastal aquifer was chosen as the location for a groundwater abstraction system to supply seawater to the Alicante I and II desalination plants (DA-I and DA-II, respectively). Groundwater catchments systems were an alternative promoted when the hydrogeological connection between the sea and the aquifer was guaranteed along the coastline (Sanz, 2004), thus allowing porous media to act as a natural pre-treatment filter and water temperature stabilizer. DA-I (https://www.mct.es/web/mct/alicante-i) and DA-II (https://www.mct.es/web/mct/alicante-ii) have been operating at full capacity since 2008, with an average production of around 120,000 m^3 d^{-1}, serving populations in the provinces of Alicante and Murcia through the Alicante Channel.

Aimed at minimizing the impact on landscape in the coastline declared SCI and SPA, the DA-II groundwater catchment system consists of a 1 km long and 3.14 m in diameter tunnel with 104 inclined drains drilled from the tunnel, each 25.5 m long and spaced at 9.6 m intervals. The tunnel is located 10 m deep and separated 50 m from the coastline. An additional supply of seawater is guaranteed by 11 horizontal, directional drilling wells under the seabed and through the Tyrrhenian sandstone (Rodríguez-Estrella and Pulido-Bosch, 2009). This arrangement, together with the 18 DA-I vertical pumping wells next to the coastline to the north of the salt marsh, is designed to avoid the effect of saltwater intrusion by creating a hydraulic barrier on the coastline (Stein et al., 2020; Pool and Carrera, 2010), and also to minimize the impact on the piezometry inland. However, considering the possible effects of abstraction on the piezometry of the less permeable Holocene sediments on which the soils of the salt marsh rest, artificial irrigation with seawater was implemented on a part of the surface (Z1, Z2, and Z3 in Fig. 1) using a pipe system (Navarro and Sanchez Lizaso, 2021) so soil moisture could be maintained and piezometric levels partially recovered.

There are other examples in the literature where former salt extraction facilities have been transformed into other land uses, such as fish aquaculture (Cunha et al., 2013) or urban waste-water treatment plants (Veríssimo et al., 2019). In the Agua Amarga salt marsh, artificially irrigating the surface was planned to conserve the ecosystem, partially reproducing a sheet of water similar to that of the solar salt works from the last century. The pre-existing high levels of salinity in the soil and groundwater were conditions that allowed seawater irrigation to be a viable solution. This is an unusual practice from a hydrogeological point of view, given that water with different origins (treated or runoff water) was the typical alternative (Shifflett and Schubauer-Berigan, 2019; Huang et al., 2021). Irrigating with seawater causes aquifer salinization. This is a negative effect that can be compared to saltwater intrusion in coastal aquifers, which is a widespread phenomenon requiring specific preventive management (Klassen and Allen, 2017). Complex interactions might emerge between the surface water and groundwater when managing coastal wetlands (White and Kaplan, 2017). In the case of Agua Amarga, the piezometric depression cone caused by abstractions causes that the seawater supplied by surface irrigation to flow towards the coast underground once it has infiltrated (see head contour lines on June 15, 2018; Figure S5), preventing the recharged seawater from flowing inland and even decreasing salinity in soils and groundwater, as numerical modelling foresaw (Alhama, 2013).

To understand the results of the field survey of soils, vegetation, and groundwater, it is interesting to present the irrigation plan. Artificial recharge by irrigation was implemented as a solution to mitigate piezometric drops, a measure that has proven to be effective in groundwater level recovery (Bouri and Dhia, 2010; Hussain et al., 2019). The 3 km long seawater distribution pipe system includes a set of 26 impulsion pumps and hydrants (see additional documentation). The system runs below the levee network constructed to separate salt ponds during saline exploitation period. It is located next to the tunnel of DA-II and near the DA-I pumping wells, where piezometric drops are greater, and as close to the coastline as possible to reduce the risks of inland groundwater salinization.

The volume of the seawater irrigation program in each area from January 2010 to March 2021 is depicted in Fig. 2c. General groundwater flow is southwest-northeast to the sea, which causes water storage in the southeastern part of the recharge area, raising piezometric levels. This caused irrigation in Z-3 to be stopped in 2015 due to high piezometric levels, while Z-1 and Z-2 continued being irrigated. In 2019, the six ponds with the lowest permeability (based on González-Alcaraz et al., 2015) were selected to maintain a permanently flooded area (Fig. 1).

The irrigation schedule was first established to stop every autumn since seasonal rainfall provided natural recharging during this period. However, in 2016, groundwater withdrawal was increased to supply desalination plants, leading to a drop in piezometric levels. It was then decided to irrigate non-stop. The total volume of seawater discharged over the saltmarsh from January 2010 to March 2021 was 6,975,722 m^3, 60% of which corresponds to the period between 2016 and March 2021.

3. Material and methods

Between 2010 and 2020, the piezometric and groundwater electrical conductivity (ECgw hereafter) were monitored during the first week of each month. In each of the piezometers, the probe "Heron Conductivity Plus" was calibrated with the conductivity standard solution of 1413 μS/cm. The probe was subsequently inserted into the borehole until it contacted the groundwater, at which point it emitted a sound, and the depth was noted and converted into the piezometry relative to sea level (m a.s.l.). Then, it was slowly inserted into the piezometer, and at every meter, the temperature and the ECgw were recorded (mS/cm corrected to be standardized at 25° temperature) once they stabilized. When the probe was recovered at the surface, it was cleaned with distilled water. In Fig. 1, the distribution of the piezometers monitoring the piezometric effect of both the irrigation plan and the abstraction of DA-I and DA-II can be seen. Two of the piezometers were located within the irrigation area (P-1 and P-8), another was more inland but within the salt marsh (P-2), and the last one was located midway between the pumping cone and the irrigation ponds (D-2). To compare the evolution of ECgw with that of the sea, the electrical conductivity of the seawater near the coastline was also monitored monthly.

In June 2012, with the help of aerial images and after several field trips, 63 sampling plots were located in the salt marsh based on vegetation cover, the dominant plant species, and the seawater irrigation plan. Surface soil samples were collected and analysed for several parameters including salinity and moisture. The most abundant ions in 1:5 soil:water extracts were (in mEq/L): Cl^- and Na^+ between ≈20 and ≈90; SO_4^{2-} between ≈20 and ≈60; Ca^{2+} between ≈20 and ≈40; and Mg^{2+} ≈10 and ≈20. More details of this initial study can be found in González-Alcaraz et al. (2014, 2015). Of the 63 initial plots, 21 plots (representative of the soils and vegetation in the marsh) were selected for long-term monitoring (Fig. 1). At each plot, three surface soil cores (≈20 cm) were extracted with a manual auger (inner diameter of 5 cm) and then mixed in the same bag to form a composite soil sample per point. An aliquot of each sample was weighed before and after heating at 65∘C until a constant weight was reached to determine soil moisture. The remaining material was air-dried and sieved through a 2-mm mesh. Soil electrical conductivity (ECs) was measured at 25∘C with a conductimeter (Crison Basic 30) in a 1:5 soil:water extract after 2 h of shaking. The percentage of vegetation cover was visually estimated in spring 2010 and spring 2020 in the 21 plots.

Data on ECs, moisture, and vegetation cover were grouped by the three irrigation areas and the adjoining areas (North, Central, and South; Z-1, Z-2, and Z-3 in Fig. 1, respectively), and statistical analyses were

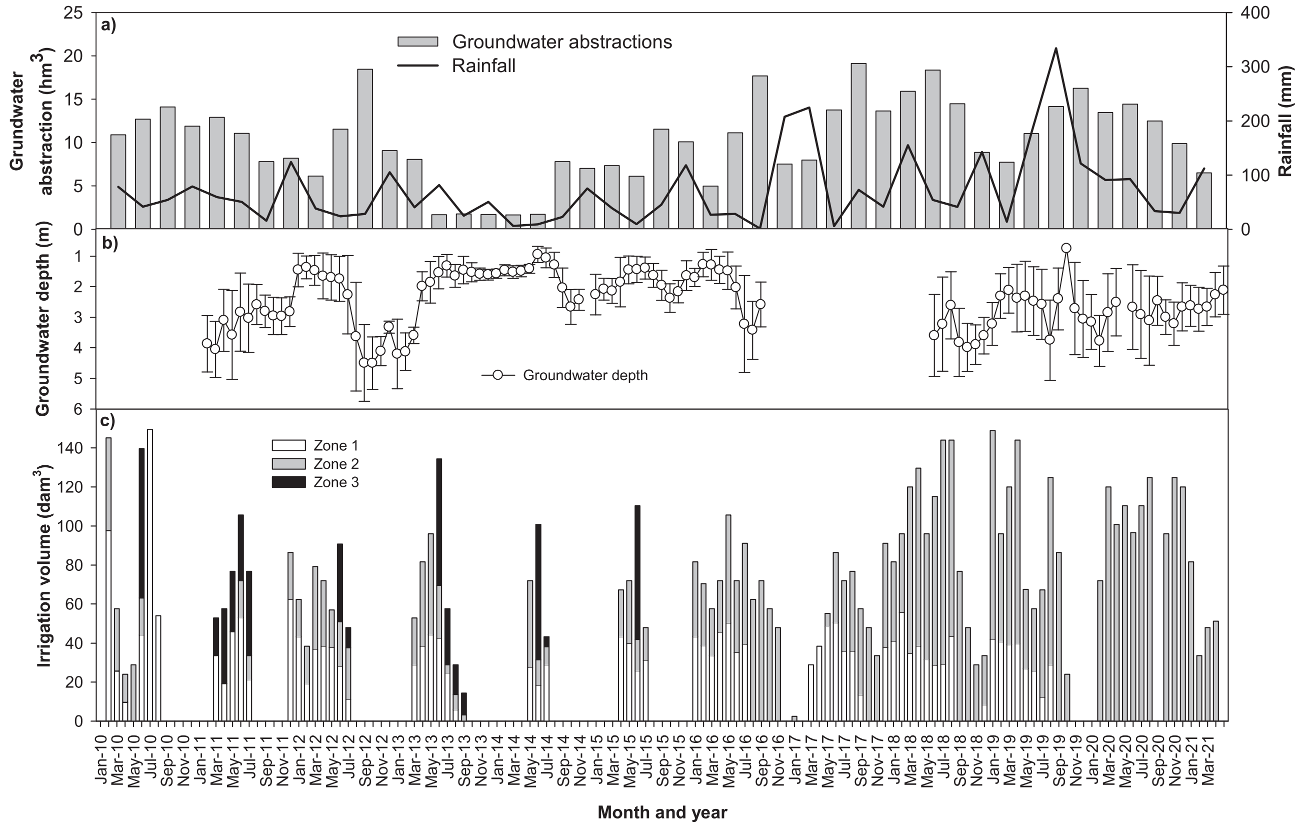

Fig. 2. Time series for: a) groundwater abstractions (DA-I + DA-II) and rainfall; b) average groundwater level depth (average of P1, P2, P8, and D2 ± standard error); gaps in the graph are periods without data; c) seawater irrigated volume in areas Z-1, Z-2, and Z-3 during the period January 2010 to March 2021.
Rain Source: https://www.tutiempo.net/clima/ws-83600.html

performed with IBM SPSS Statistic 22. The evolution of the ECs and moisture over time and the differences among the three groups (North, Central, and South) were analyzed using repeated ANOVA measures, followed by a Bonferroni post hoc test. Levene's test was applied to check the homogeneity of the variances. Differences were considered significant at $p < 0.05$. The relative influence of abstraction, irrigation water volumes, and precipitation on groundwater levels in P1, P2, P8, and D2, was studied with Spearman's rank correlations, which were considered significant at $p \leq 0.05$.

The field surveys covered the period from January 2010 to April 2021. However, due to administrative problems, no monitoring field surveys were carried out between September 2016 and April 2018, although the irrigation plan was still operational.

4. Results and discussion

In coastal aquifers, the standard groundwater level value of 0 m a.s.l. can be set as a boundary condition along the coastline. Natural factors influence this level, such as tidal motion, wave setup, atmospheric pressure, among others (Balugani and Antonellini, 2011; Wu and Zhuang, 2010). In Mediterranean regions, natural recharge (precipitation) is usually the leading cause of groundwater level fluctuation (Vallejos et al., 2015), but anthropogenic groundwater abstraction is by far the most influential (Zghibi et al., 2019; Jorreto et al., 2009). In the Agua Amarga coastal aquifer, groundwater levels reached 6.5 m in depth (-8.22 m a.s.l.) from 2010 to 2020 next to the DA-I pumping wells in the northeastern part of the saltmarsh and 3.2 m in depth (-1.4 m a.s.l.) next to the DA-II abstraction system in the eastern part of the saltmarsh.

Fig. 2 shows the quarterly data from 2010 to 2020 on abstraction volume and rainfall (Fig. 2a), the average groundwater depth below the saltmarsh (Fig. 2b), and the volume of irrigation (Fig. 2c). The average groundwater depth of the four piezometers monitored throughout this period was 2.5 m (hollow circle), ranging from less than 1.0 m in P-1 and P-8 (located next to the irrigated area) to more than 4 m in D-2 (closest to the DA-I pumping wells).

As expected, Spearman's rank correlations showed that the correlations between groundwater level in all the piezometers were significant (Table S1) since they are all situated in the same Quaternary-Noegene aquifer system ($r \geq 0.298$, $p \leq 0.003$).

In piezometers P-1 and P-8, located in Z-2 (Fig. 1), the correlations between irrigation and groundwater depth were negative ($r \leq -0.309$, $p \leq 0.001$) indicating that irrigation led to a shallower water table. In contrast, irrigation did not significantly affect the groundwater depth in piezometer P-2 ($r = 0.137$; $p = 0.155$), due to that the irrigation water flowed underground from the marsh to the coast (see groundwater flow direction in piezometric map, Figure S5). Moreover, the abstraction system generated higher hydraulic gradients on the coast than inland (where P-2 was located). The proximity of D-2 to the abstraction system determined that in this piezometer the groundwater depth was more influenced by abstraction ($r = 0.582$, $p \leq 0.000$) than by irrigation ($r = 0.270$, $p = 0.007$). The positive correlation between irrigation and groundwater depth in this piezometer was an indirect consequence of the fact that, from 2016 onwards, the volume of irrigation water increased when the volume of water abstracted increased (see Fig. 2). However, prior to 2016, the volume of irrigation water was not modulated by the volume of water abstracted, resulting in a non-significant correlation between the two volumes when considering data for the entire study period (2010–2020) (Table S1; $r = 0.089$, $p = 0.309$). Regarding the abstractions, groundwater depth had a positive and significant correlation ($r \geq 0.261$, $p \leq 0.00$) in all the piezometers.

Due to the influence of abstraction and irrigation on groundwater depth, there was no significant correlation between precipitation vs. groundwater depth ($r \leq 0.148$, $p \geq 0.125$; Table S1). However, punctual heavy rainfall events can have a noticeable effect on piezometry (Santoni et al., 2018), as reflected in September 2019 in Agua Amarga due to the exceptional Dana (Isolated Depression at High Levels) event, with 234 mm of precipitation, which raised groundwater levels to only about 0.5 m depth.

Given the significance of the correlations, the relative influence of abstractions and irrigation on groundwater levels on each piezometer can be determined by the ratio of the slopes of the corresponding linear regressions between abstraction vs. groundwater depth and irrigation vs. groundwater depth (Table 1 and Figures S3 and S4). The coefficients denote that D-2 (127.17) was the most affected by abstraction, while P-1 (43.1) and P-8 (29.25) the most influenced by irrigation.

Throughout the period studied, average groundwater levels were deeper in summer, when the abstraction rates increased, and irrigation decreased. Groundwater levels settled near the surface during the interval from March 2013 to August 2014 (around 1.5 m deep, on average), when the abstraction rate was the lowest. The increased pumping rates from 2017 caused the average depth of the groundwater to decrease to 2.8 m. As a general observation, irrigation helped avoid sharp fluctuations in average groundwater levels, such as those produced in January 2011, September 2012, 2014, 2015, 2018, 2020, and November 2019. This illustrates the importance of irrigation to keep groundwater levels close to the surface in the Agua Amarga saltmarsh.

Soil moisture (Fig. 3b) varied significantly during the 8-year period of 2012–2020 (time effect $p \leq 0.001$) in a different way in the three irrigation areas (time × area interaction $p \leq 0.012$). Soils in the southern area tended to be wetter than soils in the central and northern areas, with the latter being the driest during the period of 2018–2019, coinciding with deeper groundwater levels in piezometer D-2.

The general tendency of wetter soils in the southern area (not irrigated) than in the northern and central (irrigated) areas agrees with the presence of a surface soil layer (upper ≈20 cm) with lower bulk density (1400–1600 kg/m^3) in the southern area (González-Alcaraz et al., 2015), which implies greater porosity and, hence, greater water retention capacity. In contrast, the surface soil layers in the other areas showed very high bulk density, particularly in Z-2 (1700–2000 kg/m^3), which constrained water infiltration and moisture retention in the soil matrix.

Regarding soil and groundwater electrical conductivity (ECs and ECgw, respectively), chemical analyses of the groundwater below the salt marsh carried out in April 2009 (Alhama et al., 2012) resulted in ECgw values above 400 mS/cm. In the time series depicted in Fig. 4b, values over 300 mS/cm for piezometer P-8 can be seen in December 2009 and over 125 mS/cm for piezometer P-1 in October 2012. This high salinity is a consequence of saltwork activity (Gakweli et al., 2021), which created a layer of salt crystals over some ponds located in Z-1 and Z-2 and contributed to salinizing the subsoil and groundwater (Alhama, 2013).

During the first three years (2010–2012), important fluctuations of ECgw in piezometers P-1 and P-8 were recorded, ranging from 70 to 200 mS/cm, respectively. Variations were less pronounced in D-2 (between 38 and 61 mS/cm) and P-2 (between 14 and 26 mS/cm). This variability in ECgw can be attributed to the fact that irrigation was interrupted several times over the period and the volumes applied were not always the same (Fig. 4a).

Through the period 2013–2015, salinity slowly increased in all the piezometers, although compared to the previous triennium, the average relative difference with the seawater EC in P-8 changed from 50% to 15% (right axis in Fig. 4b). Halts in irrigation and a decreasing abstraction rate (Fig. 2a and 3c) caused the velocity of the water in the porous media to decrease, diminishing the effect of the advection process and the removal of salt from the groundwater (Zheng and Bennett, 2002).

From 2018 onwards, the ECgw in piezometers P-1, P-8, and D-2 was more and more similar to that of seawater (55,3 mS/cm on average), with a relative difference to seawater in P-8 close to 0%. The permanent irrigation and rising abstraction rate (Fig. 2a and 2c) caused an increase in the hydraulic gradient (between 1 and 15°/$_{oo}$ in a SW-NE cross-section, including the P-8 piezometer and DA-I pumping well systems) that produced greater velocity in groundwater flow and, consequently, dilution of groundwater salinity due to advection. The ECgw of P-2 remained below 25 mS/cm, and the average relative difference with respect to seawater was 84% (Fig. 4b). This shows that the irrigation plan did not increase groundwater salinity inland. Furthermore, the depression cone caused by abstraction acted not only as a hydraulic barrier to prevent seawater intrusion (Pool and Carrera, 2010) but also as a drain for seawater that was poured into the saltmarsh by the irrigation plan. Regarding the vertical depth profiles for EC_{gw} (Fig. 5), the data indicate brine accumulated below the depth of 5 m, as shown by $EC_{gw} > 100$ mS/cm between 2010 and 2013. This high salt accumulation was progressively dissolved following salt marsh irrigation with seawater, as shown by the decrease of EC_{gw}, until it reached a value similar to that of seawater (average EC = 55.3 mS/cm). The existence of brine at depths is a consequence of permanent saltwater recharge due to saltwork activity and the density-driven flow in porous media. Classical Benchmark problems like "Elder," "Salt Lake," or "Henry" presented similar scenarios (Holzbecher, 1998). The predominance of an advective process caused by the hydraulic gradient between the irrigation area and the depression cone explains the progressive dilution of groundwater salt (Alhama et al., 2021). Diverse experiences with artificial (injection wells or surface recharge) or natural (rivers, rainfall) recharge in coastal aquifers have been shown to contribute to aquifer desalination (Custodio, 2010). Agua Amarga has the special feature of carrying out this process using seawater.

Soil salinity (ECs, Fig. 3a) significantly changed over the 2012–2020 period (time effect $p \leq 0.001$) in a similar way in the three irrigation areas (time × area significant interaction, $p > 0.05$). An increasing tendency was observed between 2012 and 2015, with a sharp drop from 2015 to 2018 and stabilization in 2019 and 2020. The tendency of ECs was similar to that of ECgw (Fig. 4b). Soils in the southern area were significantly more saline in 2012 and 2013 (ECs significantly higher, $p \leq 0.015$), but the differences disappeared from 2014 onwards. The ECs values were > ≈6 mS/cm until 2016, but between 2018 and 2020, the average ECs values were ≈2-4 mS/cm.

The volume of water that the salt marsh received from 2016 onwards increased due to two factors: increased irrigation water poured into the salt marsh (Fig. 5a) and more rainfall (Fig. 5b). It is feasible that this greater volume of water contributed to the dissolution and partial leaching of salts from the upper soil layers, as shown by the general decrease in ECs. Despite the latter, in 2020, soils were still highly saline, and the salt crusts observed in many of the ancient salt ponds continued appearing over the surface, mainly during drier periods (Figure S2). These crusts contribute to habitat heterogeneity in the salt marsh (González-Alcaraz et al., 2015).

Finally, the vegetation cover (Table 2, Table S2, Fig. 6 and Figures S6 to S10) in the three areas of the salt marsh (i.e., Z1, Z2, and Z3) increased

Table 1
Relationships between abstractions and irrigation volume (hm^3/month) on groundwater level depth (m) on each piezometer for the period 2010–2021. The slope of the linear correlations (columns 2–3) and the ratio between abstraction and irrigation are shown (column 4).

	Slopes of the linear regressions (and R^2)		Slope ratios
Piezometer	Abstraction vs. Groundwater depth	Irrigation vs. Groundwater depth	Abstractions/ Irrigation
P-1	0.604 ($R^2 = 0.14$)	−0.014 ($R^2 = 0.11$)	43.1
P-2	1.327 ($R^2 = 0.26$)	0.012 ($R^2 = 0.03$)	110.60
P-8	0.468 ($R^2 = 0.05$)	−0.016 ($R^2 = 0.08$)	29.25
D-2	0.763 ($R^2 = 0.64$)	0.006 ($R^2 = 0.06$)	127.17

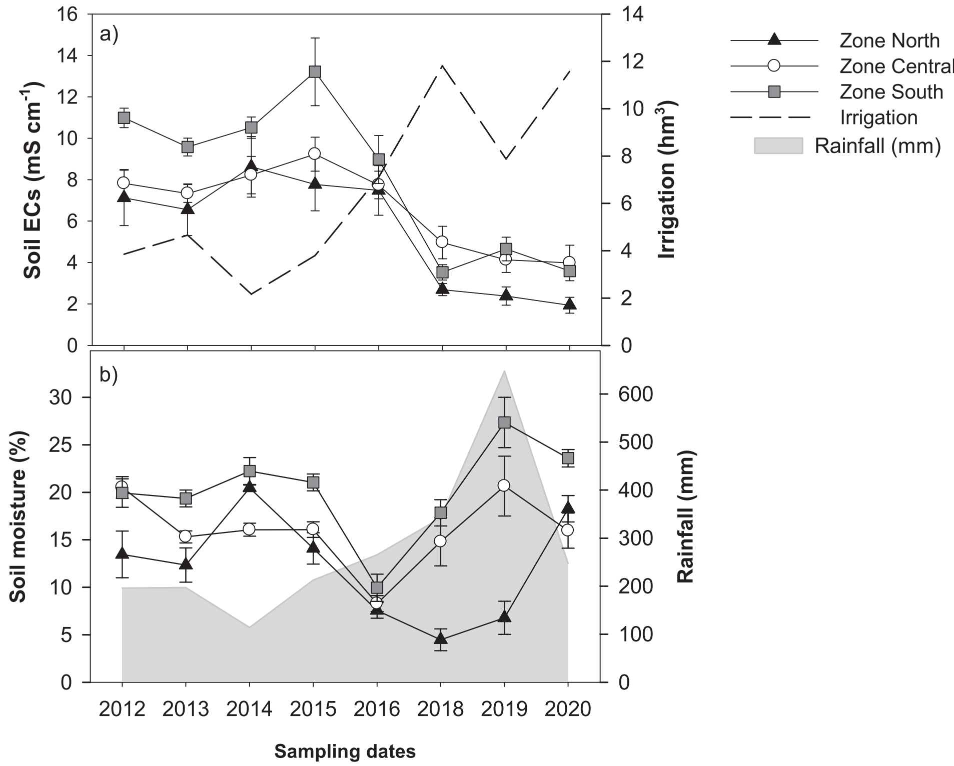

Fig. 3. a) Evolution of soil electrical conductivity (EC) over the study period and volume of irrigation water poured yearly into the salt marsh; b) evolution of soil moisture throughout the study period and total annual rainfall in the area. Values of ECs and moisture are average ± standard error.

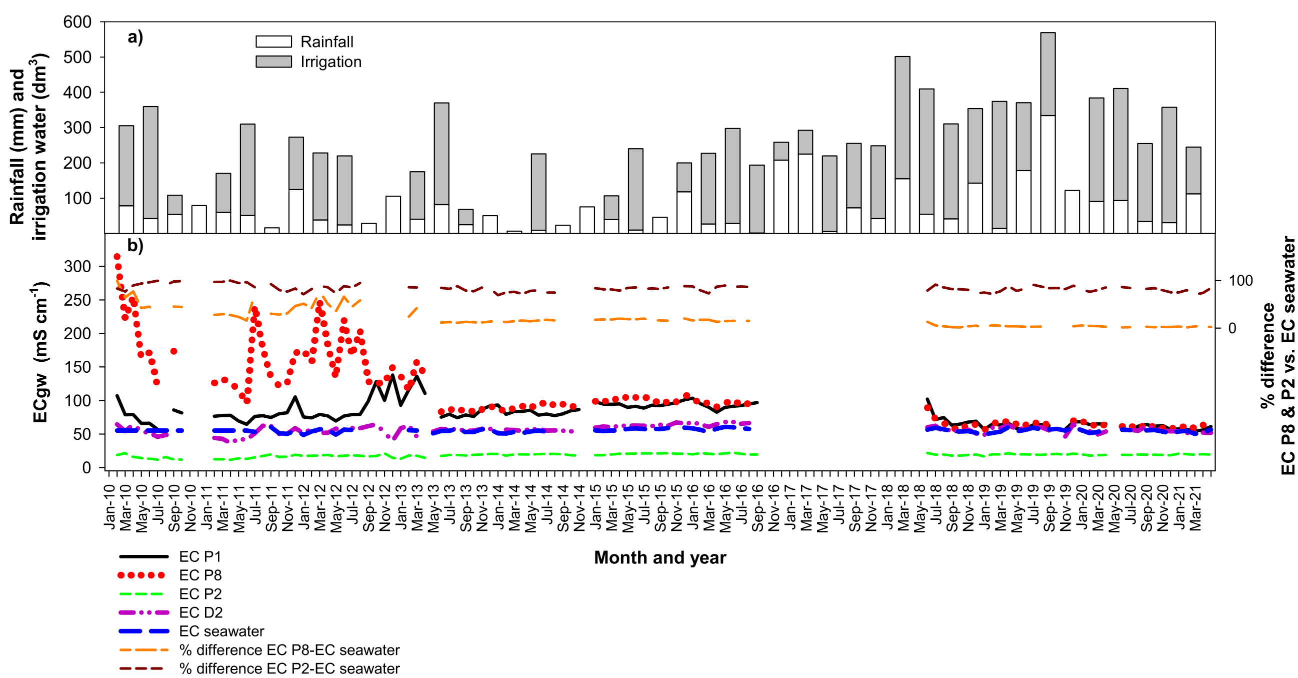

Fig. 4. Monthly time series for: a) Irrigation and rainfall temporal series; b) Electrical conductivity of groundwater at 10 m depth (ECgw, mS/cm) in P-1, P-2, P-8, and D-2 during the period 2010–2020 (left vertical axis) and percentage of relative differences between P-2 and D-2 and seawater (right vertical axis). The gaps in the chart are periods without data, because the monitoring was temporarily stopped.

between 2010 and 2020. The changes in cover percentage were more evident in Z3, which reached an average ≈34% higher in 2020, while in Z2 the increase accounted by an average of ≈19% and in Z1 ≈15%. It is of highlight that several salt ponds located in Z2 that were devoid of vegetation in 2010, had been partially colonized in 2020. The most favoured species in the whole salt marsh were *Arthrocnemum macrostachyum* and *Sarcocornia fruticosa* (Table S2) which colonized bare areas even in sites where a surface salt crust of several centimetres appeared (see additional documentation). These two halophytes are among the species most tolerant to salinity and flooding in the Mediterranean area

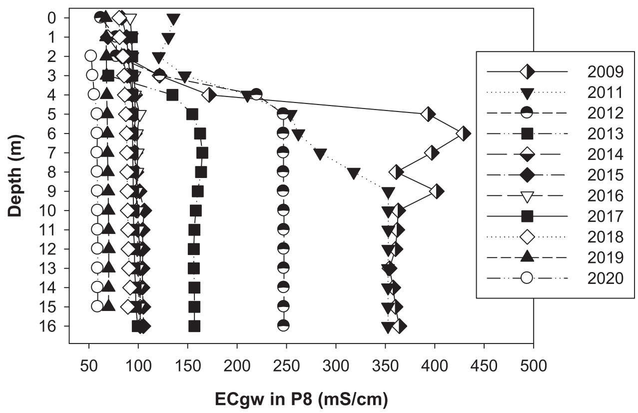

Fig. 5. ECgw (mS/cm) depth Profiles in P-8 corresponding to the highest values of each year over the period 2010–2021. Values are recorded every meter depth.

Table 2
Changes in vegetation cover between 2010 and 2012. Values are the average of the sampling plots ± standard error. Number in brackets are minimum and maximum cover.

Irrigation zone	2010	2020
	% vegetation cover	
Z1	51.2 ± 10.1	66.2 ± 10.3
(4 sampling plots)	(35–80)	(40–90)
Z2	13.7 ± 7.2	33.1 ± 7.7
(8 sampling plots)	(0–55)	(10–60)
Z3	43.9 ± 9.3	78.3 ± 4.8
(9 sampling plots)	(15–95)	(50–95)

(Álvarez-Rogel et al., 2000). Since salinity was similar in the three areas, this factor played a minor role in vegetation changes. In contrast, it is plausible to assume that the wetter soils with lower bulk density in the southern area could favour greater plant development (see additional documentation).

5. Conclusions

One of the most outstanding effects of the irrigation program was the general decrease in groundwater salinity, from values reaching 300 mS/cm in 2009 below ponds with the most intense salt industry activity, to salinity values similar to seawater (average 55.3 mS/cm) in July 2018. The seawater supply also helped avoid sharp fluctuations in groundwater levels, keeping them at an average of less than 1.0 m deep in the main irrigation area to more than 4 m deep near the DA-I pumping wells. The irrigation program did not affect inland groundwater salinity due to preferential groundwater flow to the abstraction system located along the coastline.

The average increase in vegetation cover over ten years (from 10 to 25 %) indicates that seawater irrigation enhanced the health of saline habitats, with *Arthrocnemum macrostachyum* and *Sarcocornia fruticosa* being the most favored species. Moreover, areas with prolonged flooding were extensively colonized by the aquatic species *Ruppia maritima*.

In conclusion, data collected during the monitoring programme suggest that the remediation program carried out over a decade (2010–2020), with 6,975,722 m^3 of seawater discharged over the Agua Amarga salt marsh, has contributed to the sustainability of groundwater resources and the ecosystem. If so, seawater irrigation has been shown to be an appropriate strategy to mitigate the risk of environmental impacts caused by the abstraction of water from the underlying coastal aquifer to supply the DA-I and DA-II desalination plants.

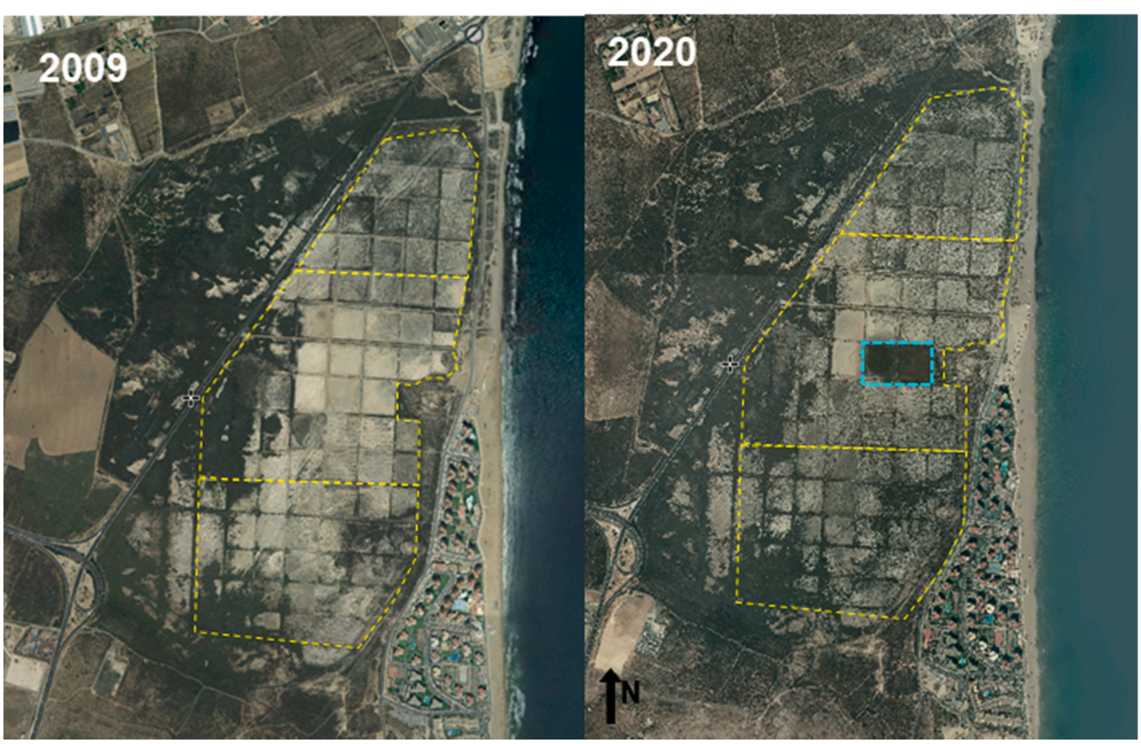

Fig. 6. Aerial images of the Agua Amarga salt marsh. Many areas devoid of vegetation or with low cover in 2009 (in white/light grey) appear colonized or with denser cover in 2020 (dark mottling/dark grey). The dotted yellow line indicates the three areas considered when comparing vegetation cover. The dotted blue line indicates the concentrated irrigation ponds with prolonged flooding since 2016. Images from PNOA (Instituto Geográfico Nacional of Spain: https: www.ign.es/web/comparador_pnoa/index.html). (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supplementary material

Supplementary data to this article can be found online at https://doi.org/10.1016/j.catena.2022.106429.

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