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ORIGINAL PAPER

Measurement and heuristic modelling of nitrogen and salt dynamics under *Salicornia* growing in a hyper-arid region and irrigated with groundwaters of differing nutrient and Salt loadings

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Abstract

New data highlight the economic value of using nitrogen-rich saline waters, either from groundwater or reject brines from desalination units, to irrigate the halophytic crop *Salicornia bigelovii* for food, fodder, and fuel in a hyper-arid environment. The greatest benefit was achieved using pressure-compensated drippers. Field measurements of drainage and leaching under the crop showed that all of the salt and nitrogen from the groundwater was returned back to the aquifer as leachate. A simple, heuristic model of groundwater quantity and quality was developed to infer the environmental impacts of irrigating crops with saline and high-nitrate groundwater in a hyper-arid environment. The rise in solute concentration in groundwater is hyperbolic. The parameters needed for this simple model are the fraction of the land that is irrigated, the initial depth of the saturated thickness, the saturated water content, and the annual rate of evapotranspiration. An indicator of the time-rise is the number of years to double the solute concentration. This is $\Theta_A h_o /2\ ET_C$, where Θ_A is the aquifer's saturated water content, h_o is the original thickness of the saturated layer, and ET_c is the annual rate of crop evapotranspiration. The general model is simple and straightforward to parameterise to predict the evapoconcentration of groundwater salinity.

Research Highlight

Salicornia, a halophyte, grew well when irrigated with nitrogen-rich, saline groundwaters.
The greatest economic productivity for irrigation with saline groundwaters was via drippers.
Measured leachate concentrations of salt and nitrogen were up to three times that applied.
A heuristic model was developed for groundwater quantity and aquifer quality.
The model predicts future aquifer volumes and evapoconcentrated solute concentrations.

Introduction

The United Arab Emirates (UAE) is a hyper-arid region with annual precipitation, P, being well less than 5% of the reference evapotranspiration, ET_o. The former sheikhdoms that now comprise the UAE signed a Perpetual Maritime Truce with the British in 1853, such that they subsequently became known as the Trucial States. In the 1950–60s the British sought to find water to expand agriculture across the 'fertile sands' of the Trucial States and Oman (Joseph and Howarth 2015). Sheikh Zayed, the first President of the UAE who ruled from 1971 to 2004, continued to expand the development of water resources and agriculture (Joseph and Howarth 2015; Al-Yamani et al. 2019). Today there is a

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continuing call for the global development of food, fodder, and fuel production in arid regions under saline conditions. Vellinga et al. (2022) outlined the actions needed to increase opportunities and capacities for agriculture under saline soil and water conditions.

Lyra et al. (2021) showed how innovative practices to grow the halophyte *Salicornia bigelovii* can provide a value chain from desert farm to consumers' fork. These initiatives rely on the use of saline groundwaters either directly, or indirectly through land-based aquaculture, to irrigate *Salicornia* under these hyper-arid and saline conditions. The benefits from the food, fodder, and fuel value of halophyte production have been well detailed (Lyra et al. 2014, 2016; Araus et al. 2021; Al-Tamimi et al. 2023a). However, less is known about the environmental impacts on groundwaters from the use of saline aquifers for irrigation. It is known that across the UAE groundwater levels are dropping and that groundwaters are high in nitrogen, with salinity concentrations rising (MOEW, 2014, 2015; McDonnell and Fragaszy 2016). The groundwater atlas of Abu Dhabi (EAD 2018) provides detailed information about the characteristics and geohydrologies of the aquifers of this Emirate. Many of the aquifers in Abu Dhabi are located in essentially endorheic basins. From global evidence it is also known that if appropriate policy actions are implemented, then recovery in aquifers is possible (Jasechko et al. 2024). In the UAE some 10% of groundwater extractions are for irrigating arid-forests, and in 2016 to reduce pressure on aquifers the resource consents of over 15% of the wells used for irrigating these trees were cancelled. The trees left to survive on their own (Al-Yamani et al. 2019).

Our earlier research on irrigation in this hyper-arid region developed a crop-factor, *K*c, parameterisation to predict crop water-use ET_c (Allen et al. 1998) from ET_o across a range of crops (Al-Tamimi et al. 2022). Additionally, we developed modified devices to measure soil-water content and leaching in saline desert soil (Al-Tamimi et al. 2023b) and validated a piston-displacement model of salt leaching under saline irrigation (Al-Tamimi et al. 2023c). The water productivity of *Salicornia* under saline irrigations was also quantified (Al-Tamimi et al. 2023a).

A simple, heuristic modelling framework is developed, which links contemporary halophyte irrigation practices with current and future impacts on the underlying aquifer in this hyper-arid region. This new research had three objectives:

- To extend earlier results of halophyte yield and gross economic water productivity, but now through irrigating *Salicornia* with lower salinity water at 10 dS m^{-1}, and to test the impact of a halving of the leaching fraction under bubbler emitters,
- To measure not only salt leaching (Al Tamimi et al. 2023a, b, c), but also the leaching of nitrogen from the rootzone of the halophyte *Salicornia* when irrigated with nitrogen-rich saline groundwaters and brines, and.
- To develop a heuristic model to predict the impact on groundwater quantity and quality from irrigation of a halophyte with saline and nitrogen-rich waters in a hyper-arid region.

Materials and methods

Experimental trials

The experiments described here were conducted using the same layout of the plots described in Al-Tamimi et al. (2023 a, b). Six of the nine treatments were repeated here. Two of three new treatments examined the impact of irrigating with lower salinity water at 10 dS m^{-1}, and the third was to test whether halving the leaching fraction would have any impact. Most of the experimental details are described in Al-Tamimi et al. (2023a). Only a brief summary is provided here.

The experiments were carried out at the International Center for Biosaline Agriculture (ICBA) (25.09° N; 55.39° E; 48 m a.s.l.) near Dubai. Whereas these new results repeat six of the earlier treatments the three new experiments reported here are for just one year, namely 2022/2023. We again consider this to be agronomically valid in this hyper-arid environment. The weather is virtually always cloud-free, and rainfall is extremely rare. Inter-annual variation is virtually non-existent. The soil at the site is a Typic Torripsamment.

Crop agronomy

The *Salicornia* seeds were sown on 15 November 2022. To enable good germination and establishment, all the plots were irrigated until 22 February 2023 with low salinity water at 10 dS m^{-1}. Then four types of saline waters were used for irrigation: aquabrine (AQ; ≈ 40 dS m^{-1}), reverse-osmosis (RO) reject brine from the desalination plant (≈40 dS m^{-1}), groundwater (GW, ≈ 22.5 dS m^{-1}), and lower salinity water (EC10, ≈10 dS m^{-1}). There were three irrigation systems: bubblers (BUB), pressure-compensated drippers (PCD), and sub-surface tape irrigation (SUB). Irrigation was stopped in the first week of August 2023.

Nine separate plots each of 8 m by 8 m were established in a square matrix layout, with 4 m borders between plots. The rows of the matrix were the different water-sources of AQ, RO, GW and EC10. The columns of the matrix were the emitter-device types of BUB, PCD, and SUB. The AQ-BUB* treatment applied 30 mm d^{-1}, whereas the AQ-BUB applied half that at 15 mm d^{-1} in order to compare the reduced leaching fraction. Within each plot, four quadrants, each of 2 m by 2 m, were created, and modified drainage fluxmeters (Al-Tamimi et al. 2023b) and vertical TDR probes of length 600 mm were installed near the centre of each quadrant.

The crop was harvested for the yield of fresh tips on 14 April 2023, total harvest fresh weight on 14 June 2023, and for seed on 15 August 2023 after irrigation had been stopped on 2 August, and the crop had dried off. Crop samples of the same number of plants were taken from randomly selected locations within each plot, although locations near to measuring devices were avoided. Only the fresh-weight harvest data are presented here in order to highlight the impact of the three new treatments. The data were analysed using one-way ANOVA. Comparisons among means were made using Fisher's least significant differences at $p = 0.05$ (5% LSD).

Results and discussion

Crop yields

The fresh weights of *Salicornia* at harvest for each of the treatments are given in Fig. 1. In agreement with the earlier results of Al-Tamimi et al. (2023a) the yields with aquabrine were the highest. However, here we show that there is no difference between AQ-BUB and AQ-BUB* for drippers indicating that reducing the irrigation to lower the leaching fraction had no impact, given that the leaching fraction was still well above unity.

And in all cases, the application of irrigation water via drippers was the same, or better, than that achieved with bubblers.

Irrigation and productivity

In Table 1 we present the seasonal irrigation amounts (I, m^3 m^{-2}) across the treatments, along with the yield (Y, kg m^{-2}) information that is presented in Fig. 1. Using the terminology of Fernández et al. (2020) we present in Table 1 the irrigation water productivity *WP* (Y / I, kg m^{-3}) and gross economic water productivity (*GEWP*, \$ m^{-3}).

The highest *WP* values were for the saline irrigation waters applied through drippers, and they exceeded 2.5 kg m^{-3}. Being an obligate halophyte, the water productivities using the lower salinity EC10 waters were less than half of those. Detailed pricing for fresh-tips, forage harvest, and seeds were given in Al-Tamimi et al. (2023a). Here for illustrative purposes, we simply use a weighted-mean price for the fresh harvest of US\$1.3 kg^{-1} to calculate the *GEWP*. For all of the saline waters, the highest economic returns were for the crops irrigated by drippers, being above about \$3.50 m^{-3}. Since two of the saline water sources are derived from the reject brine of desalination units, it is pertinent to note that the operational cost of desalination is now, thanks to solar-powered units, of the order of just \$0.55–0.63 m^{-3} (Dawoud et al. 2024). As well, there is the primary value that is derived from the desalination units through the use of 'freshened' brackish groundwater to irrigate high-value crops (Dawoud 2017). The use of desalination units to irrigate both high value crops with freshened groundwater, and halophytes with the reject brines, provides economic benefits for agriculture in hyper-arid regions with saline groundwaters.

Salt

We also sought to measure using modified drainage tension fluxmeters (DFM) (Al-Tamimi et al. 2023b), the leachate loads of salt leaving the rootzone of the halophyte *Salicornia*

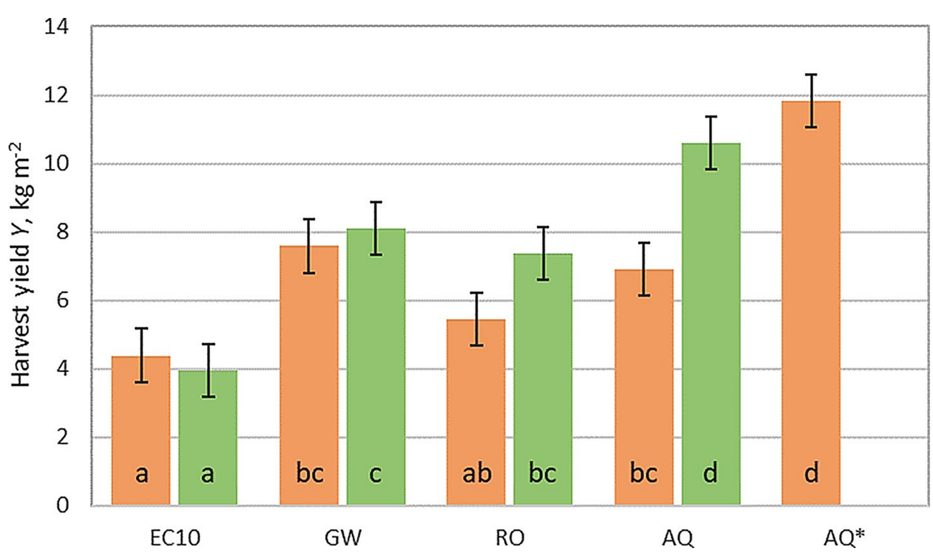


Fig. 1 The mean fresh weight of *Salicornia* at harvest (kg m^{-2}) for the nine plots: low salinity water EC10, groundwater GW, reverse-osmosis water RO, and aquabrine AQ. The different columns are for bubblers (left, v BUB) and pressure compensated drippers (right, v PCD). The AQ application rate was 15 mm d^{-1}, whereas AQ* was 30 mm d^{-1}. The error bars are the pooled standard error of the mean. Means with the same least significant difference (LSD) comparison letter are not significantly different ($p > 0.05$)

Table 1 The seasonal application of irrigation, I (m^3 m^{-2}) to the nine treatment plots of aquabrine (AQ), reject brine (RO), groundwater (GW) and low salinity (EC10), by two types of emitters of bubblers (BUB) and pressure compensated drippers (PCD). The AQ-BUB treatment applied 15 mm day^{-1}, whereas the AQ-BUB* treatment added 30 mm day^{-1}. Also shown are the modelled drainage losses ($I - ET_c$) (m^3 m^{-2}), plus the measured drainage losses over the 2022–2023 season from the four tension-fluxmeters within each of the plot. Crop evapotranspiration over the season, ET_c (m^3 m^{-2}), was modelled following Al-Tamimi et al. (2022). The biomass fresh-weight yields, Y (kg m^{-2}) are given along with the water productivity of $WP = Y / I$ (kg m^{-3}) and the gross economic water productivity (GEWP, \$ m^{-3}) using a mean-weighted price of \$1.3 kg^{-1} adapted from Al-Tamimi et al. (2023)

Treatment	Irrigation I (m^3 m^{-2})	Modelled Drainage (I - ETc) (m^3 m^{-2})	Measured Drainage (m^3 m^{-2})	Biomass Yield Y (kg m^{-2})	Water Productivity WP (kg m^{-3})	Gross Economic Water Productivity (\$ m^{-3})
AQ-BUB*	6.37	4.69	3.42	11.83	1.86	2.40
AQ-BUB	3.38	1.80	1.33	6.90	2.04	2.63
AQ-PCD	2.95	1.47	0.57	10.61	3.60	4.64
EC10-BUB	4.15	2.50	0.84	4.38	1.06	1.36
EC10-PCD	3.17	1.70	0.30	3.95	1.25	1.61
GW-BUB	4.47	2.62	1.31	7.58	1.70	2.19
GW-PCD	3.05	1.58	1.18	8.09	2.65	3.42
RO-BUB	4.37	2.69	1.10	5.46	1.25	1.61
RO-PCD	2.51	1.03	1.11	7.38	2.94	3.79
Average Values	**3.82**	**2.23**	**1.24**	**7.35**	**2.04**	**2.63**

under irrigation regimes that ensure there is a salt leaching fraction to flush excess salts from the rootzone. The results add new findings to those of Al-Tamimi et al. (2023a) and include the three new treatments. The electrical conductivities of the leachates measured by the DFMs are shown in Fig. 2. These results include, where relevant, the concentrations measured during the previous season. These data were used in conjunction with the drainage measurements to calculate the mass balances of salt lost from the rootzones over the 2022/2023 season (Table 2), along with the salt loading measured in the irrigation water that was applied.

Averaged across all nine plots, the amount of salt added of 55.9 (± SD 26.9) kg m^{-2}, was equal, given natural variability, to that lost in the leachate of 64.9 (± 37.9) kg m^{-2}. For the previous season, Al-Tamimi et al. (2023a) found that the salt loading was 79.6 (± 48.2) kg m^{-2}, and that leaching was 70.4 (± 69.6) kg m^{-2}. Again, the salt loss balanced with the salt added. The halophyte *Salicornia* has been found to accumulate salt at between 37 and 52% of dry weight (DW) (Grattan et al. 2008). Al-Tamimi et al. (2023a) found that in 2022/2023 the *Salicornia* yielded about 1.75 kg m^{-2} on a dry weight basis (DW). So even if the salt content were 50% DW, the amount of salt removed in the vegetation would be less than 1 kg m^{-2}. So it is reasonable to assume that the total amount of salt applied in the water used to irrigate an obligate halophyte is lost in the leachate.

Nitrogen

The nitrate concentrations in the leachate were also measured by the DFMs and are shown in Fig. 3, along with the nitrate concentrations in the irrigation water. Unlike salinity, the nitrate levels vary temporally, albeit they are quite high. Groundwaters in the Abu Dhabi Emirate are often naturally high in nitrogen (McDonnell and Fragaszy 2016). The groundwater concentrations can exceed 50 mg L^{-1} nitrate in many areas around Al Ain and the Liwa Oases.

The average nitrate concentration across all the waters used for irrigation, prior to the establishment of the treatments, was 30.2 mg L^{-1} nitrate (Fig. 3). From Table 2 it can be seen that the average load of nitrogen applied over the 2022/2023 season through the irrigation waters was 374 (± 149) kg-N ha^{-1}. The average seasonal leaching loss of nitrogen measured by the nine tension-drainage fluxmeters was found to be 382 (± 178) kg-N ha^{-1}. In a mass-balance sense, as much nitrogen that was applied in the irrigation water was lost as leachate back to groundwater (Table 2).

However, the nitrogen removed by the *Salicornia* at harvest was found to be 303 (± 130) kg-N ha^{-1}. Prior to transplanting, organic fertiliser was applied to the plots at a rate of 539 kg-N ha^{-1}. Geissler et al. (2021) showed that depending on the constituent make-up and C: N ratio of organic amendments, some 32–72% of the nitrogen in the compost could be mineralised in the first 100 days. Therefore, it would not be surprising if 56% (viz. 303¸539 as a %) of the nitrogen in the compost were to be mineralised and taken up by the *Salicornia* crop. This would support the notion that under current land-management practices using pre-planting compost, the total amount of nitrogen applied in the irrigation water would be leached back to groundwater in the drainage.

The leaching fraction

Figures 2 and 3 show the time course of the concentrations of salt and nitrate in the irrigation water applied to the plots,

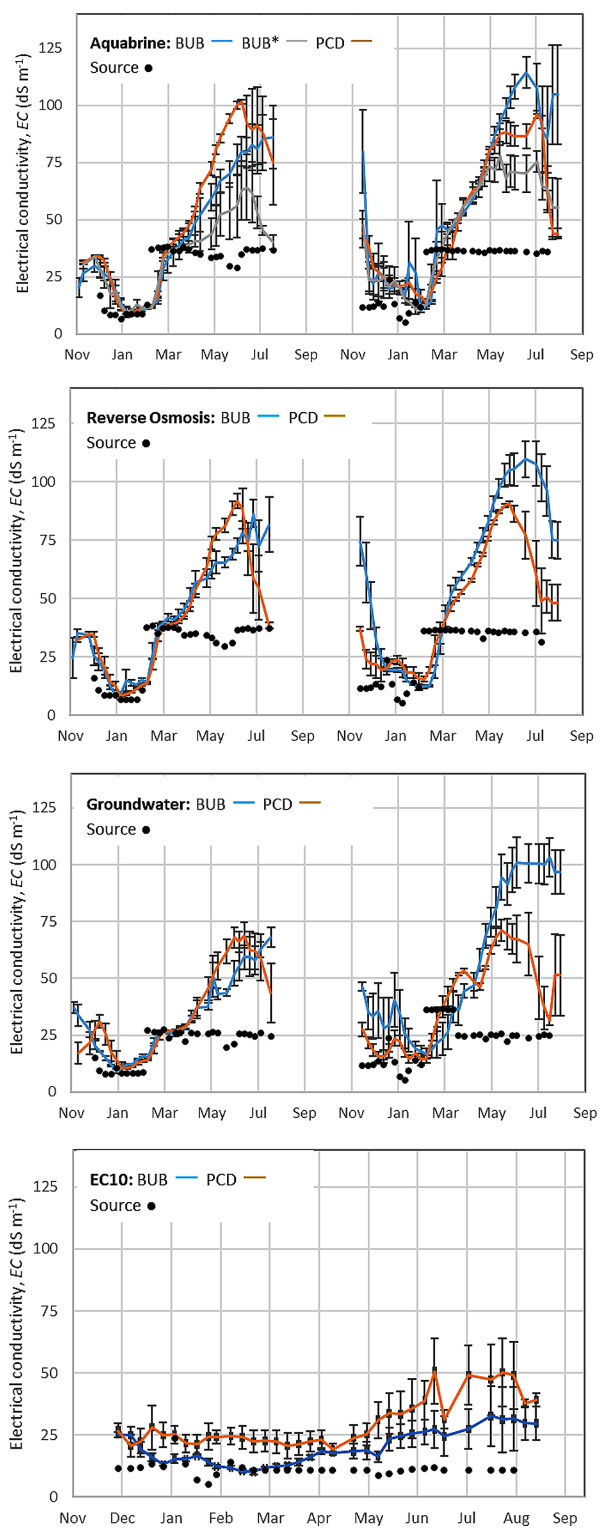


Fig. 2 The electrical conductivity (EC, dS m^{-1}) of the leachate measured over two years (2022 and 2023) in drainage under plots of a *Salicornia bigelovii* crop in the United Arab Emirates by tension drainage fluxmeters (DFM) under irrigation with aquabrine (AQ, top), reverse osmosis water (RO, upper middle), groundwater (GW, lower middle), and water at *EC* 10 dS m^{-1} (EC10, bottom, for just 2023). Water was applied either through bubblers (BUB) or pressure compensated drippers (PCD). There was in 2023 an extra aquabrine treatment (AQ-BUB*) where water was applied at 30 mm d^{-1}, rather than 15 mm d^{-1}. The switch between low salinity irrigation to the treatment waters was on 23 February 2022 and 22 February 2023. The original irrigation was water was at EC at 10 dS m^{-1}, and then under aquabrine and reverse-osmosis brine at about 40 dS m^{-1}, and groundwater at 25 dS m^{-1}, and in 2023 low salinity water at $EC = 10$ dS m^{-1}. The bars represent the standard errors of the measures from 4 DFMs for each emitter type

C_{in} (kg m^{-3}), relative to that in the leachates leaving the rootzone, C_{out} (kg m^{-3}), destined for groundwater below via the unsaturated vadose zone. Ayers and Westcot (1994) noted that for a leaching fraction, *LF*, defined as.

$$LF = \frac{Depth\ of\ water\ leached\ below\ the\ rootzone}{Depth\ of\ irrigation\ water\ applied} \tag{1}$$

the C_{out} in leachate will then be given by.

$$C_{out} = C_{in}\, LF^{-1} \tag{2}$$

The lower the *LF*, with less water draining through the profile, the higher the relative C_{out} in the leachate. Equation [1] for the *LF* can be written as being the drainage, namely ($I - ET_c$) divided by irrigation *I*. Thus using this simple mass-balance approach,

$$\frac{C_{out}}{C_{in}} = LF^{-1} = \frac{I}{I - ET_c} \tag{3}$$

The average measured numerator of irrigation is given in Table 1 as 3.82 m^3 m^{-2}. There are two ways to infer the drainage in the denominator: that directly measured which is on seasonal average 1.24 ($\pm$0.9) m^3 m^{-2} (Table 1), or that found using the modelled ET_c (Al-Tamimi et al. 2022) of 2.23 ($\pm$1.1) m^3 m^{-2}. These are not significantly different at $P = 0.05$. For inferential purposes using daily values of *I* and ET_c in Eq. [3], we chose to use the modelled ($I - ET_c$) as it has fewer errors-of-observation. The average daily values of $I / (I - ET_c)$ across all nine treatments are shown in Fig. 4, along with the 14-day running mean to show the general seasonal trend.

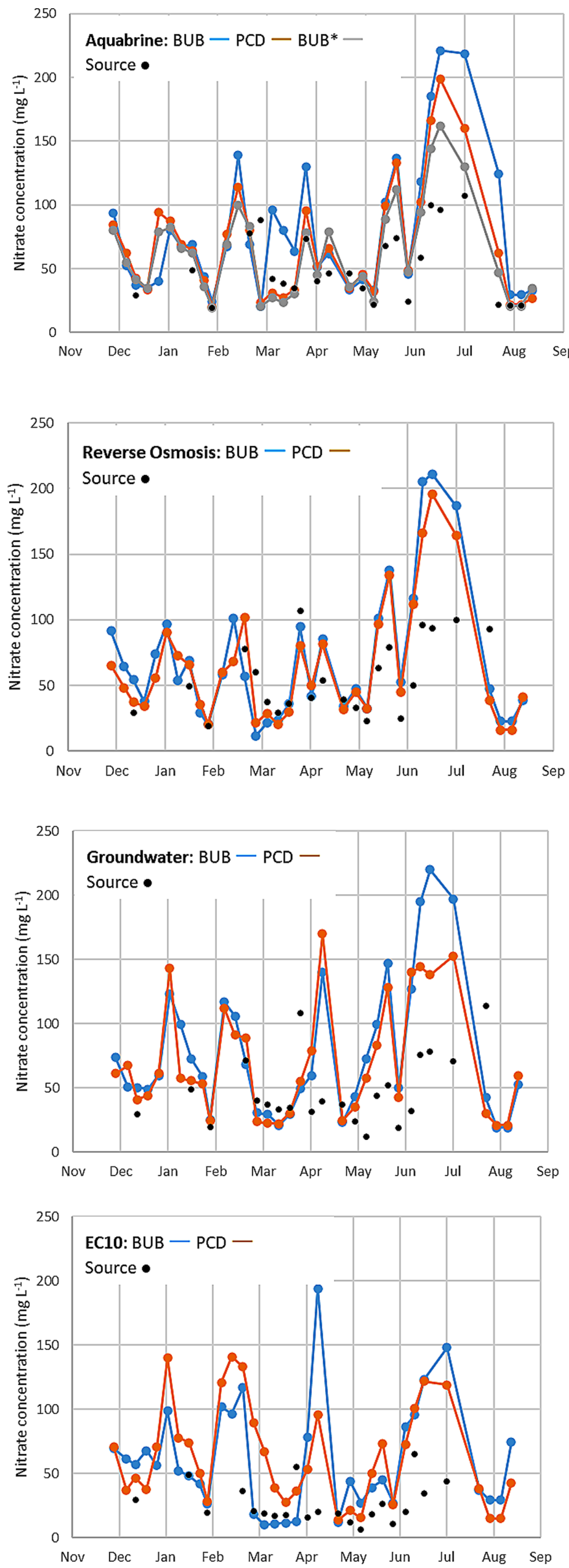
The seasonal pattern in Fig. 4 mimics the salt concentration ratios in Fig. 2 for salt. It is more difficult to discern this trend in the nitrate concentrations (Fig. 3) because of the temporal variation in the C_{in}. However, it is clear that at the peak of the growing season between May and July, C_{out} well exceeds C_{in} for nitrate. Between 3 May and 5 July 2023, the average $I / (I - ET_c)$ was 3.5 ($\pm$1.3). The measured ratios of C_{out} / C_{in} (Eq. 3) were 3.0 ($\pm$1.4) for salt, and 2.1 ($\pm$0.5) for nitrate. These are not significantly different at $P = 0.05$. This demonstrates that irrigation and crop water-use dictate both the concentration of solutes leaching back to groundwater

Table 2 The seasonal loadings of salt (kg m^{-2}) and nitrogen (kg ha^{-1}) from the various irrigation waters applied to the nine treatment plots of aquabrine (AQ), reject brine (RO), groundwater (GW) and low salinity (EC10), by two types of emitters of bubblers (BUB) and pressure compensated drippers (PCD), along with the average leaching losses measured over the 2022–2023 season from the four tension-fluxmeters within each of the plot. The AQ-BUB* treatment applied 30 mm day^{-1}, whereas the AQ-BUB treatment added just 15 mm day^{-1}. The crop evapotranspiration over the season, ET_c (m^3 m^{-2}), was modelled following Al-Tamimi et al. (2022)

Treatment	ET_c (m^3 m^{-2})	Salt Load (kg m^{-2})	Salt Lost (kg m^{-2})	Nitrogen Load (kg ha^{-1})	Nitrogen Lost (kg ha^{-1})
AQ-BUB*	1.61	117.0	143.5	325	248
AQ-BUB	1.72	51.6	58.4	715	769
AQ-PCD	1.47	62.3	53.2	429	516
EC10-BUB	1.69	35.0	34.1	456	442
EC10-PCD	1.46	25.5	35.3	281	381
GW-BUB	1.89	37.5	89.3	312	339
GW-PCD	1.46	51.2	45.8	304	181
RO-BUB	1.72	71.5	96.1	343	270
RO-PCD	1.47	51.2	28.2	197	292
Averages	**1.61**	**55.9**	**64.9**	**374**	**382**

(C_{out}, Eq. 3), and the total loading of solutes onto groundwater of C_{out}. $(I - ET_c)$ (kg m^{-2}).

These simple mass-balance calculations can be used to develop a heuristic model to link land-management practices to groundwater quantity and quality. We use different units for salt concentration, and each is fit-for-context. The conversion from EC units in dS m^{-1} can be obtained by dividing mg L^{-1} by 640, and of course 1 mg L^{-1} is 10^3 mg m^{-3}. It is pertinent to outline the scale of salinity we describe here. Freshwater is generally taken as being less than 1,000 mg L^{-1} (or 1.5 dS m^{-1}), brackish water is between 3,000 and 10,000 mg L^{-1} (4.7–15.5 dS m^{-1}), and seawater is around 30,000 mg L^{-1} (47 dS m^{-1}). In Fig. 4, it is found that the salt concentration of the leachate can exceed 100 dS m^{-1}, which is about two times the salinity of sea water.



Heuristic modelling of impacts on groundwater

Heuristic modelling

From these new observations of salt leaching we developed a heuristic model to predict the impact of land management on the salinity of groundwater. A heuristic approach was adopted so that the simplicity of the model's parameterisation to assess environmental impacts would be commensurate with the simplicity of the findings from our empirical observations of the gross economic productivities of using saline waters to irrigate halophytes. If our purpose were to examine the hydrogeological dynamics of salt in aquifers

Fig. 3 The nitrate concentration (C, mg L^{-1}) of the leachate measured over 2023 in drainage under plots of a *Salicornia bigelovii* crop in the United Arab Emirates by tension drainage fluxmeters (DFM) under irrigation with aquabrine (AQ, top), reverse osmosis water (RO, upper middle), groundwater (GW, lower middle), and water at EC 10 dS m^{-1} (EC10, bottom, for just 2023). Water was applied either through bubblers (BUB) or pressure compensated drippers (PCD). There was an extra aquabrine treatment (AQ-BUB*) where water was applied at 30 mm d^{-1}, rather than 15 mm d^{-1}. The switch between low salinity irrigation water to the treatment waters was on 22 February 2023

under irrigated halophytes. then other modelling systems would have been apposite. One such approach would have been the use of detailed mechanistic models such as SALTMOD (Oosterbaan 1997). The onerous task of such parameterisation would, however, limit our ability to assess the benefit-cost trade-offs of using saline irrigation for food and fodder production.

A heuristic method is a modelling approach for finding a simple, yet practical, solution to a problem. The heuristic approach was extended to environmental issues by Harte (1988) in his book "Consider a spherical cow: A course in environmental problem solving". Here, in order to solve heuristically the problem of predicting the impact of irrigation on groundwater quantity and quality, we are guided by a strong evidence-base of field measurements, and the underpinning law of mass conservation. Consider an aquifer and overlying unsaturated vadose zone as a rectangular prism (Fig. 5). Our representation here of a desert aquifer mimics the mixing-cell model that Pauloo et al. (2021) used to assess anthropogenic basin closure and groundwater salinization (ABCSAL) in the Tulare Lake Basin in California. In a subsequent paper, Wang et al. (2024) used detailed mechanistic modelling of aquifer dynamics to assess hydrological basin 'openness' within the framework of ABSCAL, namely the degree to which the balance between groundwater inflows and outflows offset groundwater extractions. However, in our endorheic case, aquifer inflows and outflows are negligible relative to irrigation demands. Our goal is to derive, via heuristic modelling, a closed-form analytical expression for ABSCAL that would apply to irrigated agriculture in this hyper-arid desert region.

Modelling groundwater impacts

We assume that only some fraction of the total area of land above a given closed-system aquifer (T, ha) is actually irrigated, namely A (ha) (Fig. 5). Outside of the total area, T, there are no-flow boundaries at the base and surface, as well as along the perimeter walls. The only flows are of water and solutes drawn from, and returned to, the aquifer under the irrigated area. A seasonal equilibrium across the whole aquifer domain is assumed, so that when impacts are predicted for the aquifer directly under area A, the wider impact for the whole aquifer can be assessed using a multiplier of A/T. The depth of the unsaturated vadose zone above the aquifer is d (m). The depth of the saturated layer at any time t is h_t. Over one year, the change in the depth of the saturated layer is Δh. The mobile water content for flow through the unsaturated vadose zone is Θ_V (m^3 m^{-3}). The volumetric water content of the saturated aquifer is Θ_A (m^3 m^{-3}). The phreatic surface separates the vadose zone and aquifer. As groundwater extraction continues this water table drops, whereas the respective water contents of the unsaturated zone and groundwater are considered time invariant. The local impact of any capillary fringe is ignored, which for these desert sands is reasonable. We are working in the hyper-arid deserts of Abu Dhabi, where reference evapotranspiration ET_o is greater the 2000 mm y^{-1}, and there is generally less than 50 mm of rainfall. Hence, we ignore rainfall. This lack of rainfall also means that natural recharge of groundwater by drainage is negligible, and we also ignore natural flows of water into and out of the groundwater system.

The annual amount of water drawn from the aquifer to irrigate the crops within A is I (m^3 m^{-2}). The seasonal evapotranspiration by the crop over the year is ET_c (m^3 m^{-2}). The drainage of water back to the aquifer, which carries with it solutes, is therefore $I\text{-}ET_c$.

The concentration of solutes of either salt or nitrogen leaving the groundwater in the irrigation water is C_{out} (kg m^{-3}), and the concentration of solutes returning to the groundwater in the drainage $I\text{-}ET_c$ is C_{in} (kg m^{-3}). Nitrogen fertiliser is applied at the rate of N_i (kg ha^{-1}), and the loss of nitrogen to the system via crop uptake and removal by harvest is N_o (kg ha.^{1}). As noted above, we consider that in a mass balance sense N_o is balanced by the mineralised nitrogen-fraction from the organic fertiliser N_i, such that the loading of nitrogen brought in by the irrigation water is lost back in total by drainage eventually back to groundwater.

To maintain analytical simplicity for heuristic clarity, we necessarily assume a quasi-steady-state system. However, in reality there is a temporal dimension to the transmission of leachate impacts down through the unsaturated vadose zone to the saturated depth of groundwater. The initial time t_0 in our groundwater-quality predictions is essentially when the leachate first arrives from above, having traversed the vadose zone. The drainage through the vadose zone is $I - ET_c$ such that velocity of transmission is $v = (I - ET_c)/\Theta_V$, where Θ_V is the mobile water-content of the vadose zone. For our experiments with *Salicornia* the average drainage rate was found to be 2.23 m^3 m^{-2} y^{-1} (Table 1). For a typical vadose zone with Θ_V = 0.3 (Al Tamimi et al. 2023c), v would be about 7.4 m y^{-1}. The transit time for the piston displacement of solutes to reach the saturated layer will establish the initial time $t_0 = d / v$ when leached solutes first reach back to the groundwater. This t_0 will be the origin

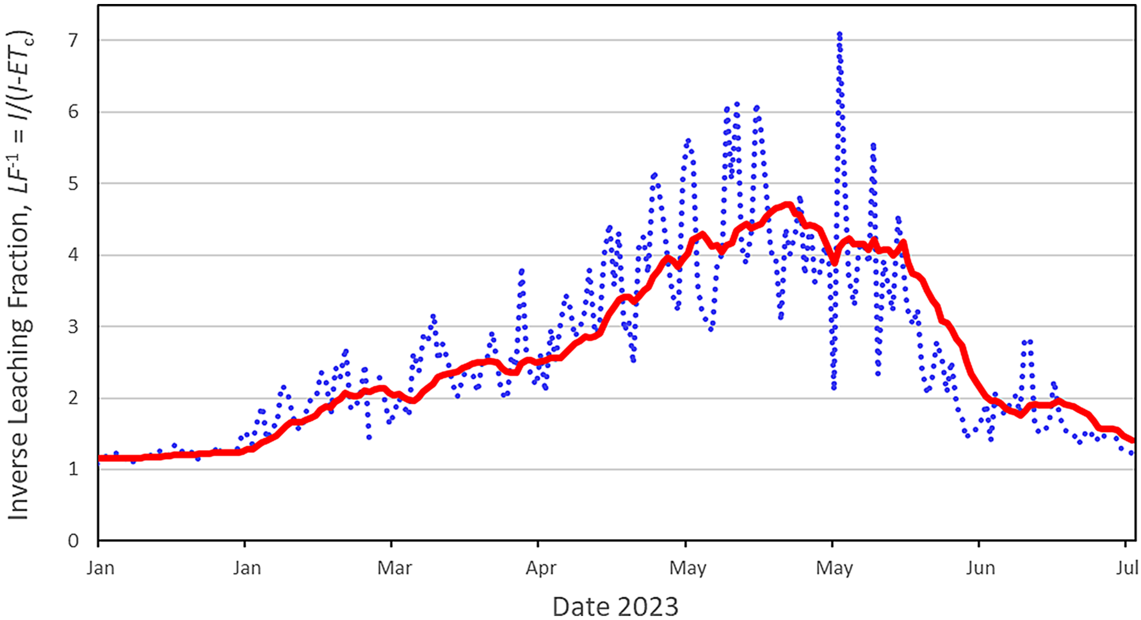
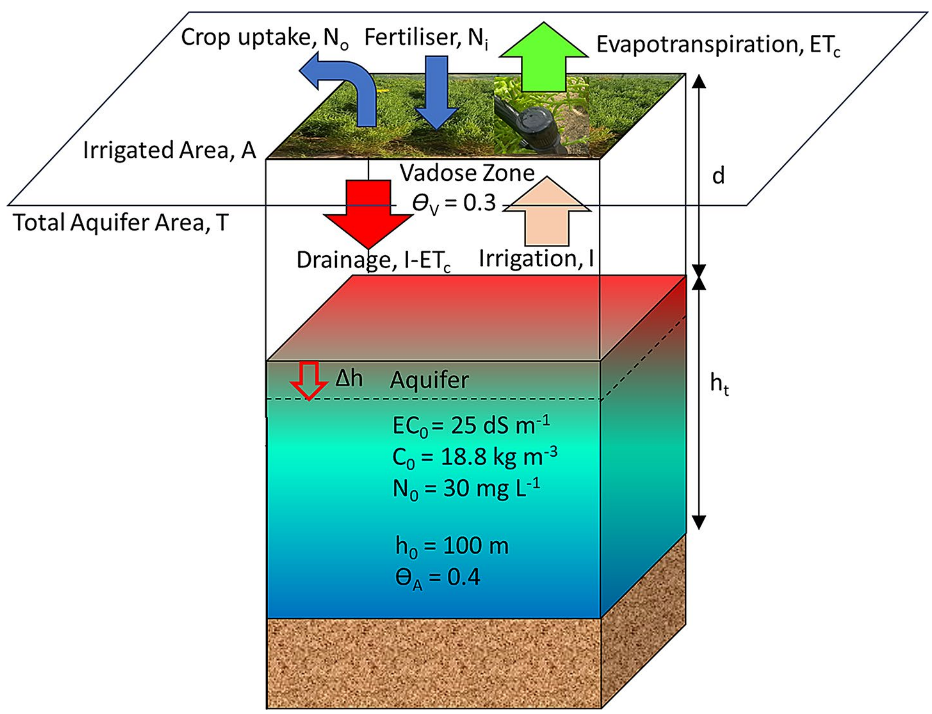


Fig. 4 Average daily values (- - -) across the 9 plots of the inverse of the leaching fraction, LF, namely $LF^{-1} = I/(I - ET_c)$ where I is the measured daily amount of irrigation applied ($m^3\ m^{-2}$) and ET_c is the modelled crop evapotranspiration ($m^3\ m^{-2}$). This inverse leaching fraction would correspond to the ratio of the solute concentration in the leachate, C_{out} ($kg\ m^{-3}$), divided by the concentration in the irrigation water, C_{in}, so $LF^{-1} = C_{out}/C_{in}$. Here I was measured daily for each of the 9 plots, and ETc was found using the crop-factor model proposed by Al-Tamimi et al. (2022). The line (—) is the 14-day running mean of the daily values

Fig. 5 A schematic representation of a closed-system aquifer covering a spatial land area of T (m^2), within which an agricultural area of A (m^2) is irrigated with I ($m^3\ m^{-2}$) of groundwater drawn from the aquifer. The evapotranspiration from the crop is ETc. Solutes are leached back to the aquifer through the unsaturated vadose zone in the drainage of ($I - ET_c$). The crop is fertilized with an amount of nitrogen N_i (kg-N ha^{-1}), and the amount of nitrogen taken off by the crop is N_o (kg-N ha^{-1}). The depth of the unsaturated vadose zone above the aquifer is d (m) with a vadose-zone mobile-water content of Θ_V ($m^3\ m^{-3}$). At any time t, the depth of the saturated layer is h_t (m), and every year it changes by Δh. The volumetric water of the saturated layer of the aquifer is a time-invariant Θ_A ($m^3\ m^{-3}$)



of the abscissa ($t=0$) in Fig. 6. Typically the depth of the vadose-zone layer in the UAE is about 50 m, such that t_o = 6.7 y, being the delay for the solutes to first reach groundwater following the initiation of irrigation using saline water from the aquifer.

Aquifer volume

For the aquifer over the growing season of one year, an amount I ($m^3\ m^{-2}$) of water is drawn to irrigate the crop, and an amount ($I - ET_c$) ($m^3\ m^{-2}$) is returned to groundwater as drainage. Over the year, the net loss of water to the groundwater system is ET_c. Volumetrically, this comprises a net-usage volume ET_c. A (m^3), such that this loss results in an annual volumetric removal of water from the aquifer of A. Δh. Θ_A (m^3). Combining these means that the annual change in the depth of the saturated layer is $\Delta h = ET_c / \Theta_A$ (m). It is assumed that the annual usage of irrigation (I) and evapotranspiration (ET_c) remain unchanged. At time $t > t_o$, after a number of years n_t, this can be generalised to

$$h_t = h_0 - \frac{n_t \, ET_c}{\theta_A} \tag{4}$$

Here h_o is the initial depth of the saturated layer (m) at time t_o. Water would have been extracted prior to t_o, and this would have commenced immediately as irrigation began. The total depth-amount of water stored in the aquifer initially at t_o is h_o. Θ_A (m^3 m^{-2}), and when consumed at an annual rate of ETc the number of years to exhaust this groundwater stock is $n_f = h_o \Theta_A / ET_c$.

For illustrative purposes, we take an initial saturated depth of h_o = 100 m at t_o, which is representative of the aquifers surrounding Al Ain, and along the Liwa oases (EAD 2018). We assume that the saturated water content of the aquifer is Θ_A = 0.4 with ET_c = 1.61 m^3 m^{-2} y^{-1} (Table 2). The time course of the draw-down of the aquifer is shown in Fig. 6, with the time-to-exhaustion $n_f = 24.8$ years. This is an average decline in the groundwater level of 4 m y^{-1}, which is comparable to the 2.75 m y^{-1} decline observed around Al Ain at Al Wagan and Remah (EAD 2018). This assumes that all of the land-surface is irrigated, namely $A = T$. This will generally not be the case, so the draw-down result (Eq. 4) will need to be multiplied by $A / T < 1$ to account for the fractional irrigation of the land above the aquifer. If only one third of the land area were irrigated, then n_f would be 74.5 years, with rates of water-table decline of around 1.3 m y^{-1} (MOEW 2015).

Groundwater quality

The two solutes of concern we consider here are salt and nitrate. The mass of solute in the aquifer at any time t is M_t (kg), and this will be equal to the concentration, C_t (kg m^{-3}, or kg L^{-1}) times the aquifer volume V_t (m^3, or L): $M_t = C_t$. V_t. The initial mass of solute in the groundwater is $M_0 = C_0$. V_0. As seen in Table 2, the mass of salt will be time invariant, as the salt taken out in the irrigation water is returned in the drainage. Likewise, in a mass-balance sense, the nitrate mass in the groundwater volume and vadose zone will under current practices be unchanging with time, as the nitrogen removed by the harvest of the crop is essentially equivalent to that mineralised from the organic fertiliser applied at sowing. Hence the nitrate removed by the irrigation water is eventually returned in sum in the drainage (Table 2).

Solutes in drainage through the vadose zone will begin to arrive back in the groundwater some time t_o after the leaving the rootzone in drainage. It is assumed that the solutes move via piston displacement through the unsaturated soil of the vadose zone (Al Tamimi et al. 2023b). After the

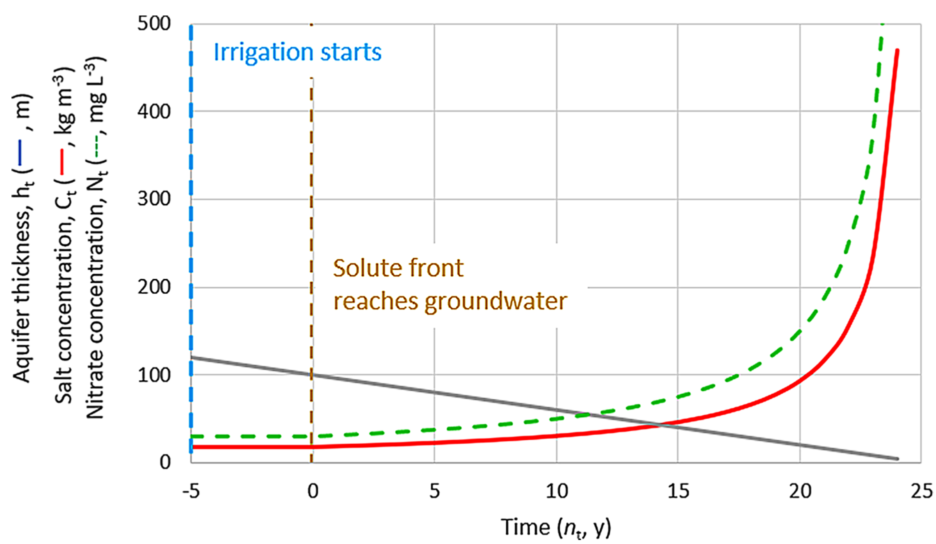


Fig. 6 The predicted temporal draw-down in the aquifer depth with the number of years using the prediction of the depth of the saturated layer h_t from Eq. (1) (—). The initial aquifer thickness is assumed to be h_o = 100 m and saturated water content Θ_A (m^3 m^{-3}), and the annual rate of evapotranspiration from the irrigated crop is 1.61 m^3 m^{-2}. The time course in the concentration of salt and nitrate in groundwater predicted by Eq. [3] is given for salt (—) and nitrate (----). The initial values were for salt $C_0 = 18.8$ kg m^{-3}, and nitrate $N_0 = 30$ mg L^{-1}. Five years after irrigation starts the solute front reaches groundwater. The origin of the abscissa, $t = 0$, is time t_o when the impacts of solute leaching are first felt in groundwater at the base of the unsaturated vadose zone of depth d. Note that for graphical convenience the numerical values on the vertical axis refer to different units in relation to the parameters graphed of aquifer depth (m), salt concentration (mg m^{-3}), and nitrate concentration (mg L^{-3})

initiation of irrigation, but prior to t_o, the water level will begin to drop straightaway according to Eq. [4]. However, the leachate consequent upon this irrigation will not arrive at depth d until time t_o, such that the concentration of solutes in the aquifer will remain unchanged at C_o for $t < t_o$. After t_o, because the plants are not taking up the solutes drawn up in the irrigation water, they are returned in sum to the aquifer via the leachate. We assume that over the timescales that are appropriate here, the influent solutes become well mixed into the aquifer. Density differences will aid this mixing, as suggested by Al-Tamimi et al. (2023c). From well-mixed mass conservation, we can therefore assume $M_t \approx M_o \ \forall \ t > t_o$, and the concentration of solute in the groundwater at any time after t_o is now:

$$C_t = C_0 \frac{V_o}{V_t} = C_0 \frac{A \, \theta_A \, h_0}{A \, \theta_A \, h_t} = C_0 \frac{h_0}{h_t} \quad (5)$$

where h_t is given above by Eq. [4], which upon substitution results in

$$C_t = \frac{C_0}{\left(1 - n_t \left[\frac{ET_c}{\theta_A \, h_0}\right]\right)} \quad (6)$$

From Eq. [6] it is possible to predict the time-course in the concentration of solute in the groundwater as a function of the number of years, n_t, since the beginning of irrigation to supply water to a crop that loses water to the atmosphere at an annual rate of ET_c. This equation provides an analytical closed-form expression for the term of 'salt evapoconcentration' coined by Pauloo et al. (2021).

As can be seen from the form of Eq. [6], the time- rise in concentration is a hyperbola for $t > t_o$. The initial rise in concentration is gradual, however as time progresses the concentration rises rapidly and then asymptotes to infinity at the time to groundwater exhaustion. A useful ready-reckoner for the rise in solute concentration, akin to an inverse half-life, would be the numbers of years required when $t > t_o$, to realise the first doubling of the solute concentration in groundwater, n_2 (y). From Eq. [6] this can be found as

$$n_2 = \frac{\theta_A \, h_0}{2 \, ET_c} \quad (7)$$

Our simple model predicts a doubling of the salinity in 12.5 years, assuming $A = T$. This rate of salinity rise of 1.5 kg m^{-3} y^{-1} is not dissimilar to the rises of around 2.3 kg m^{-3} y^{-1} shown in regions of the UAE by EAD (2018) and MOEW (2014,2015). Our predictions show a further doubling in just the 5 years after the first 12.5 years of leachate returns (Fig. 6). This analysis is for an aquifer underneath an area of land that is fully irrigated $A = T$. Clearly this is not the case (EAD 2018. Especially p77), namely $A < T$, then the concentrations will need to be multiplied by A / T.

Well before the time-to-exhaustion (n_f, years) of the groundwater resource (Eq. 4), the utility value of the aquifer will have become worthless because of the high salinity, such that it would be neither practical to desalinate, nor useful for irrigating halophytes. Changed practices will be needed to ensure the sustainability of land use and the protection of groundwater.

It is likely that the current trend of using inland desalination units for irrigated agriculture will continue. Dawoud et al. (2024) noted, however, that there are three challenges that need to be assessed and evaluated. The first is to discharge the brine into an evaporation bund. The second is what we are doing here, use the water to irrigate halophytes, although we show that this can have impacts on groundwater. The third is to develop zero level discharge (ZLD) desalination units. The latter would allow for the desalinated groundwater to be used for high value crops and eliminate the need for discharge of brine onto land. New technologies, even using renewable energy sources, are being developed apace for minimal and zero liquid discharge desalination technologies (Prado de Nicolás et al. 2023). These would help protect groundwater quality by reducing salt leaching. To protect groundwater quantity, it is important that policies be developed, and regulations implemented to limit groundwater extraction to levels commensurate with whatever groundwater recharge rates exist. Sadly, the acceleration of groundwater deepening highlights this urgent need for more effective measures to address groundwater depletion (Jasechko et al. 2024). They then presented specific cases of where depletion trends have been reversed following policy changes, and this highlights the potential for depleted aquifer systems to recover, given natural recharge. To assess the ability of these desert aquifers to recover will not only require aquifer monitoring, but also endorheic quantification of the 'basin openness' of ABCSAL using groundwater dynamics.

Conclusions

Our experiments have produced data and results that highlight the economic value of using nitrogen-rich saline waters, either from groundwater or reject brines from desalination units, to irrigate the halophytic crop *Salicornia bigelovii* for food, fodder, and fuel. The greatest benefit was achieved using pressure compensated drippers to supply the water to the plants. Our measurements of drainage and leaching under the crop showed that, in sum, all of the salt and nitrogen drawn from the groundwater to supply the irrigation, was returned back to the aquifer in the drainage water as leachate.

From this comprehensive evidence-base of measurements, a simple heuristic model of groundwater quantity and quality was developed to infer the environmental impacts of irrigating crops with saline and high-nitrate groundwater in a hyper-arid environment. The key parameters needed to parametrise this simple model are the fraction of the land surface above the aquifer that is irrigated, the initial depth of the saturated thickness of the aquifer, the saturated water content of the aquifer, and the annual rate of evapotranspiration from the crop, which can be found using the crop-calculator model of Al-Tamimi et al. (2022). The model is simple and straightforward to parameterise. It is easily understood and could be useful for assessing the impacts and trade-offs of policy developments and regulation options.

Tables.

Author contributions MT, Conceptualisation, Methodology, Investigation, Writing, Review, Editing SG, Conceptualisation, Methodology, Investigation, Writing, Review, EditingWD, Conceptualisation, Methodology, Investigation, Writing, Review, Editing AM, Conceptualisation, Methodology Investigation, Writing, Review, EditingDL, Conceptualisation, Resources, Writing, Review, EditingKA, Conceptualisation, Resources, Writing, Review, EditingMD, Conceptualisation, Funding, Conceptualisation, Investigation, Writing, Review, EditingPK, Conceptualisation, Writing, Review, EditingPK, Conceptualisation, Writing, Review, EditingLK, Conceptualisation, Methodology, Investigation, Writing, Review, EditingAM, Methodology, Investigation, Writing, Review, EditingBC, Conceptualisation, Methodology, Investigation, Writing, Review, Editing.

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Data availability Data will be made available on request.

Declarations

Competing interests The authors declare no competing interests.

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