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Exploring desertification in coastal groundwater-dependent agriculture through an integrated simulation model

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Abstract

Groundwater degradation is a global problem linked to irrigation agriculture and aggravated by climate change. In drylands, where aquifer recharge is low, irrigation has emerged as an engine of economic growth. This problem falls under the paradigm of desertification, as it fits the definition of this complex problem in that the degradation of drylands is due to climatic variations and inappropriate human activities. Land Degradation Neutrality (LDN), the response of the United Nations Convention to Combat Desertification to the lack of progress in tackling desertification, is integrated into Sustainable Development Goal 15.3 and provides the adequate framework for implementing effective solutions. LDN prioritizes prevention strategies, and early warning systems coupled to integrated simulation models is a sound approach. We analyze the dynamics of a coastal aquifer in southern Spain, a "desertification landscape" according to the Spanish National Action Plan to Combat Desertification. For this purpose, we have (i) adapted a generic desertification model that considers socio-ecological systems as a particular case of predator-prey systems; and (ii) coupled to this model a risk analysis to calculate the probabilities of groundwater salinization under the current scenario of water resources use, driven by the expansion of greenhouse agriculture supported by external water transfer. The risk of desertification is close to 100%: groundwater salinity is 40 dS m^{-1} (well above tomato tolerance, 3.5 dS m^{-1}), and 2.4 Mm^3 yr^{-1} water transfer is needed to support the 631 ha of greenhouses. This worrying result suggests that complimentary solutions should be promoted to deactivate the ongoing process of desertification. Among them, we propose reclaimed water, diversifying the economy, or promoting crops adapted to aridity. This simulation framework shows how to explore the future of a socio-ecosystem under current scenarios and others that consider climate change, the energy crisis, or the impact of alternative solutions.

Keywords

Groundwater degradation; desertification; early warning system; System Dynamics; LDN; southern Spain

1. Introduction

The vast majority of the world's freshwater reserves are underground. Indeed, about 98–99% of total water reserves is groundwater, which amounts to about 8–10 Mkm^3 of freshwater [1]. It is not easy to know precisely these volumes, nor has it been historically easy to exploit them. Until the 1960s, its use was limited by the technological incapability to drill deep boreholes and install pumping equipment. Several factors came together to popularize groundwater exploitation. Geological knowledge, the spread of pumping equipment, rural electrification, and cheap energy led to an explosion in its use [2]. In addition to its abundance, what makes this water source really useful is its reliability, especially in drylands, where surface water is scarce and has an irregular regime. In addition, groundwater tends to be of good quality (with little or no treatment) and available where and when it is needed. This local accessibility has earned it the label of a "democratic" resource [1]. All these factors contribute to their increasing role in the water economy.

The main water use is agricultural, accounting for 70% (2,800 km^3 out of 4,000 km^3) of freshwater consumption worldwide (FAO, 2021). The expansion of irrigated agriculture into semi-arid areas with scarce precipitation and surface water sources has greatly increased the reliance of irrigated crops on groundwater withdrawal [4]. These were estimated at 820 km^3 yr^{-1} in 2018, a 19% increase relative to 2010 [5]. They represent over 30% of agriculture freshwater withdrawals and continue to grow at around 2.2% per year [5]. The link between intensive agriculture and groundwater depletion is evident: Irrigated areas under stress correlate strongly with intensive groundwater use and depleting aquifers [6]. In addition, 2 billion people currently depend on groundwater [7] as their primary source of drinking water. Globally, it is estimated that between one-fifth and one-third (600–1,100 km^3 yr^{-1}) of the total water withdrawals are drawn from groundwater [7,8].

Only a minor part of the enormous groundwater reserves is more or less regularly replenished by aquifer recharge. Aquifer recharge decreases with aridity, becoming virtually nil in hyper-arid areas [9]. When groundwater was recharged before 12,000 BP we refer to 'fossil groundwater'. For instance, the Nubian aquifer is considered the largest groundwater reserve in the world [10] with 542,000 km^3 [11] and is the consequence of rainfall recharge occurred between 20,000 and 49,000 years ago [12]. Massive withdrawals have led to a global groundwater crisis [7]. The swift decline of large bodies of water, such as the Ogalalla [13] and the Arabian [14] aquifers, are just some of the most remarkable cases. The shocking extent of non-renewable groundwater use become evident to the larger public since 2009, when the first analyses of the Gravity Recovery and Climate Experiment (GRACE) mission [15] were published [16], and after the first global assessment of groundwater depletion [17]. The problem is both quantitative and qualitative, evidenced by the falling water tables [18] (boreholes now can be drilled over 2 km depth to reach groundwater [7]) and contamination by salinization and percolation of pesticides, fertilizers and other chemicals [5]. In addition, water depletion triggers seawater intrusion in coastal aquifers [19], rendering groundwater reserves unusable.

Groundwater degradation in drylands can be considered a case of desertification (Martínez-Valderrama, Guirado, & Maestre, 2020b; Martínez Fernández & Selma, 2004), as it is defined by the United Nations Convention to Combat Desertification (UNCCD) as: "land degradation in arid, semi-arid and dry sub-humid areas resulting from various factors, including climatic variations and human activities", where land degradation is "the reduction or loss of the biological or economic productivity." (UNCCD, 1994). Even though some of the most emblematic desertification cases, such as the Aral Sea [23] –which gives a syndrome name under which similar cases around the world are grouped [24]–, water degradation is excluded from global desertification maps (Prince, 2016). Desertification cartography has historically show soil and vegetation degradation (e.g., Global Assessment of Human-induced Soil Degradation (GLASOD) project [26]). The emergence of irrigated fields, greening the landscape, responds more to the idea of a barrier against desertification than to that of something problematic. This association of images gives rise to misconceptions (Prince & Podwojewski, 2019) and ignores the source of problems linked to overexploitation of water resources.

One of the "desertification landscapes" spotted in the Spanish Action Plan to Combat Desertification (PAND) is "Irrigated areas at risk of desertification" (MAGRAMA, 2008). In Spain drylands occupies 74% of the territory [30] and 20% is already desertified (Martínez-Valderrama et al., 2016; Sanjuán et al., 2014). One third of all irrigated land in the European Union is located in Spain [32], with 3.8 Mha [33], which uses 65% of available water resources (FAO, 2021). Between 2000 and 2016, groundwater use in the Spanish agriculture has increased from 4.08% to 22.4% [34], which means that agriculture uses 73% of groundwater —about 7 Mm^3 yr^{-1}— (De Stefano, Martínez-Cortina, & Chico, 2012). The decline and deterioration of groundwater bodies and their associated [36–39] have continued to degrade due to the constant increase in irrigation surface [40].

Despite the scientific value of PAND, it has not been followed by effective actions to halt desertification (Martínez-Valderrama et al., 2022). The increase in the frequency and intensity of droughts, particularly in the Mediterranean region and southern Africa [42], the high profitability of the irrigated horticultural agriculture demanded in the European markets (Martínez-Valderrama, Guirado, & Maestre, 2020a), have consolidated and expanded the "Groundwater-dependent coastal agriculture", as pointed out in the PAND update, the National Strategy to Combat Desertification in Spain (MITERD, 2022). It is a variant of the desertification landscape mentioned above, the study of which is the subject of this paper. The cross-cutting nature of desertification asks for a multidisciplinary framework to approach it. System Dynamics (SD) is postulated as a very suitable tool for this purpose (Martínez-Valderrama, Ibáñez, Gartzia, & Alcalá, 2021). The reason is that desertification results from the dynamic and generally complex interaction between a wide range of factors and processes, specific to a diverse fields of knowledge [45–48]. Specifically, our objective is to study the expansion of greenhouse agriculture in a small coastal basin in the coast of Granada province in southern Spain (Fig. 1) and its impact on groundwater by (i) implementing an integrated simulation model for the Gualchos Stream basin (GSB); (ii) calculating the desertification risk by simulation of the former model.

Next section presents the study area and details the integrated simulation model and risk analysis. Results are presented in Section 3 and discussed in Section 4. Finally, the main conclusions drawn from the study are presented in Section 5. Complementary Figures, the complete set of equations of the model, and the parametric values and steps followed for the calibration of the model are included as Supplementary Material (SM) of this paper.

2. Material and Methods

2.1. Area of study

In southern Spain, the coast of Granada province is 71 km long (Fig. 1a) and has 129,588 permanent inhabitants [49], and around the double during summer due to seasonal tourism [50]. From a geographical point of view, the coast of Granada includes several ephemeral and semi-permanent rivers and streams that drain the southern versants of coastal sierras (e.g., Almijara, Guájares, Lújar, Contraviesa) with peaks of almost 1,900 m above sea level (m a.s.l.) [51]. The proximity of the mountainous elevations to the coastline makes these water courses steeply sloping with small plains at their outlets. The central area is occupied by the Guadalfeo River Delta. This unique permanent river drains the southern and southwestern versants of Sierra Nevada Mountains with peaks over 3400 m a.s.l. The steeped topography, coupled with irregular rainfall and sparse vegetation, means high water erosion [52].

This region belongs to the Alpujárride tectonic complex from the Internal Domain of the Alpine Betic Cordillera. The area is tectonically active as a consequence of the convergence between the African and Eurasian Plates, which ended with the collision of the Internal and External Betic domains during the early Miocene [53]. From a regional hydrogeological point of view, the geological formations can be catalogued as (i) Triassic to Paleozoic schist, phyllite, quartzite and graphite-rich schist forming the regional low-permeability metapelitic bedrock; (ii) Triassic carbonates (dolostone, limestone and marble) forming productive regional aquifers; and (iii) Miocene to Quaternary carbonates and detritic sediments (alluvial, colluvial and glacis) varying from low- to high-permeability formations forming small aquifers of local relevance (IGME & Junta de Andalucía., 1998; IGME, 1988). These detritic aquifers consist mainly of Quaternary alluvial coarse sediments (gravel and sand) deposited in the riverbeds and outlets of the ephemeral and semi-permanent water courses that drain from west to east the southern versants of coastal sierras [55]. These are generally small (many do not exceed 5 km^2 in surface and 70 m in thickness) and high-yielding (coarser granulometries induce high hydraulic conductivities) aquifers (Calvache and Pulido_Bosch, 1996; Calvache and Pulido-Bosch, 1997; Pulido-Leboeuf, 2004).

The Castell de Ferro aquifer (CFA) is one of these small, high-yielding alluvial coastal aquifers. It has about 3 km^2 in surface and 60 m in maximum thickness in the coastal fringe, and extends from the mid-slope to low-lying areas of the Gualchos Stream valley and its most important tributaries (Fig. 1d). The aquifer alluvial length is about 8.5 km and its width increases from 200 m in the mid-slope area to 700 m in the low-lying area [55]. Surface runoff in the GSB is usually nil for most of the year, and only in extraordinary rainfall events will the stream function in a torrential manner. In the GSB, average annual temperature is 18.9°C and potential evapotranspiration is around 940 mm yr^{-1} [58]. Average annual precipitation is 419 mm, ranging between 178 to 802 mm yr^{-1} over the period 1980/81–2011/12 (Junta de Andalucía, 2016). Hence, the CFA has scarce groundwater resources, estimated at 3 Mm^3 yr^{-1}. Outflows are mainly from groundwater pumping, estimated at 2 Mm^3 yr^{-1}, and groundwater discharge to the sea, estimated at 1 Mm^3 yr^{-1} (Benavente et al., 1988; IGME & Junta de Andalucía., 1998). The coastal sector of the CFA includes high-salinity groundwater and records events of seawater intrusion during dry seasons when lower aquifer recharge rates and higher groundwater pumping rates occur [58]. Groundwater salinity in the GSB mid-slope areas and summits is also noticeable due to higher atmospheric salinity contribution in the region [60]. The groundwater quantity and quality statuses of the groundwater body is poor, according to the Hydrological Plan of the Andalusian Mediterranean Watershed (AMW), the water authority [61].

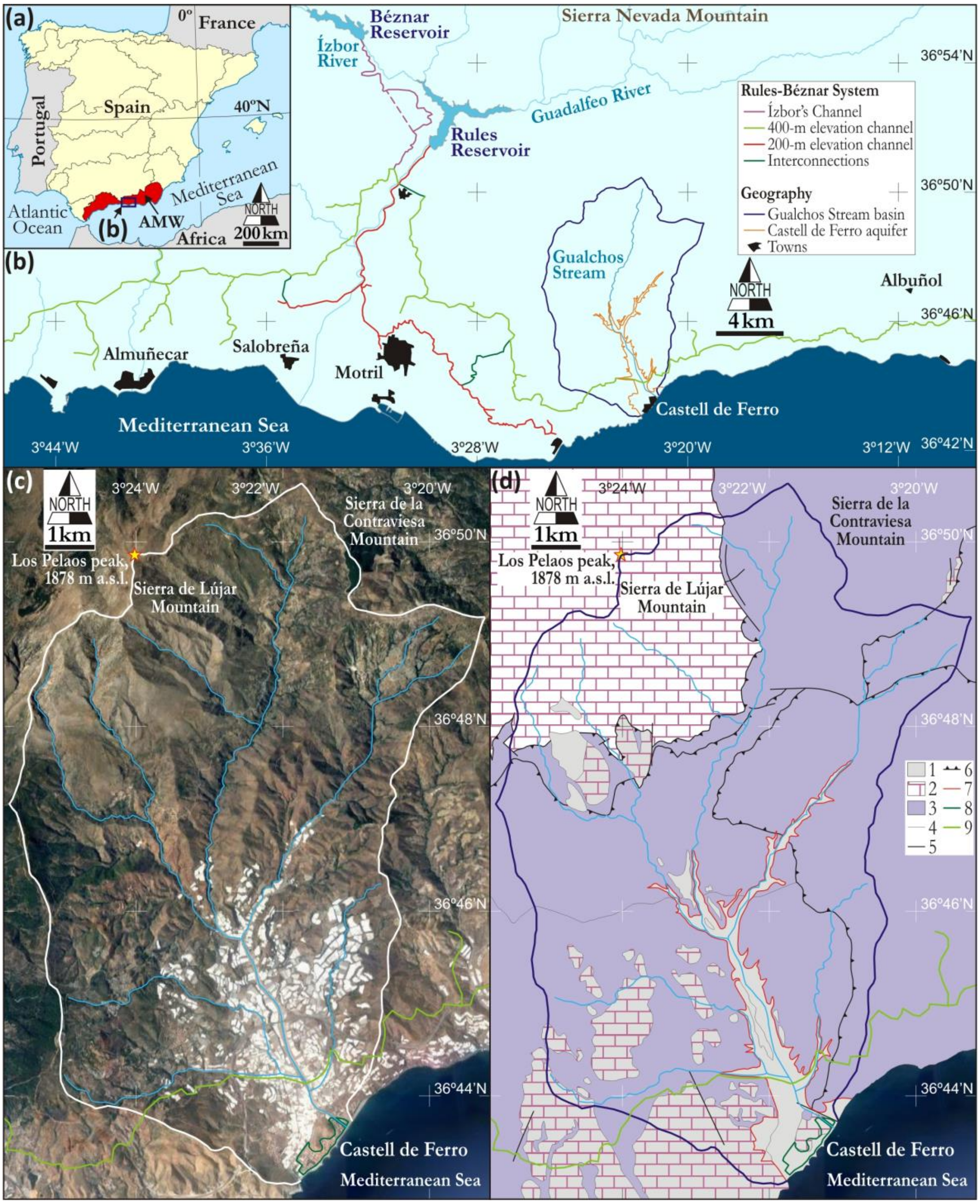

Figure 1. (a) Location of the study area in the Andalusian Mediterranean Watershed in southern Spain. (b) The Rules–Béznar System (RBS) of hydraulic infrastructures (reservoirs and channels) in the coastal fringe of Granada province. (c) Satellite image of the Gualchos Stream basin (GSB) in 2021 (source: Google Earth), showing the greenhouses occupation at mid-slope and low-lying areas. (d) Hydrogeological map of the GSB showing the geological formations catalogued as aquifer (1—Quaternary

alluvial and glacis; 2—Triassic dolostone, limestone and marble) and low-permeability metapelitic bedrock (3—Triassic to Paleozoic schist, phyllite, quartzite and graphite-rich schist), as well as other geological, hydrogeological, hydraulic and geographic features (4—undifferentiated geological contact; 5—fault; 6—thrust; 7—Castell de Ferro alluvial aquifer; 8—urban area; 9—the 400-m elevation channel from RBS). Modified from (IGME & Junta de Andalucía., 1998; IGME, 1988).

As well as in the rest of the Mediterranean coast, agriculture land uses and anthropization have been historically very important to build the landscape identity [62]. The coastal region of Granada bases its intense economic activity on two main pillars, tropical and extra-tropical agriculture and tourism. The extraordinary spatial expansion of greenhouses has led to the saturation of the traditional agricultural landscapes of the coastal plains [50,63]. The origin of this change in land use is found in the Campo de Dalías in Almería province, some 50 km east of Castell de Ferro. There, field trials carried out in 1954 successfully developed a cultivation technique known as "enarenado" (sanding-plot). Protecting this artificial soil from the persistent wind of the area using techniques and skills associated with the long-established table grape industry and traditional horticulture in fertile coastal plains and valleys, emerged the precursor to the area's modern greenhouses [64]. The economic success of this intensive model, known as the 'Almeria Miracle' [64,65], triggered its expansion in nearby eastern (first) and western (later) areas, taking advantage of the combination of favorable climatic conditions for horticultural crops (mild winters and plenty of sunny hours) and the existence of groundwater.

In the GSB greenhouse agriculture started at the end of the 1970s [66] and since then it has been growing to reach 631 ha in 2019 [67] (see SF 1). The visual footprint of greenhouse colonization is evident (Fig. 1c), as is the impact on the groundwater resource. From 1982 onwards, groundwater quality showed clear signs of qualitative deterioration during dry periods [58]. Nowadays, irrigation is possible thanks to the water transferred from the Rules Reservoir (Fig. 1b), which since 2003 has been supplying water to this and other areas with the same problem of seawater intrusion. The Rules Reservoir together the Béznar Reservoir form the the Rules–Béznar System (RBS) (Fig. 1b), a hydraulic infrastructure of reservoirs and channels devoted to regulate and transfer the surface water resource generated in the southern (Guadalfeo River) and southwestern (Ízbor River) versants of Sierra Nevada Mountains to the coastal fringe of Granada province. The greenhouses produce mainly tomatoes and cucumbers, thanks to the exogenous supply of water from the RBS, which are largely exported to European markets [68].

2.2. SD and the Aquacoast model

SD aims to build dynamic, complex and comprehensive models capable of exploring the long-term impacts of alternative decisions, taking into account the laxity of the laws regulating the behavior of socio-ecological systems and the scarcity of data [69]. SD has been used to study the interaction between groundwater dynamics and the evolution of intensive irrigation agriculture in drylands [20,70–73]. SD is a tool specially designed for modeling complex systems that facilitates the recognition of multiple interactions among disparate but interconnected sub-systems [74]. In essence a SD model consists of a system of ordinary differential equations that makes a stock-and-flow representation of the system under study. The change in a variable stock is determined by the flows that affect it. The model's structure as a whole, which is made up of causal feedback loops including non-linear relationships and delays, constitutes a holistic and easily overlooked cause of its behavior [75].

The model we have used is adapted from Aquacoast [76]. It stems from a saga of dynamic models developed to study desertification (Martínez-Valderrama et al., 2016), based on a generic eight-equation dynamic model [77]. All of them come from realizing that the main relationships between human activities and natural resources resemble the predator-prey interactions [78,79] that have been modelled in ecology for near a century [80,81]. In this paper the main feedbacks between groundwater availability and irrigation agriculture are represented. The model equations can be found as Supplementary Material (SM). The following is a brief description around the sketch in Figure 2.

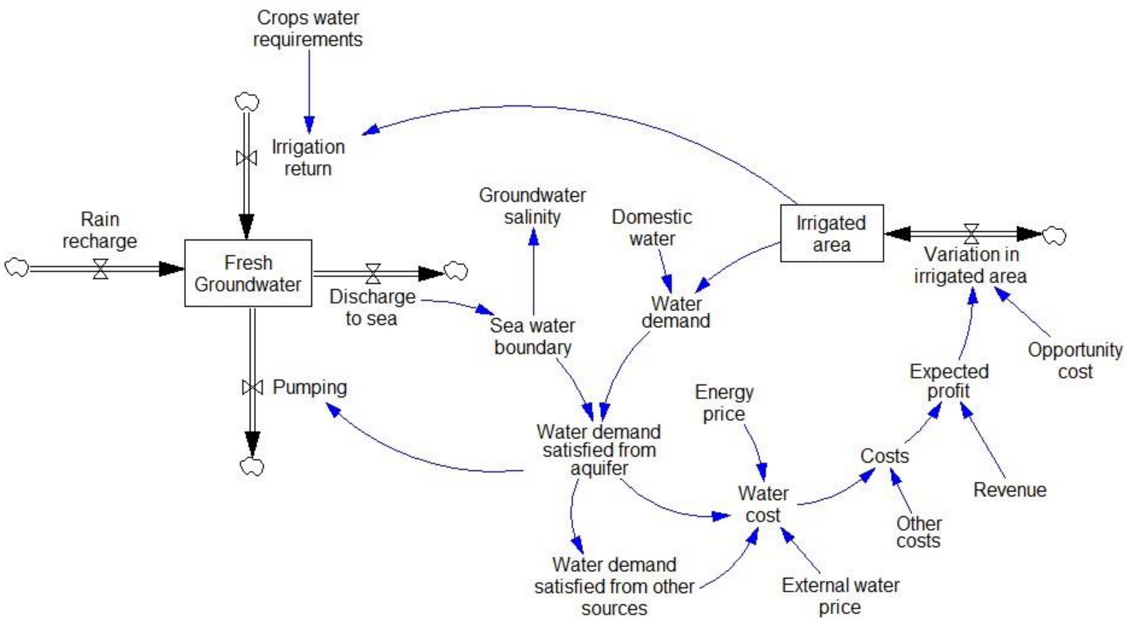

Figure 2. Stock and flow structure of the Aquacoast model adapted for the GSB case. Rectangles represent the stock variables of the model and double arrows and valves represent flow variables. Auxiliary variables and parameters can influence (blue solid arrows) any other component.

The model hinges on the interaction of two stock (*N*) variables, "Fresh groundwater" and "Irrigated area". The dynamics of the first of these is the result of the aquifer's water balance. In a natural regime, rainfall recharge contributes to the aquifer storage, which discharges to the sea after a period of residence below the ground. When the groundwater body is exploited, two flows are added: (i) groundwater pumping to supply domestic and irrigation uses; and (ii) the return to the aquifer of fraction of the water used for irrigation. Groundwater quality (i.e., water salinity in the model) of this coastal aquifer hydraulically connected to the sea depends on the saltwater–freshwater interface (SFI) dynamics. Under a disturbed regime induced by groundwater pumping, the freshwater component decreases and SFI displaces inland, This dynamic is reflected in the model by the variable "Sea water boundary" (see Eqs. 5–7 in SM). Salinity in this band ("Groundwater salinity", Eq. 21 in SM) is one of the indicators proposed to assess the risk of desertification in this case study. The model uses the electrical conductivity (dS m^{-1}) as a proxy of the content of salts, which is practical in high-salinity, groundwater-dependent coastal agriculture areas.

The irrigated area, on the other hand, follows the classical goal-seeking behavior scheme, that reaches an objective –"Desired irrigation surface"– in a given time, i.e., the time for farmers to carry out their plans (Eq. 9 in SM; Sterman, 2000). The current irrigation area converges towards desired irrigated area according to the short-term profitability of the agricultural activity and the opportunity cost, i.e., the expected returns from other alternative economic activities (Eq. 8 in SM). Thus, the idea implemented in the model is that the irrigated area increases when the activity is profitable (compared to the average wage in the region of Andalusia [82]), which represents the opportunity cost in our model) and decreases when it is not.

To complete this mechanism, we need to calculate the benefit of the agricultural activity, which depends on costs and revenues. Due to the nature of the problem, the cost of water is an important variable. It has two components, since the water used for irrigation comes from the aquifer and external sources. The model considers the possibility of transferring water from other basins at a given cost, as is the case in the area. This water dependency is the other indicator chosen to assess the risk of desertification ("Water Demand Other Sources", Eq. 13 in SM). To the extent that the cost of water increases (e.g., due to an energy price increase), the profit decreases and slows down the irrigated area growth or even shrinks it. However, if farmers' income increases (e.g., due to higher prices for their products) and balances off the previous loss, the greenhouse area will expand.

2.3. Risk of desertification

SD models yield time trends for all the included variables. However, SD is a flexible enough tool to support different data sources and to accommodate multiple analyses. Thus, it is possible to use statistical or stochastic models within its structure and program routines to implement advanced sensitivity analyses, optimizations and probability calculations (Martínez-Valderrama et al., 2020). Specifically, here the model serves to implement a risk analysis for exploring the sustainability of the system. For this analysis, the two indicators chosen to represent the condition of sustainability or desertification of the system are "Groundwater salinity" and "Water Demand from Other Sources".

The underlying idea is to calculate the long-term equilibrium of the system for the default scenario (Tables 1 and 2 in SM), i.e., its steady-state. The values of many model parameters widely vary over time (e.g. precipitation or prices), and if this were reflected in the model, the position of its steady state would also be constantly changing. Moreover, integrated models have many uncertainties due to measurement errors in parameters and data used for calibration, and assumptions about model structure, including assumed constants [84]. Hence, the steady-state associated to the baseline scenario is just one of the outcomes of the system. It is necessary to expand the number of scenarios and calculate their associated stationarities (steady states) to assess the likelihood of the system to desertify.

The options for generating different scenarios are manifold. In our case, we have generated random values for two parameters, one related to climate ("average rainfall recharge", Eq. 2 in SM) and the other to economy ("revenue per hectare", Eq. 16 in SM). With the 200 sets of values, we have obtained 200 steady states of the two indicators chosen to represent the sustainability, or desertification, of the system by running the model under such scenarios during a long enough simulation period. The result is represented in a scatter plot where the two proposed indicators' values are displayed. The equilibrium points can be obtained from the isoclines of the model, i.e., $dN/dt=0$ [78]. However, when the model acquires a certain complexity, it is impossible to solve the system of equations analytically. Alternatively, these equilibria can be obtained by numerical iteration by adopting long enough simulation periods [77]. The most likely system trajectory will be the one with the highest point density in the resulting cloud of points.

3. Results

3.1. Scenarios analysis

The evolution of the main variables of the modeled system accurately reflects what has happened in the GSB. The expansion of highly profitable greenhouse agriculture (Fig.

3a) led to the quantitative (Fig. 3b) and qualitative (Fig. 3c) deterioration of the CFA. The high technical efficiency of this irrigation (drip irrigation technology reaches 85–95% efficiency in the region, Alcalá et al., 2021; WWF, 2009) failed to halt this process. On the contrary, the rebound effect [87], driven by this high profitability, pushed farmers to demand more water resources to continue with their profitable economic activity. The start of the RBS operations (Fig. 1b) has solved these needs and those of neighboring basins. Exogenous water inflows have grown exponentially until fluctuating in a steady pattern (Fig. 3d) due to the stabilization of the greenhouse area (Fig. 3d). The practical inability, with current technology, to supply water for more greenhouses in a terrain with a complex orography together the low pumping yielding in the outcropping low-permeability metapelite bedrocks have limited the expansion of agriculture to high-slope areas.

The oscillations shown in the trajectories are associated with the seasonality of precipitation. This effect propagates throughout the system, affecting fresh groundwater (Fig. 3b), groundwater salinity (Fig. 3c) and imported water (Fig. 3d). When the aquifer is recharged (the model considers some months of rain and some months of drought; see Eq.2 in SM), the SFI recedes, improving groundwater quality and thus requiring less off-site water. In the dry season, the lack of water inputs displaces the SFI inland, groundwater quality deteriorates, and more water is transferred to the basin. The time-lag in farmers' decision-making, based on annual and non-seasonal variables, means that these pulses are not spread over the irrigated area.

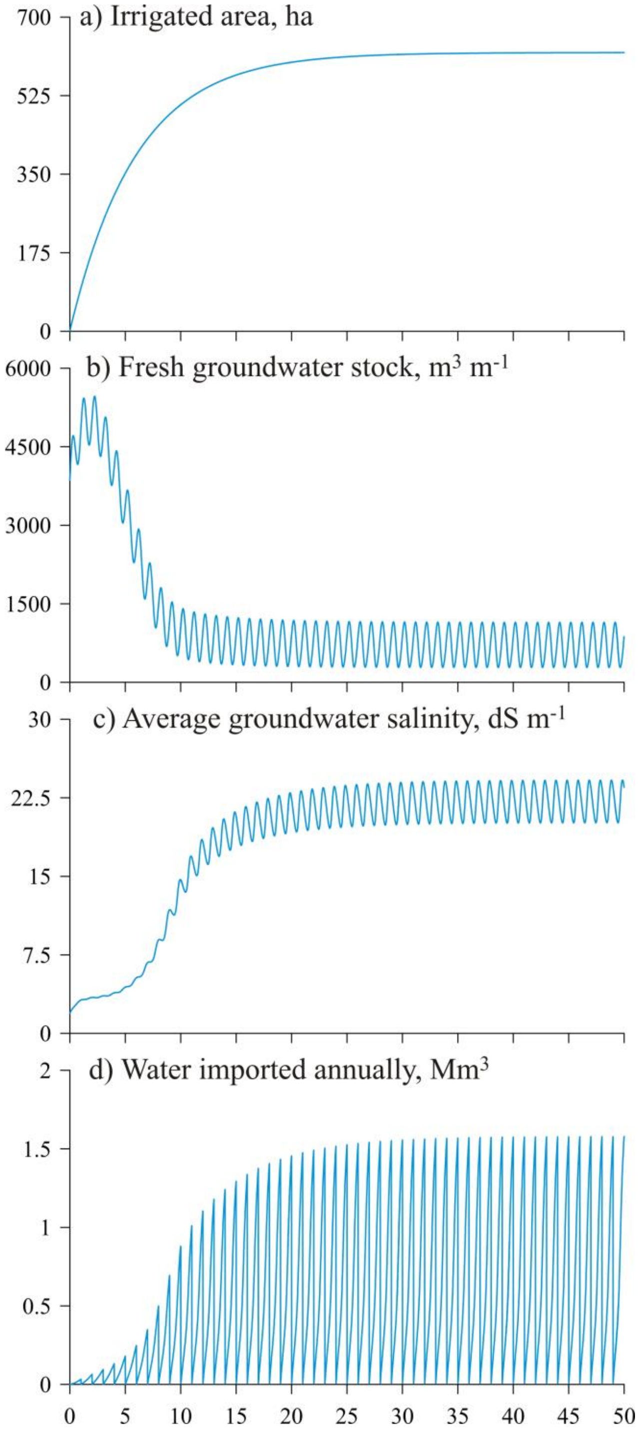

Figure 3. For the baseline scenario, temporal trends over 50 years of the (a) irrigated area, (b) fresh groundwater, (c) groundwater salinity, and (d) water imported in the GSB.

The model allows us to implement "what if" questions to explore the outcomes of various actions or policies and the associated effects and trade-offs. We have set four alternative scenarios (see ST 3) to examine the effects of climate change and the energy crisis. Figure 4 shows the evolution of the above variables for these scenarios compared to the baseline scenario (Fig. 3). Precipitation is expected to drop for the eastern Andalusia Region between 14.4% and 26.6% [88]. These changes are implemented in the model through the "average rainfall recharge" (scenarios I and II). The environmental variables "fresh groundwater" and "groundwater salinity" worsen, but the greenhouse area remains the same as water transfers increase to compensate additional degradation induced by the local water supply gaps. We have to consider that the decrease in precipitation, and the rise in temperatures, will also affect the water inflow magnitude of the reservoirs from where the water comes to the GSB (Fig. 1b), challenging one of the hypotheses of the model, that total water crop requirements are covered (Eq. 10 in SM).

Another current situation is the rise in energy costs (scenario III). According to data released by the Ministry of Agriculture, Fisheries and Food [89], the cost of energy for agriculture increased by 83.09% interannually, and the prices received by producers of agricultural products increased on average by 35.09% (fertilizers increased in price by 84.87%). This is reflected in the parameters "energy price", "price one cubic meter external water", "other costs". As can be seen in the middle column of charts in Figure 4, this scenario relieves the pressure on the resource, and the greenhouse area is far from what was achieved in the baseline scenario (27.8% less). The groundwater quantity and quality statuses of the aquifer notably improves, and it is not necessary to import as much water.

The simulation of the rise in energy costs can be more consistent if we simultaneously consider its impact on the selling price of products. Fruit and vegetable products have increased their selling price by 35% in the study area [90]. Applying this idea, we can formulate a more complex scenario (scenario IV) that includes the effect of energy price increment on production costs and "revenue per hectare" and climate change impact. In this case, we opted for an intermediate value between scenarios I and II, i.e., a drop of 20.5% in "average rainfall recharge". The right-hand column of charts in Figure 4 shows the increase in income offsets the losses of scenario III, and the greenhouse area equals that of the baseline scenario. However, this scenario results in a heavier impact on groundwater. The aquifer is drained earlier than in the baseline scenario, and its groundwater salinity is higher. A larger volume of water is imported from outside the basin to achieve the same irrigated area.

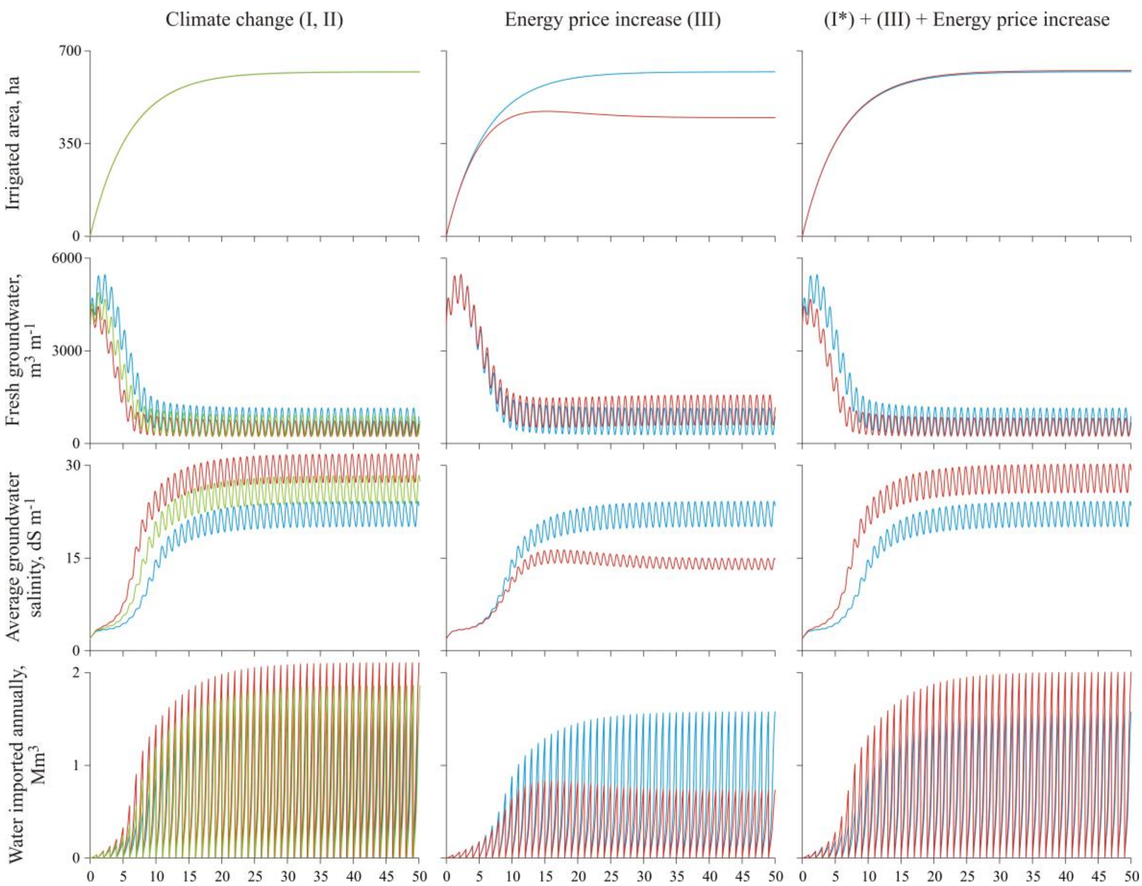

Figure 4. Comparison of the baseline scenario (blue line) with alternative scenarios (green and red lines). The scenarios can be found in ST 3. In scenario I, a 14.4% drop in

"average rainfall recharge" is simulated (green line in the left column charts). In scenario II, this drop is 26.6% (red line in the left column charts). Scenario III considers an 85% increase in "energy price", "price of one cubic meter of external water" and a 35% increase in "other cost per hectare" (red line in the middle charts). Finally, scenario IV implements scenario III with an intermediate value of "average rainfall recharge" in scenarios I and II (red line in the right column charts).

3.2. The risk of desertification in the GSB

The regular use of a SD model is aimed to find out how the system behaves under different scenarios. This methodology enables also to implement other types of analysis (Martínez-Valderrama, Ibáñez, Gartzia, & Alcalá, 2021). The calculation of the cloud of equilibrium points serves as an Early Warning System (EWS). The aim is to visualize the most likely end state of the studied system, given the conditions set by the baseline scenario. We start from a pristine aquifer, in which fresh groundwater storage is maximum and groundwater salinity is 1.366 dS m^{-1} (see ST 2), there is no groundwater pumping, and the area covered by greenhouses is zero. We have chosen two variables to illustrate the degradation of the system. Groundwater salinity is one of them since beyond certain threshold of the irrigation water salinity, crop yields decline to zero. The other variable is the volume of imported water, which reflects the dependence of the system on external factors.

As can be seen in Figure 5, the generated endpoint cloud adopts a curious shape. The boundary responds to the “water demand from other sources” equation (Eq. 13 in SM). As groundwater salinity increases, water is brought in from outside the basin to meet all the water needs of the crops (this is one of the assumptions of the model). In this way the imported water is directly proportional to the local groundwater salinity.

To calculate the probability of desertification risk it is necessary to establish degradation thresholds. Strict criteria would be to define it as (i) the maximum water salinity tolerated by tomatoes, the main crop in the area, which is 3.5 dS m^{-1} according to Magán Cañadas et al. (2009); and (ii) an almost no dependency, i.e., water imported <0.1 $Mm^3\ yr^{-1}$) on external inputs to the basin. The risk of aquifer degradation (i.e., the risk of desertification) would be 97.5%, which is the maximum of (i) 97.5% of the endpoints above 3.5 dS m^{-1} and (ii) 97% of the endpoints below 0.1 $Mm^3\ yr^{-1}$. The red area in Figure 5 indicates the most likely end states of the system with the baseline scenario. There is a 74.5% probability that (i) groundwater will have an average salinity above 40 dS m^{-1}, and (ii) the basin needs to import more than 2.4 $Mm^3\ yr^{-1}$ to support the current 631 ha of greenhouses. It is concluded that, with the current land use, the area is doomed to desertification.

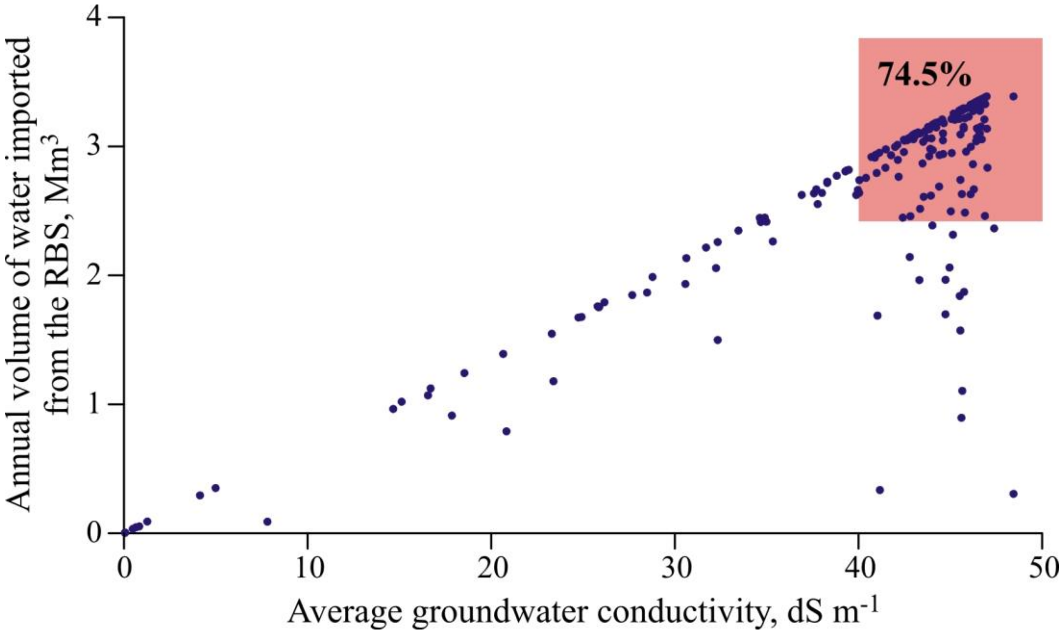

Figure 5. Cloud of endpoints after the simulation of 200 stochastic scenarios for water inflow parameters. The red area, defined by groundwater salinity >40 dS m^{-1} and water imported > 2.4 $Mm^3\ yr^{-1}$, denotes the most likely area towards which the system is heading, as it accounts for 74.5% of the endpoints.

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4. Discussion

4.1. Water resources degradation and desertification

Desertification is an ambiguous and complex concept surrounded by many paradoxes (Reynolds, 2021). As a result, there is still no consensus on the extent and importance of the problem, and solutions have not been as effective as intended. One of the most remarkable gaps is inclusion of water resources degradation linked to irrigation pressure. Historical desertification maps have captured desertification through soil and vegetation cover trends (e.g. GLASOD Project [26]) and remote sensing techniques have gradually become the foremost means in monitoring and assessment of desertification [93]. Among these methods prevails the use of the Normalised Difference Vegetation Index (NDVI) [94], a proxy for Net Primary Productivity (NPP) (Prince 1991). Following NDVI, irrigated areas can be interpreted as re-greening and thus represent an improvement in land condition. Unless irrigated areas are explicitly identified as anomalies (e.g. del Barrio et al. 2021), there may be confused with solution that tackle desertification, despite its enormous impact on groundwater. It is necessary to complement this productivity indicators with the impact on water resources. Satellite observations, such as the GRACE mission (Tapley et al. 2004), provide new space-based insights into the global nature of groundwater depletion [7] and benefit water management [97]. The use of integrated simulation models, which can be coupled with risk or sensitivity analysis, is another approach to help implement EWS (Martínez-Valderrama et al., 2020) and give a more complete picture of desertification problems.

The model presented here links economics with groundwater dynamics, providing a collection of indicators that portray the stability of the socio-ecological system analyzed. Water productivity can be very high in climatically favorable areas for

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horticultural crops. However, the toll on water resources is so high as to put this source of wealth at risk of collapse. More extreme cases of land-uses changes induced by groundwater usage are found in hyper-arid areas. The greening of the Arabian desert, due to the massive exploitation of its fossil groundwater, led to the appearance of green crop circles that stood out on the desert sands (Fig. 6) (Martínez-Valderrama, Guirado, & Maestre, 2020a). These green spots undoubtedly represent a rebound in NPP, but the consequence has been the quick groundwater storage depletion at an annual rate of 15.5 $km^3\ yr^{-1}$ [99], and a ban on further use [9]. Integrating modeling coupled with risk analysis can support the assessment of land-use changes and their impact on water resources by exploring the sustainability of policies.

Figure 6. Crop circles in the Al Jawf Region in Saudi Arabia. Groundwater irrigation results in a gain in NPP. Monitoring systems that ignore other indicators (e.g. aquifer piezometric level) may lead to misinterpretations of desertification. Source: Axelspace Corporation Wikicommons:

https://commons.wikimedia.org/wiki/File:Circular_irrigations_in_Al_Jawf_Region,_Saudi_Arabia.jpg (accessed on 28 June 2022).

4.2. SD as a suitable tool for Land Degradation Neutrality implementation

Monitoring and surveillance are priority strategies in the framework of Land Degradation Neutrality (LDN), which places prevention above reducing or reversing land degradation (Orr et al., 2017). LDN is the UNCCD's response to the lack of progress in tackling desertification [101] and seeks “to maintain or enhance the land-based natural capital, which comprises the edaphic, geomorphological, hydrological and biotic features of a site” [102]. LDN was adopted as target for Sustainable Development Goal 15 [103], and to be operational it must be managed at the landscape level through integrated land use planning (e.g. irrigation and watershed plans). It is here that the holistic view of a model such as the one presented can be extremely useful for processing the interaction of socio-economic and environmental features. As we have seen, the analysis at the basin level offers us an assessment of the impact of economic activities in the area but also of the off-site effects. A DS model serves to contrast, within the same framework, the economic aspirations of a place and its resource constraints.

The environmentalist and developmental visions of the convention have always coexisted within the UNCDD [104]. Combating desertification means making efforts to preserve the environment and prevent its degradation. But it also means offering economic opportunities to the population, so that they can live in these territories. The characteristics of drylands (low and highly variable rainfall) mean that permanent intensive land uses pose a serious threat to their sustainability –some authors claim that with less than 250–300 mm yr^{-1} rainfall, drylands should be kept under pastoralism

[105]– making difficult to match their economic growth to that of richer countries. About 72% of drylands occur in developing countries [106], accounting for 90% of the two billion people living in drylands [107]. It is expected that there will be some support or transfer of resources from wetter areas to drylands, especially if they are within the same country. Once again, this requires wise land use planning, where this transfer is compensated for and does not result in a drain on water and financial resources [105] that spreads unsustainability to other territories. Fueling a water supply model has widened water gaps [108]. It is necessary to include other types of solutions, such as water pricing [109], crop changes (more adapted to aridity) [110], or economy diversification [111], e.g., by promoting sustainable tourism since massive tourism can contribute to degrade the system more [73]. Further developments of the Aquacoast tool (model + EWS) can provide at least three contributions (i) simulate the effect of complementary solutions to water transfers such as the reuse of reclaimed water, diversification of crop and land uses, or the use of economic tools such as water banks [112]; (ii) include other desertification indicators by selecting economic variables such as "Profit per hectare" or "Irrigated area"; and (iii) expanding the study area to assess land degradation from a broader scope where intra-territorial trade-offs are allowed. This perspective fits with LDN framework, which allow compensation within the same land type [102]. If neutrality is required at the level of a catchment as small as the one studied, the sustainable options for economic development are short. When the trade-off is at a larger scale (e.g., the Rules–Béznar System and the coastal aquifers of Granada province; Fig. 1b.), then it is possible that some areas take advantage of a favourable situation (e.g., market positioning, long growing season due to mild winters, etc.). In this case, it must be considered that the "Source" zones must be compensated by the "Sink" zones, where wealth is created, i.e., there must be a territorial distribution of

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wealth, for which there are various mechanisms such as those mentioned above water banks.

4.3. Water-food-energy nexus under the holistic vision of SD

The simulations carried out highlight the feasibility of SD models to address the water-food-energy (WEF) nexus. Integrated approaches are needed to identify the synergies and trade-offs between the various components and manage the nexus [113,114], as WEF interdependencies are central to the global sustainability question [115]. SD, by its holistic nature, is well suited to this multifaceted paradigm. There are several examples of its application around the world, such as in Iran [116], China [117], or Spain [114]. The question is not whether or not to consider the nexus, but how to address it. In the small basin analyzed, we have seen how water use depends on energy, the price of which is affected by an energy crisis fostered by reaching peak fossil fuel production [118] and exacerbated by the war in Ukraine. In addition to the cost of caughting, elevation and transferring water, or shipping goods, chemical fertilizers have skyrocketed in price. The country's leading fertilizer manufacturer has stopped production on several occasions because it is not profitable [119]. Suddenly, one of the premises on which the economic model in the area rests, cheap energy, is falling apart. At the same time, international competition could drive down product prices (Martínez-Valderrama, Guirado, & Maestre, 2020a), further deepening the crisis in the sector, which thought that climate change would be its main concern. Indeed, recent findings indicate that extent of dryland in the Mediterranean region has been expanding in the past decades and will continue to expand in the coming decades due to the stronger warming effect than other regions [120].

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Dynamic simulation tools offer us the possibility of analyzing how the system behaves in the face of apparently improbable scenarios, but which, when they happen, can completely change the expectations we had about the system's evolution. The model presented here, by offering a handle on the food, water, and energy aspects, enables us to explore the future of the system under the WEF paradigm. This type of research is critical if we want to provide affordable and reliable resources in an environmentally sustainable way [9].

5. Conclusions

Groundwater is freely accessible to numerous people scattered over an area, who may exploit it and may influence its quantity and quality. These individuals have their own objectives on a different scale from the scale of an aquifer and in general they have no knowledge of the aquifer’s properties and potential [1]. This “silent revolution” is largely a market-driven phenomenon (Llamas & Martínez-Santos, 2005), as cost of groundwater, excluding externalities, is in most cases only a small fraction of the economic value of the guaranteed crop (in the profit and loss account of greenhouses in the southeastern continental Spain account for 2.8% of the costs [122]). Added to this is the difficulty in measuring the groundwater resource (global groundwater storage historical estimates vary from 7 to 23 Mkm^3 [99]), which generates great uncertainty in its governance. In this complex context, groundwater monitoring is necessary. The development of EWSs, which warns whether their use allows for enduring economic development, can support policies that reconcile ecology and economy.

In the drylands, the spatial and temporal variability of natural resources has led to opportunistic behavior of their inhabitants. The aim is to take advantage of peak productivity periods triggered by rains and create reserves to survive the uncertain dry periods, which are generally much longer (i.e., the pulse-reserve paradigm [123]. Climate change has fueled this strategy, and irrigated agriculture is considered a refuge from the predicted increase in aridity. The transformative capacity of the environment in the service of this opportunistic instinct creates ephemeral economies that quickly devastate resources that took decades or centuries to accumulate. This situation has already been experienced in the southeast of the Iberian Peninsula. Lead mining (e.g., Sierra de Lújar Mountain in the GSB western summit, Sierra de Gádor Mountain in the nearest Almería province) resulted in the rapid enrichment of the area but also in the deforestation of its coastal mountain ranges, whose orography, combined with torrential rains, led to the loss of fertile land [124] and the generation of coastal plains at the river outlets [125]. More than 150 years later, these places are still depopulated, with little chance of development. The groundwater anthropization in the Mediterranean region is leading to the widespread collapse of these crucial reserves [126]. The desired enrichment coincides with signs of desertification (i.e. groundwater degradation), but the strong social support for irrigation [127] discourages the implementation of more moderate water use policies.

In the light of the results obtained, it seems necessary to reconsider water use in the studied area, which is an example of the shared problematic with other neighboring areas. The enormous dependence on external resources, coupled with increasing energy costs, dangerously increases the vulnerability of the area. The use of alternative water resources (reclaimed water and stormwater harvesting), the diversification of the economy (sustainable tourism), the use of crops more adapted to aridity, or the application of economic tools to redistribute the wealth generated with the water that arrives from outside, can help to preserve this strategic and vital resource.

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