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A Preliminary Lumped Assessment of Pollution Risk at Aquifer Scale by Using the Mean Residence Time. Analyses of Potential Climate Change Impacts

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Abstract: An assessment of the risk for groundwater pollution and vulnerability to pumping can help identify strategic groundwater bodies to define sustainable management measures of groundwater resources. In this paper, we propose a new method to make a preliminary estimation of the risk for groundwater pollution at the aquifer scale through the lumped turnover time index (T index). A new lumped index (L-RISK index) was defined to assess the significance of the risk for pollution at the aquifer scale. Both L-RISK and T indices were employed to calibrate a linear regression model that showed a good inverse correlation in the eight aquifers of the Upper Guadiana Basin (Spain). This novel method can be applied to analyze a wide range of aquifers with limited information in order to identify potential strategic aquifers. It also allows one to make a preliminary assessment of the impacts of climate change on L-RISK. The results showed a high variability of the T index in the eight aquifers (8–76 years). Three of them had significant greater mean T values, which could be considered to be the main strategic groundwater resources. In the future, the T index will increase between 8 and 44%, and the L-RISK will decrease in all aquifers (1–18%).

Keywords: groundwater pollution risk; residence time; vulnerability; droughts management; climate change impacts; adaptation strategies

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1. Introduction

Groundwater is a valuable renewable resource that plays an important role in providing water supplies across the world, especially in semiarid areas where rivers are usually ephemeral. The interactions between anthropogenic activity, the hydrosphere, and climate affect groundwater resources and produce environmental impacts on human health, groundwater quality, and groundwater quantity depletion. This degradation produces negative effects—not only in exploitation for different uses (human consumption, irrigation, etc.) but also in groundwater-dependent ecosystems [1–3]. Therefore, the protection of groundwater resources is a priority issue to be considered to achieve the sustainable management and maintenance of the good status of water bodies according to the Water Framework Directive (WFD) [4].

The degree of protection of groundwater from contaminants mainly depends on intrinsic vulnerability and pollution risk due to anthropogenic activity. For many years, several approaches have been developed to assess intrinsic groundwater vulnerability [5–8]. Process-based methods [9] analyze pollutant movement in groundwater by using transport modelling. Statistical methods try to find a correlation between the influencing factors and contaminants [10]. Overlay and index methods [6,7] are based on the rating of hydrogeological factors that influence groundwater vulnerability. Index approaches are the most widely used through methods such as DRASTIC (D: Depth to water; R: Net recharge; A:

Aquifer media; S: Soil media; T: Topography; I: Impact of vadose zone; C: Hydraulic conductivity) [6] due to its simple implementation [8]. All these methods provide distributed results that can be displayed on maps where different vulnerability levels are represented. Groundwater vulnerability can also be summarized at the aquifer scale through global indices that allow for lumped results between different aquifers to be compared [11–13]. In order to perform a risk assessment, we need to combine hazard and pollution vulnerability maps, with the hazard being defined as the potential source of contamination derived from the human activity at the land surface [14]. Pollution risk is usually assessed by considering different land uses as potential sources of contamination, as proposed in COST (European Cooperation in Science and Technology) action 620 [14], although pollution risk assessment can also be focused on the risk from only one source of contamination, e.g., landfills [15,16]. For that purpose, some contamination indices have been developed and widely applied [15,17,18] to assess the impact of a pollution source in groundwater, but they are usually applied at specific sites because they require a detailed hydrochemical analysis.

In addition to the groundwater quality issues, the protection of groundwater resources should also take the quantitative status of the aquifer into account. Therefore, in general, the term groundwater vulnerability can be applied to both intrinsic vulnerability to pollution and renewable resources (or vulnerability to pumping) [19,20]. An estimation of renewability is essential for the sustainable exploitation of groundwater resources and it can be approached through mean residence time or the mean age of the water leaving a system [2,20]. The understanding of the timescales of groundwater resources helps to identify problems related to the qualitative and quantitative status, as well as to groundwater-dependent ecosystems [21].

Some authors have linked the concept of vulnerability to pollution and/or to pumping with the mean residence time [22–24] for different purposes. It has been employed to define groundwater source vulnerability and delineate protection zones in karstic aquifers [25,26]. Residence time has also been employed to validate distributed groundwater vulnerability maps in any type of aquifer by using different approaches, concluding that high vulnerability values are related to short residence times [27–29].

Different approaches have been used to estimate residence time (also called renewal time, groundwater age, or travel time through the saturated zone). Some authors have used contaminant transport models [30,31] or environmental and artificial tracers [32,33]. Age dating is a valuable way to analyze changes in groundwater systems [34] and a quantitative measure of the renewable capacity of groundwater [35]. It is also an important indicator of groundwater susceptibility to anthropogenic contamination [36]. Residence time is usually assessed in a distributed way [37,38], but it can also be estimated in a lumped way by applying simple index approximations, such as the turnover time index (T index), which expresses the ratio between storage (S), and recharge (R) [20,39]. It is related to the resilience of groundwater bodies to temporal pumping increases.

Aquifers with a short residence time are more vulnerable to exploitation [20,40], which will be exacerbated in the future due to climate change. Climate change will not only affect the quantity of water resources but also their quality, especially in agricultural areas. Impacts of climate change on groundwater resources has been addressed in several studies, especially in semi-arid areas [34,41]. They conclude that climate change will involve a decline of groundwater levels due to a decrease of the recharge and an increase in water requirements to supply agricultural and urban demand. Groundwater vulnerability has also been analyzed under climate change scenarios. Some studies revealed that the vulnerability is not very sensitive to climate change [12,42] because many factors considered in the index-based assessment are static. However, other authors [43] found different patterns of groundwater vulnerability to contamination between drought, average, and wet periods. Groundwater vulnerability has also been proven to significantly shift under future land use change scenarios [44,45], although these changes vary from site to site [46].

Distributed approximations of residence time have been employed to assess and validate intrinsic groundwater vulnerability in several studies [22,23,37]. These distributed

approaches, more precise and detailed than lumped approaches, demand higher computational efforts and need data that are not always available, which makes it impossible to use a single method to validate the vulnerability maps from different case studies [13,14]. This drawback usually makes the results of different sites scarcely comparable. However, lumped approaches provide preliminary evaluations that can be usually applied to large areas with limited information [47], thus allowing one to compare the status of many different groundwater bodies.

In this paper, we propose a new method to make a preliminary estimation and/or validation of the risk for groundwater pollution at the aquifer scale through a lumped index based on the mean residence time concept. A regression model was defined to assess the risk for pollution from the lumped mean residence time and to estimate potential impacts of climate change scenarios on pollution risk. The proposed methodology was applied to the Upper Guadiana Basin (Spain), which is composed of different aquifer typologies. The obtained results will be useful to identify potential strategic aquifers, where more exhaustive analyses should be performed to define conjunctive use measures for the sustainable management of water resources systems, especially during critical droughts.

The spatial distribution of groundwater residence time and groundwater vulnerability to contamination has been related in previous works [22,23]. Nevertheless, as far as we know, none of them assessed groundwater pollution risk from the mean residence time approximation, and none of them studied the potential impacts of climate change. The novel lumped approach proposed in this paper can be applied to extensive areas with limited information. In the literature, we found examples of lumped vulnerability indices (e.g., [11,13]), but none of these studies were about groundwater pollution risk. The use of this lumped approach, based on mean residence time and considering not only the resilience to periods of intensive pumping [20] but also the pollution risk, is also novel.

2. Materials and Methods

2.1. Methodology

The proposed methodology intended to demonstrate the utility of the lumped mean residence time as a preliminary approach to validate groundwater pollution risk at the aquifer scale.

The inputs to obtain the lumped indices included variables (physical and hydraulic properties) and parameters (overlying layers and hydrogeological characteristics) of the aquifer media that could come from direct observation (field measurements) or other techniques (geological and flow models). Climatic and socio-economic variables also impact (directly or indirectly) risk of groundwater pollution and mean residence time, and they should be considered in a risk of pollution assessment.

Figure 1 depicts the flowchart of the methodology including the modelling framework to assess the future impacts of climate change on risk assessment.

The analysis of future impacts required the generation of local climate scenarios and their propagation through previously calibrated recharge and groundwater flow models.

2.1.1. Pollution Risk Assessment

Risk of groundwater contamination is defined as the probability that groundwater will become contaminated due to the anthropogenic activity in the overlying land surface [14,48]. A risk assessment considers the interaction between the contaminant load coming from the land use (hazard) and the groundwater vulnerability to pollution.

A hazard map can be obtained from a land use map at adequate detail in order to determine the contaminant load from different activities. Land uses change over time, especially in agricultural areas through crop rotation, as well as pollutants leaching from each type of crop. Pollutant loading from crops is highly variable, depending on irrigation system and efficiency, soil characteristics (which also vary regarding the type of crop), crop management system, fertilizer management, climatic conditions, etc. [49,50]. The complex relationship between these variables requires a multi-disciplinary approach [51], and a real

detailed approximation by crop patterns is not always feasible. Instead, a more general land use classification can be used to define different rates that approach the probability of contamination that reaches groundwater.

Figure 1. Flowchart of the methodology. L-RISK: lumped risk index; O-DRASTIC: Groundwater vulnerability assessment method based on DRASTIC method; CROPWAT: Computer program for irrigation planning and management.

Groundwater vulnerability depends on the intrinsic hydrogeological characteristics that make an aquifer susceptible to contamination, such as aquifer media, hydraulic conductivity, and the type of soil above the aquifer. Groundwater vulnerability can be assessed by using index-based methods such as DRASTIC or other approaches that allow one to display different vulnerability classes in a map. A vulnerability map should be validated by expert criteria or by contrasting it with contaminant concentrations or other techniques.

Hazard and vulnerability maps are combined to define a risk map, considering that the highest risk will exist when the most dangerous hazard is located over the highest vulnerability zones [52,53].

The combination of hazard and vulnerability provides a risk class that must be previously defined, e.g., following a matrix concept [14].

The lumped risk index (L-RISK index) assesses the significance of each risk class at the aquifer scale by a weighted average. The L-RISK index is defined as:

$$L - RISKINDEX = \frac{\sum_{i=1}^n RISK\ INDEX\ class_i \times S_i}{S}, \quad (1)$$

where n is the number of pixels (or zones with a unique value of risk) in the map, $RISK\ INDEX\ class_i$ is the value of risk class in the pixel or zone i , S_i is the surface area in the pixel or zone i (m^2), and S is the total surface area of the aquifer (m^2).

The lumped index evaluates the overall pollution risk of an aquifer. It allows for the results of different aquifers to be compared in order to establish appropriate management measures regarding land use, type of crops, fertilization, and irrigation techniques.

2.1.2. Mean Residence Time

The mean residence time is an indicator of the susceptibility of the groundwater resource to pollution. It provides an approximation of the time that a contaminant event needs to reach a zone within the aquifer. The mean residence time can be approached by using tracers or particle tracking in flow models in order to obtain distributed results, but it can also be estimated as a T index [20] for a whole aquifer.

In this paper, we used the T index, defined as the ratio between the mean volume of a resource stored in a aquifer and the mean aquifer recharge in the studied period:

$$T = \frac{V}{R} \quad (2)$$

where T is the turnover time index (years), V is the volume of the groundwater resource (Mm^3), and R is the groundwater recharge (Mm^3/year).

The volume of a groundwater resource can be calculated from geological or flow models that provide accurate results, but it can also be obtained from existing data from river basin authorities. The aquifer recharge can be calculated by applying different approaches, from simple balance calculations to complex deterministic models previously calibrated in the studied area. Both variables (groundwater resource volume and recharge) can be considered as the mean values in a large historical period or the mean values in a more recent period (e.g., a river basin management plan).

2.1.3. Lumped Validation of Pollution Risk Maps

Based on the assumption that a higher groundwater vulnerability is related to a shorter residence time, we attempted to establish a correlation between the T index and the L-RISK index at the aquifer scale.

A simple linear regression model was defined in order to approximate the L-RISK as a function of the mean residence time, calculated as the T index, in the studied aquifers. Different transformations of the analyzed variables ($y = \text{L-RISK}$ and $x = T$) were performed (Table 1) in order to determine the model that provided the better correlation.

Table 1. Tested regression models and transformation of variables (* means the combinations of the target and explanatory variables).

TR(y) = a × TR (x) + b		TR (x)				
		-	x^2	$\sqrt{x}$	Log (x)	1/x
TR(y)	-	*	*	*	*	*
	y^2	-	*	-	-	-
	$\sqrt{y}$	-	-	*	-	-
	Log(y)	-	-	-	*	-
	1/y	-	-	-	-	*

2.1.4. Propagation of Hydrological Impacts to L-RISK

A future assessment allows one to analyze the mean change of a groundwater system due to exploitation and climate change. Climate change will affect groundwater recharge, stored resource volume, and, therefore, the T index. Groundwater vulnerability and, therefore, risk of pollution will also experience changes due to variations of mean groundwater levels and recharge.

The climatic series of the future potential scenarios must be downscaled and propagated by using previously calibrated recharge and flow models. Those models are used to calculate the future T index, and the calibrated regression model is used to estimate the future L-RISK index.

2.2. Study Area, Data and Modelling Framework

2.2.1. Location, Hydrogeology and Historical Climatic Data

The case study was the Upper Guadiana Basin, located in the center of the Iberian Peninsula. It covers an extension of approximately 14,000 km². The topography was found to be predominantly flat in the central area and undulating in the north (Sierra de Altomira) and south (Campo de Montiel) boundaries.

In general, the drainage network was found to be poorly defined, although it showed strong natural interaction between groundwater and surface water [54]. The Guadiana River and several tributaries have risen to over one hundred wetlands that make up UNESCO's "Mancha Húmeda Biosphere Reserve." Most of them were found to be groundwater-dependent wetlands and are also in the Ramsar Convention list [54]. The Upper Guadiana Basin is composed of eight groundwater bodies, including detrital and carbonated aquifers. Figure 2 shows the main geological formations (a) and hydrogeological characteristics (b). The northern aquifers are heterogeneous, mainly composed of Cretaceous and Neogene formations, and they consist of clays, sand, marls, calco-dolomitic, and Paramo limestone materials [55] from medium to high permeability. The central zone is characterized by the presence of Neogene–Quaternary materials, alternating detrital and carbonated formations with medium, high and very high permeability. The southern area is predominantly composed of fractured and highly permeable Jurassic calco-dolomitic materials that mainly consist of dolomite and oolitic limestones, as well as thick levels of marls [55,56]. The hydraulic conductivity in the Upper Guadiana Basin is highly variable. In the northern area, the conductivity is low (below 1.5×10^{-4} m/s although there is a small area with a conductivity higher than 10^{-3}). The central area has values between 1.5×10^{-4} and 10^{-3} m/s, and the conductivity in the southern area is mainly in the range from 3.5×10^{-4} to 5×10^{-4} m/s [19].

Figure 2. Case study: (a) geological map; (b) hydrogeological characteristics and permeability; (c) land use map.

The climate of the study area can be considered continental and semiarid. Historical climate data from 1974–2015 were collected from Spain02 project [57]. The mean annual precipitation from this dataset was found to be 433 mm/year, and the mean temperature was 14.6 °C. Summer months are characterized by high temperature and scarce rainfall, causing the mean potential evapotranspiration to be notably high; therefore, the recharge and runoff are low in most cases [54,55]. The mean annual recharge obtained from the calibration of Sacramento model in the period of 1974–2015 was 46.8 mm/year.

The main land use in the Upper Guadiana Basin is agriculture, which covers approximately the 80% of the total area (Figure 2c). The use of nitrogen fertilizers has led to high levels of nitrate in many groundwater areas. Moreover, the expansion of irrigated agriculture resulted in an intensive groundwater withdrawal (mainly in the central aquifers), that caused the depletion of the water table by more than 20 m in some areas between the

mid-1970s and the first decade of the new century, which led to a degradation of important wetlands such as Las Tablas de Daimiel Natural Park. Since the WFD came into effect, there has been a growing awareness regarding the consequences of overexploitation, and more sustainable policies have been applied in the last few decades.

2.2.2. Groundwater Flow (Modflow) Model

A numerical groundwater flow (Modflow) model from the river basin authority was used to obtain some necessary data to apply the proposed methodology. The model simulated the groundwater flow and river–aquifer relationship in the eight groundwater bodies in the Upper Guadiana Basin. It provided hydraulic head maps and the flow budget in the modelled groundwater bodies.

For input, the flow model required recharge series, which were obtained from a calibrated Sacramento Soil Moisture Accounting (SAC-SMA) model of the US National Weather Service River Forecast System in the study area.

The pumping schedule and distribution were defined from historical data in the period of 1974–2015.

2.2.3. Vulnerability Map

For the purpose of this paper, we employed a vulnerability map generated in a previous work [19]. That map was obtained from an optimization of the DRASTIC method in an attempt to harmonize the vulnerability assessments of different aquifer typologies. The new method, called O-DRASTIC (Optimum DRASTIC) (see Appendix A), maintains all the parameters from the original DRASTIC method while varying their weights and classification. Data to compute depth to water (D) and conductivity (C) were obtained from the Modflow model, and net recharge (R) data came from SAC-SMA model. Mean values of D and R in the historical period were considered to calculate O-DRASTIC. The other parameters came from official databases (A: hydrogeological map of Spain 1:200,000; S: soil map of Spain 1:1,000,000; T: digital terrain model 100 × 100 m cell size; and I: lithostratigraphic map of Spain 1:200,000). The O-DRASTIC vulnerability map (Figure 3a) was validated by using nitrate concentrations [19], and it provided better results than the original DRASTIC model in the study area.

Figure 3. (a) Groundwater vulnerability map (O-DRASTIC) and (b) mean historical groundwater resource (m).

2.2.4. Groundwater Resource Map

From the flow model simulation, we calculated the mean resource map of the upper aquifer for each groundwater body in the historical period (1974–2015). The combination of the geometry, hydraulic head, and storage coefficient provided a mean volume of resource expressed as water column (m) for each cell within the aquifers (Figure 3b).

2.2.5. Generation of Local Climate Scenarios

A statistical method was used to define local future climate change scenarios for the case study. It was based on the historical information (Spain02 dataset [57]; see Figure 4) for the adopted reference period (1976–2015) and nine regional climate model (RCM) simulations (CCLM4-8-17 and RCA4 nested to CNRM-CM5, EC-EARTH, and MPI-ESM-LR; HIRHAM5 and RACMO22E nested to EC-EARTH; and WRF331F nested to IPSL-CM5A-MR) available in the CORDEX (Coordinated Regional Downscaling Experiment) project [58]. We used the most pessimistic emission scenario, RCP (Representative Concentration Pathway) 8.5, and the future temporal horizon of 2016–2045. The future local scenarios were generated by applying the first and second moment correction techniques under the bias-correction approach. In this approach, a perturbation (transformation function) was applied to the control series of the RCM simulations to obtain another series with statistics (mean and standard deviation) more similar to the historical series. The same transformation function was applied to the future simulations of the RCM to obtain the local climate change projections. Finally, an equally feasible ensemble of individual climate change projections were proposed in order to define more robust climate scenarios that were more representative than those based on a single model.

Figure 4. Monthly mean historical and future estimated climatic variables (precipitation (P; mm) and temperature (Ta; °C)). The historical (1974–2015) data were obtained from the Spain02 dataset, and the future (2016–2045) series was generated by applying the proposed methodology.

3. Results and Discussion

3.1. Pollution Risk Assessment

The risk of pollution map was computed by combining the groundwater vulnerability map (O-DRASTIC) and the hazard map.

The hazards were defined by assigning a rate to the different categories of the land use and land cover (LULC) map. Though it did not provide detailed information about the crop typologies, the LULC map displayed the main agricultural uses in this area. Based on literature [59,60], rates were assigned to the different land use categories regarding irrigation, since irrigated agriculture is considered the major factor that contributes to the diffuse pollution of surface and groundwater bodies [49]. The leaching of nitrogen was not estimated due to the uncertainties in some variables that influence on it [49], such as the crop rotation and climate variability in the historical period. We think that it would encumber the proposed assessment and would be out of the scope of this paper. Therefore, for a preliminary lumped assessment, we decided that a rough classification could afford an approximation of the potential hazard. Previous studies [59] also considered land use

as a potential variable to estimate the contaminant loading parameter. Table 2 shows the rates assigned to the land uses.

Table 2. Rates of land use (LU).

LU	Rate
Agroforestry areas	7
Airports	2
Annual crops associated with permanent crops	8
Broad-leaved forest	2
Burnt areas	2
Complex cultivation patterns	8
Coniferous forest	2
Construction sites	2
Continuous urban fabric	10
Discontinuous urban fabric	10
Dump sites	9
Fruit trees and berry plantations	7
Industrial or commercial units	8
Inland marshes	1
Land principally occupied by agriculture, with significant areas of natural vegetation	5
Mineral extraction sites	3
Mixed forest	3
Natural grasslands	3
Non-irrigated arable land	5
Olive groves	6
Pastures	5
Permanently irrigated land	8
Road and rail networks and associated land	2
Sclerophyllous vegetation	3
Sparsely vegetated areas	3
Sport and leisure facilities	2
Transitional woodland-shrub	2
Vineyards	5
Water bodies	1

The combination of groundwater vulnerability and hazard maps provided the risk of pollution map. The risk map was classified following the matrix technique displayed in Figure 5a, and the lumped risk index (L-RISK index) is summarized at the aquifer scale following Equation (1).

Figure 5. (a) Pollution risk map (numbers indicate the L-RISK index for each aquifer) and (b) lumped mean residence time map in the historical period.

3.2. Mean Residence Time

The groundwater resource map (Figure 3b) and the output recharge for each aquifer from the flow model described in Section 2.2.2 were used to approximate the mean residence time at the aquifer scale as the T index following Equation (2) (Figure 5b).

Figure 5 shows clear differences in the L-RISK and T indices between the different aquifers, although a higher heterogeneity is observed in the T index values due to the fact that nearly the 70% of the total area in the Upper Guadiana Basin is occupied by non-irrigated arable land and vineyards, which represent medium hazard (value 5); therefore, the risk map was smoothed.

The L-RISK index helped to determine the most vulnerable aquifers, overcoming the drawbacks of the subjective nature of visual analysis in a distributed map [13]. Vulnerability lumped indices have been shown to be useful to compare results between different aquifers in previous studies [11,13].

Two groups of aquifers could be distinguished with the T index, with each showing significant differences in their mean values in the historical period: aquifers had a T under 20 years, and others had a T value higher than 40 years (Figure 6). The aquifers that had lower T values showed lower volumes of resource than the recharge, which makes them extremely vulnerable to pumping [20,21] and suffering drastic variations due to exploitation [2].

We also analyzed the volume of resources, recharge, and the T index in dry and wet periods (1993–1995 and 2010–2014, respectively). We observed that some aquifers experienced large variations in the T index when comparing the dry and wet periods (bars in Figure 6), which was mainly driven by groundwater recharge. Despite the variation in groundwater recharge between dry and wet periods, this component was not significant compared to the groundwater volume in aquifers with higher T values. Therefore, these aquifers always maintain a higher residence time, which means that they are more resilient to climate variability [61] but might be susceptible to groundwater depletion over long dry periods if the withdrawals are not controlled. On the contrary, other aquifers showed lower T values in both wet and dry periods, with higher contributions of recent infiltration and, thus, higher risks for pollution [62].

Figure 6. Groundwater recharge (Mm^3/year), volume of resource (Mm^3), and lumped turnover time index (T index) for each aquifer in the Upper Guadiana Basin in the historical period (mean, wet, and dry historical periods).

3.3. Lumped Validation of Pollution Risk Maps

Different regression models (Table 1) were tested in order to demonstrate the correlation between the T and L-RISK indices in the eight aquifers of the Upper Guadiana Basin. The coefficient of determination R^2 of the different models varied in the range of 0.51–0.69. The best combination of the model was then obtained for the transformation of square root in both variables ($\sqrt{L - RISK} = a \times \sqrt{T} + b$).

The results (Figure 7) showed that the T index can be a potential lumped approach to assess and/or validate the risk of pollution at the aquifer scale. This method can be useful in regional or national studies in which several aquifers are analyzed at the same time in order to establish priority management measures to achieve the objectives of the WFD. The validation of risk (or vulnerability) maps is usually performed by employing different methods according to data availability and the type of aquifer. Previous studies [13,14] that analyzed and compared groundwater vulnerability in several aquifers employed a different distributed validation method for each aquifer, thus forfeiting the advantage of a harmonized methodology that makes the results comparable.

Figure 7. Relationship between the target and explanatory variables for the best linear regression model ($\sqrt{L - RISK} = a \times \sqrt{T} + b$).

On the other hand, the T index as a lumped approach of groundwater risk for pollution can help to quantify the degradation of an aquifer due to groundwater vulnerability and recent contamination, but it would also allow for more efficient management measures to be established in order to remediate the contamination of groundwater. The renewal time of groundwater helps one to understand many problems related to groundwater supply and quality, as well as groundwater-dependent ecosystems [2,21].

The T index provides a preliminary estimation of the risk for pollution that only requires the volumes of resource and recharge. These data are usually available from river basin authorities. The use of the T index as vulnerability indicator may help to overcome the drawbacks of data availability when applying more complex methods and the ambiguity when different vulnerability methods are applied to the same pilot area.

In order to prove the potential suitability of this model to provide future estimations under climate change scenarios, we validated it in two historical periods with different climatic conditions (dry and wet historical periods). The recharge in the dry and wet periods varied between -60% and $+40\%$, respectively, compared to the average. Figure 7 shows the relationship between the transformation of the T and L-RISK indices for the mean, dry, and wet historical periods and the fitted linear regression for the mean historical values. The R^2 was similar for the three periods, which showed that the model could be useful to estimate the future mean L-RISK index if the variation of the mean future recharge regarding the mean historical one is within the range of the dry and wet historical periods.

3.4. Impacts of Future Potential Climatic Scenarios

The estimated future climatic data allowed us to estimate the impacts of climate change through the SAC-SMA and Modflow models in order to obtain the recharge and groundwater resource volume to calculate the T index. The future pumping schedule was generated by using the CROPWAT (Computer program for irrigation planning and management) model [63] to calculate net irrigation demands according to the climate change scenario. This tool allows one to estimate water requirements for each kind of crop from precipitation and temperature data. Thus, the future climatic series was applied to a business-as-usual management scenario based on the assumption by maintaining the current (2015) spatial crop distribution in the future. The forecasted increase of mean temperature and decrease of mean precipitation in the climate change scenario implies higher crop water requirements according to CROPWAT calculations. Therefore, although the future scenario does not encompass land use changes, irrigation volume will have to increase in order to maintain crop exploitation, as in the last year of the historical period (2015).

The simulated climate change scenario showed a decrease in precipitation and an increase in temperature, which involved a lower recharge (33.4 mm/year in the simulated future scenario) and a decline in groundwater levels exacerbated by higher crop water requirements. This would be reflected in a reduction of groundwater vulnerability to contamination.

The T index was found to increase in the future regarding the mean historical for all the studied aquifers (Figure 8a). It showed a heterogeneous variation depending on the aquifer. The difference between future and historical T index values was found to vary in the range from 1 to 45 years (Figure 8b). It was observed that the lowest risk for pollution correlated to the largest the difference between future and historical T index values. The differences in the T index between historical and future assessment were mainly due to the reduction in the recharge, which was larger in the aquifers with high T values. The change in the mean recharge in the future with respect to the historical reflected a decrease between 15 and 56% depending on the aquifer, whereas the volume of resource showed a maximum variation of 23%.

Figure 8. (a) Historical (medium, wet, and dry periods) and future T index and historical and future L-RISK for the best linear regression model; (b) change between future and historical T (years) and L-RISK (%) indexes.

These recharge changes were related to climate variables, which include precipitation and influence water crop requirements. The management of resources regarding exploitation due to a future pumping schedule was also considered in the Modflow simulation (explained above in this section).

A linear regression model calibrated in the historical period was employed to predict the impacts of the potential future climate change scenario on risk of pollution (L-RISK index).

Risk for pollution will decrease in the future in all aquifers. Aquifers with a high risk in the historical period will continue having high risk of pollution in the future (if the sources of contaminants do not drastically change). In relation with the potential sources of contamination, the T index have an idea of the time in which the prevention measures could be effective [64]. Aquifers with high T index values will become less vulnerable to pollution (provided that land uses do not entail a major hazard), but greater efforts will be needed in order to maintain the mean reserves due to drought exacerbation [20]. Nevertheless, if sustainable use and a good quantitative status remain over time, these aquifers could have great strategic potential in drought management because they offer high reliability to groundwater managers. They could provide significant pumping rates without varying their hydrological operations [2]. However, aquifers with younger water (lower T values) will continue to be the most impacted by contamination and climate change [21], and more efforts will be required to maintain a good qualitative status, although the effect of prevention measures is being rapidly reflected. In these aquifers, management strategies will require that the groundwater extractions are lower than the mean future recharge in order to preserve groundwater status and groundwater-dependent ecosystems [2].

In summary, the novel approach proposed in this paper can be useful in cases with limited information, even if we do not have a hydrological model. It can produce preliminary results, covering wide areas (national or regional assessment), that help to identify

potential strategic aquifers where more exhaustive analyses should be performed to define conjunctive use measures for the sustainable management of water resources systems during drought-critical periods. If data are available to perform a future assessment, the obtained results can also be useful to define sustainable groundwater planning based on the potential exploitation and risk of pollution of the groundwater. This will help to contribute the sustainable management of groundwater resources and groundwater-dependent ecosystems.

3.5. Hypotheses and Limitations

The proposed approach presented in this paper involved some hypotheses and limitations regarding data and methodology.

3.5.1. Data

- The hazard assessment considered the potential contamination from the different land uses based on the literature, but it did not consider the permissible quantity of fertilizers applicable according to the type of crop (which involves the potential load of nitrate into groundwater). This would require a complex analysis that was out of scope of this paper.
- The underlying vulnerability method may have involved some degree of subjectivity due to the own formulation of the method.
- Data to calculate the necessary variables for the T and L-RISK indices came from a calibrated flow model, which presented its own limitations.
- The climatic variables (precipitation and temperature) used in this study came from Spain02 project dataset [57]. This dataset has been employed and validated in many research studies.

3.5.2. Method

- The proposed method required a risk for pollution assessment, which could have involved some degree of subjectivity in the analysis. In this paper, a modification of the DRASTIC method (O-DRASTIC) was used to assess groundwater vulnerability, and it was combined with land uses to obtain the risk of pollution. Other vulnerability and/or risk assessment methods could be used to apply the proposed methodology.
- The T index could be obtained from available data from river basin authorities or by using a detailed groundwater model. The source of these data could influence the accuracy of results.
- We assumed a linear relationship between the mean residence time (T index) and the lumped risk of pollution (L-RISK index) in the studied aquifers. This allowed us to use a simple regression model.
- We assumed that the model was suitable in the range of variability of the future climatic variables.
- The method was demonstrated to be useful in the eight aquifers of the Upper Guadiana Basin, but it should be tested in a larger range of aquifers.
- The local climatic scenario used in this study was generated for the horizon of 2016–2045, assuming the most pessimistic emissions scenario (RCP 8.5). Observed biases were corrected by applying a simple statistical correction technique.
- The proposed method was useful to analyze the impacts of climate change on the risk of pollution at the aquifer scale, assuming that a business-as-usual management scenario was maintained in the future. Significant changes in the type of crops, which may involve higher fertilizer requirements, would have great impact on groundwater contamination, and the linear model would not allow one to approach a future assessment.

4. Conclusions

This paper presents a new method to make a preliminary assessment and/or validation of groundwater pollution risk maps at the aquifer scale by using a lumped approach of the mean residence time (T index). The study showed that the T index is a useful indicator to attempt a first approach of groundwater risk for pollution in order to determine which aquifers are in risk according to the WFD. The used method allowed us to identify potential hot spots (aquifers at risk) that will require further detailed risk assessments to analyze the most suitable management practices. The methodology assumed a linear relationship between the T and L-RISK indices, which was used to estimate the impacts of climate change under a future scenario. The methodology was applied to eight aquifers in the Upper Guadiana Basin. The results showed a good correlation between the T and L-RISK indices, not only in the mean historical period but also under variable historical climatic conditions (dry and wet historical periods). A regression model was used to estimate the future L-RISK index, which forecast a lower risk of pollution. Despite that, sustainable management will be necessary because the residence time of groundwater will also vary and involve a larger response time. The Mancha Occidental I, Mancha Occidental II, and Campo de Montiel aquifers presented the highest risk for groundwater pollution, and they were also found to have the lowest T index. Moreover, these aquifers have important associated groundwater-dependent ecosystems, and more efforts will therefore be required to maintain good qualitative and quantitative status. The T index is expected to increase in all aquifers under the future scenario. Aquifers with larger T values will become more resilient, and they could be identified as strategic aquifers in drought periods because they will allow for the maintenance of sustainable exploitation under climate change scenarios if the appropriate management measures are implemented.

In conclusion, the T index provides a rough assessment of the historical risk of pollution at the aquifer scale and gives an idea of the sustainability of groundwater resource exploitation under climate change scenarios. This index is a lumped index that can be calculated using easily available data, and it allows for results at the aquifer scale to be compared in order to establish priority measures in aquifers at risk.

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Appendix A. O-DRASTIC Method

The O-DRASTIC method was developed by the authors of [19] in an attempt to harmonize vulnerability assessments for different aquifer typologies. It was obtained by an

optimization process, and it was validated in carbonated and detrital aquifers in the Upper Guadiana Basin (Spain), ultimately showing better results than original DRASTIC [6].

This method considers the seven parameters/variables influencing the vulnerability to contamination as original DRASTIC: depth to water table (D), net recharge (R), aquifer media (A), soil media (S), topography (T), impact of vadose zone (I), and hydraulic conductivity (C). The classification and rate of importance assigned to the parameters are summarized in Table A1.

Table A1. Ranges and rates for O-DRASTIC parameters.

Groundwater Table Depth (m)	
Ranges	Ratings
0–1.5	10
1.5–4.6	9
4.6–9.1	7
9.1–15.2	5
15.1–22.9	3
22.9–30.5	2
>30.5	1
Net recharge (mm)	
Ranges	Ratings
0–50.8	1
50.8–101.6	3
101.6–177.8	6
177.8–254.0	8
>254.0	9
Aquifer media	
Ranges	Ratings
Massive shale	2
Metamorphic/Igneous	3
Weathered metamorphic/Igneous	4
Thin bedded sandstone, Limestone	6
Shale sequences	6
Massive sandstone	6
Massive limestone	8
Sand and gravel	9
Basalt	10
Karst limestone	
Soil media	
Ranges	Ratings
Thin or absent	10
Gravel	10
Sand	9
Peat	8
Shrinking and/or aggregated clay	7
Sandy loam	6
Loam	5
Silty loam	4
Clay loam	3
Muck	2
Non-shrinking and non-aggregated clay	1
Topography (% slope)	
Ranges	Ratings
0–2	10
2–6	9
6–12	5
12–18	3
>18	1

Table A1. Cont.

Impact of vadose zone	
Ranges	Ratings
Silt/Clay	1
Shale	3
Limestone	6
Sandstone	6
Bedded limestone, sandstone, shale	6
Sand and gravel with significant silt and clay	6
Metamorphic/Igneous	4
Sand and gravel	8
Basalt	9
Karst limestone	10
Hydraulic conductivity (m/d)	
Ranges	Ratings
15	1
30	2
45	4
60	6
75	8
>75	10

The values of the parameters are weighted by a weight to obtain the O-DRASTIC index, which is calculated following Equation (A1):

$$O - \text{DRASTIC index} = 1 \times D + 5 \times R + 5 \times A + 5 \times S + 1 \times T + 5 \times I + 1 \times C \quad (\text{A1})$$

The O-DRASTIC index is classified into five vulnerability levels (Table A2):

Table A2. O-DRASTIC classes.

O-DRASTIC Index	Class Value
<107	Very low
107–130	Low
130–138	Moderate
138–146	High
>146	Very high

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