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Evaluation of a GIS-Based Integrated Vulnerability Risk Assessment for the Mancha Oriental System (SE Spain)

**Ángel Moratalla · Juan José Gómez-Alday ·
David Sanz · Santiago Castaño · Jorge De Las Heras**

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Abstract It is widely recognized that groundwater-vulnerability maps are a useful tool for making decisions on designating pollution-vulnerable areas, in addition to being a requirement of European Directive 91/676/EEC. This study addressed the vulnerability of the Mancha Oriental System (MOS) to groundwater contamination with an integrated Generic and Agricultural DRASTIC model approach. In the MOS, groundwater is the sole water resource for a total population of about 275,000 inhabitants and for 1,000 km^2 of irrigated crops. DRASTIC vulnerability maps have been drawn up for two different years (1975 and 2002) in which the potentiometric surface has dropped dramatically (80 m in some areas) due to the considerable expansion of irrigated croplands. The quality of available resources has also deteriorated due to the agricultural practices and the discharge of wastewater effluents. Vulnerability maps are used to test the data on nitrate, sulphate, and chloride contents in groundwater in the Central and El Salobral-Los Llanos hydrogeologic domains of the MOS for 2002. Regardless of the method applied, the dramatic alteration in land use leads to a change in the DRASTIC index and vulnerability to groundwater contamination decreases for the study period. Vulnerability in the MOS increases in areas where the irrigation return flow is notable. The lack of a statistical correspondence between the DRASTIC index and the spatial distribution of nitrate, chloride, and sulphate contents and the distribution of the pollution load suggest that this method does not accurately assess the risk of the MOS to groundwater pollution.

Keywords Vulnerability mapping · Groundwater protection · DRASTIC model · Spain

Á. Moratalla · J. De Las Heras
Water Laboratory. Regional Center for Water Research (RCWR),
University of Castilla-La Mancha (UCLM), Ctra. de las Peñas, km 3.2, 02071 Albacete, Spain
e-mail: Angel.Moratalla@uclm.es

J. J. Gómez-Alday (✉) · D. Sanz · S. Castaño
Remote Sensing and GIS Group. Institute for Regional Development (IRD),
University of Castilla-La Mancha (UCLM), Campus Universitario, s/n, 02071 Albacete, Spain
e-mail: JuanJose.Gomez@uclm.es

1 Introduction

Vulnerability maps can be used to delineate priority areas for monitoring networks in the surveillance of potential pollution sites, to select areas for waste disposal and agricultural and industrial development, to define critical areas for the maintenance of ecosystems, and for the monitoring and assessment of transboundary groundwater systems. Groundwater vulnerability is considered an intrinsic property of groundwater that depends on its sensitivity to human and natural impacts defined by the combination of natural and human factors (Seeling and Nowatzki 2001; Babiker et al. 2005; Jamrah et al. 2008). In this sense, Vrba and Zaporozec (1994 and references therein) distinguish specific (or integrated) vulnerability when integrating the potential impacts of specific land uses and contaminants. Certain extensive agricultural practices and urban and industrial-related waste are the most significant threats to groundwater. Prevention of this kind of pollution is critical for effective groundwater management. In fact, European Directives have been established to minimize water body deterioration (OJEU 1991; OJEU 2000; OJEU 2006). The Mancha Oriental System (MOS) was declared a vulnerable zone to nitrate pollution (OJCM 2003) by the Regional Government (Junta de Comunidades de Castilla-La Mancha). Groundwater vulnerability was addressed on the basis of the nitrogen load from agricultural activities (irrigation), groundwater consumption for urban supply, nitrate contents in groundwater, and the proximity of geologic materials to the ground surface. The boundaries of the vulnerability zone established were not delineated taking into account hydrogeologic criteria, such as the hydrogeologic system boundaries.

The concept of groundwater vulnerability to contamination was introduced by Margat (1968), and since this work many techniques have been developed for analysing aquifer vulnerability. Models can be grouped into overlay/index, process-based simulation, and statistical inference approaches. A thorough overview of existing methods and their limitations can be found in Zhang et al. (1996), Tesoriero et al. (1998), Foster (2002), Babiker et al. (2005), Murray and McCray (2005), Almasri (2008), Chitsazan and Akhtari (2008), Sener et al. (2009), and Kaur and Rosi (2011), among others. For mapping vulnerability in the MOS, the DRASTIC method was selected because it is in agreement with the hydrogeological knowledge available and it is the most popular and standardized method. It was originally developed in the United States for achieving nationwide consistency (Aller et al. 1987) and has since been put into practice for different aquifer systems globally (Al-zabet 2002; Vias et al. 2005; Ettazarini 2006; Almasri 2008; Al-Hanbali and Kondoh 2008; Sener et al. 2009). The target of the model is to categorize which zones are worthy of special attention, but it is not intended to predict the occurrence of groundwater contamination. Basically, the model is constructed by designating mappable units and superposing a relative numerical rating system. As agriculture is the main activity in the study area, the Agricultural DRASTIC index (ADi) maps were developed using the weights from the pesticide DRASTIC index proposed by Aller et al. (1987). This model is a suitable method for regions where modern irrigation methods and nitrogen-based fertilizers are mainly used (Ettazarini 2006; Remesan and Panda 2008). In these areas, it is appropriate to use a method that gives more weight to soil and topography impacts. For comparison purposes, the Generic DRASTIC index (GDi) has also been calculated following Aller et al. (1987).

The DRASTIC index is very sensitive to parameter ranges and weightings and the values assigned to those parameters are modified according to the particular characteristics of the study area. This method has been successfully applied in many studies (Evans and Myers 1990; Secunda et al. 1998; Fritch et al. 2000; Baalousha 2006; Ettazarini 2006; Wen et al. 2008; Sener et al. 2009; Boughriba et al. 2010), although it does not always provide reliable estimates of the contamination potential of groundwater bodies (Stark et al. 1999; Rupert 2001; Pérez and Pacheco 2004; Stigter et al. 2006; Baalousha 2006; Almasri 2008). Usually, method validation is addressed by comparing the DRASTIC vulnerability map to groundwater contamination derived by geostatistical interpolation of hydrochemical datasets (Secunda et al. 1998; Al-Adamat et al. 2003; Antonakos and Lambrakis 2007; Assaf and Saadeh 2008; Chitsazan and Akhtari 2008; Jamrah et al. 2008; Sener et al. 2009) and as a percentage of detection frequencies (as used by Murray and McCray 2005).

Vulnerability maps for the MOS on the basis of GDi and ADi models have been drawn up for 1975 and 2002. During this period, land use changed dramatically as a consequence of an increase in irrigated cropland to about 1,000 km^2, requiring about 398 Mm^3 $year^{-1}$ of groundwater supply. Certain groundwater pollutants have increased in tandem with the expansion of the irrigated crop area and urban development. For instance, nitrate contents in MOS groundwater jumped from a mean value of 8.1 mg l^{-1} in 1975 to 33.0 mg l^{-1} in 2002. Nitrate contents in groundwater are higher than 25 mg l^{-1} in 53% of wells monitored and exceed the EU maximum allowable contaminant level of 50 mg l^{-1} (OJEU 1991) in 14% of wells sampled. Moratalla et al. (2009) point out that the nitrate source can be associated with the use of inorganic fertilizers applied to crops, although high concentrations of nitrates and chloride confined to localized areas may be related to urban pollution sources (e.g., wastewater disposal from the city of Albacete and other small towns). Other pollutants such as anthropogenically derived sulphate may be contributing to MOS groundwater degradation as well. Anthropogenic sources of these solutes include farming products (such as animal manure, fertilizer, and irrigation return flow), household sewage, landfill leachates, and industrial effluents (Hem 1986; Hudak and Sanmanee 2003). In this study, the effect of the amount of cropland under intensive irrigation on the DRASTIC index spatial variability was analysed since this change has significant consequences on groundwater flow and recharge patterns. The DRASTIC vulnerability maps for groundwater contamination were checked against nitrate, sulphate, and chloride groundwater concentrations derived by geostatistical interpolation, and the potential pollution load. We also analysed the most effective parameter for explaining both the groundwater vulnerability index and the calculated chemical species concentration maps.

2 The Study Area

The MOS extends over an area of 7,260 km^2 and is located in southeastern Spain within the catchment basin of the Júcar River (Fig. 1). The hydrogeologic system is characterized by a wide central plain (average elevation of 700 m above sea level). The topographic relief is quite flat up to the system boundaries, where the tectonics increase in complexity (Fig. 1). The Júcar River intersects the MOS, running from north to east. The Valdemembra and Ledaña streams enter from the north and flow into the Júcar River in the central part of the MOS. In addition, the Jardín and Lezuza streams enter the MOS by the southwestern boundary, and infiltrate into the plain. The Tajo-Segura Channel (140 km in length) is a hydraulic infrastructure crossing the MOS from north to south; it is used for irrigation and urban water supply in the nearby Segura River Basin. The Doña María Cristina Channel (DMCC) carries sewage from municipal wastewater effluents from Albacete and other small towns.

The regional carbonate aquifer consists of three hydrogeological units (HU) known as HU7 (Mid Jurassic), HU3 (Upper Cretaceous), and HU2 (Upper Miocene). HU6 (Upper

Fig. 1 Location of study area in the Júcar River Basin (JRB) and simplified geological map of the MOS ▶ showing sample supply wells. Segura River Basin (SRB). Hydraulic infrastructures are the Tajo-Segura Channel (TSC) and Doña María Cristina Channel (DMCC). Hydrostructural domains, Northern Domain (ND), Central Domain (CD), El Salobral-Los Llanos Domain (SLD), Moro-Nevazos Domain (MND), Pozocañada Domain (PCD), and Montearagón-Carcelén Domain (MCD). Position of hydrostratigraphic cross-sections are indicated by Roman numerals I–I′ and II-II′

Jurassic) forms a locally important carbonate aquifer (Fig. 1). The HU8 (Lower Jurassic), HU5 (Upper Jurassic), HU4 (Lower Cretaceous), and HU1 (Lower to Upper Miocene) are composed of marly to fine- to coarse-grained detrital deposits and are considered to be regional aquitard or aquiclude units. The marl and clay deposits of HU8 are regarded as the basal regional impermeable layer. HU9 (Upper Triassic) is made up of gypsum- and halite-bearing clays with heavily variegated colours. Based on the geological structure, groundwater level evolution, and hydrochemical zones, the MOS can be divided into six hydrogeologic domains (Fig. 1): the Northern Domain (ND), Central Domain (CD), El Salobral-Los Llanos Domain (SLD), Moro-Nevazos Domain (MND), Pozo Cañada Domain (PCD), and Montearagón-Carcelén Domain (MCD). The hydrochemical facies are in agreement with the carbonate and evaporite deposits from the MOS. The water type ranges from CaMgHCO3 in the north and centre to CaMgHCO3-CaMgSO4 in the south. The MOS hydrogeology is described in detail in Sanz et al. (2009, 2010). The climate in the study area is continental and semi-arid, with extreme temperatures occuring in both summer and winter. During the summer, the average monthly temperature is about 22°C. In contrast, during the winter season it is about 6°C. The mean annual precipitation (1946–2002) is close to 350 mm, ranging from 280 mm yr^{-1} in southern areas to 550 mm yr^{-1} in northern ones. The precipitation pattern has high inter-annual variability, reaching as low as 150 mm yr^{-1} in dry years and as high as 750 mm yr^{-1} in humid ones.

Irrigated crops covered an area of 170 km^2 in 1975 (Arenas et al. 1982). A remarkable increase in irrigated cropland occurred between 1985 and 2002, with approximately 1,000 km^2 of cropland currently under irrigation (Sanz et al. 2009). Groundwater abstractions (398 Mm^3 $year^{-1}$ for irrigation; 8 Mm^3 $year^{-1}$ for urban supply) are not balanced with available groundwater resources (323 Mm^3 $year^{-1}$, estimated by Estrela et al. 2004), which has brought about a progressive drop in groundwater levels that amounts to about 80 m in the SLD. The regional groundwater flow system has changed considerably in tandem with the progressive rise in groundwater abstractions. In 1975, the effects of groundwater exploitation for irrigation purposes were not noticeable, and the MOS was considered to be in steady state conditions, with regional groundwater flow converging towards the Júcar River. The expansion of irrigated cropland, though, has brought about notable changes in groundwater flow, with shifts in flow directions observable in 2002 to sectors where cones of depression had developed as a result of irrigation withdrawals. The Júcar River also changes in behaviour along the river course. The loss of saturated thickness in the aquifer system due to groundwater abstractions is reduced due to recharge from the Júcar River.

The deterioration in MOS groundwater quality is evident when comparing the hydrochemical data bases for 1970–1975 and 1998–2004 in the CD and DSL (Moratalla 2010). Mean concentrations of nitrate and sulphate in groundwater rose dramatically between 1970–1975 and 1998–2004 (Table 1) in the CD and DSL. In the CD, nitrate contents vary from below detection limits to a mean value of 31.3 mg l^{-1}; the mean sulphate concentration has also increased, from 86.9 for 1970–1975 to 170.9 mg l^{-1} in 1998–2004. In the SLD domain, mean nitrate concentrations range from below detection limits to a mean value of 23.9 mg l^{-1}. An increase in sulphate contents is also evident,

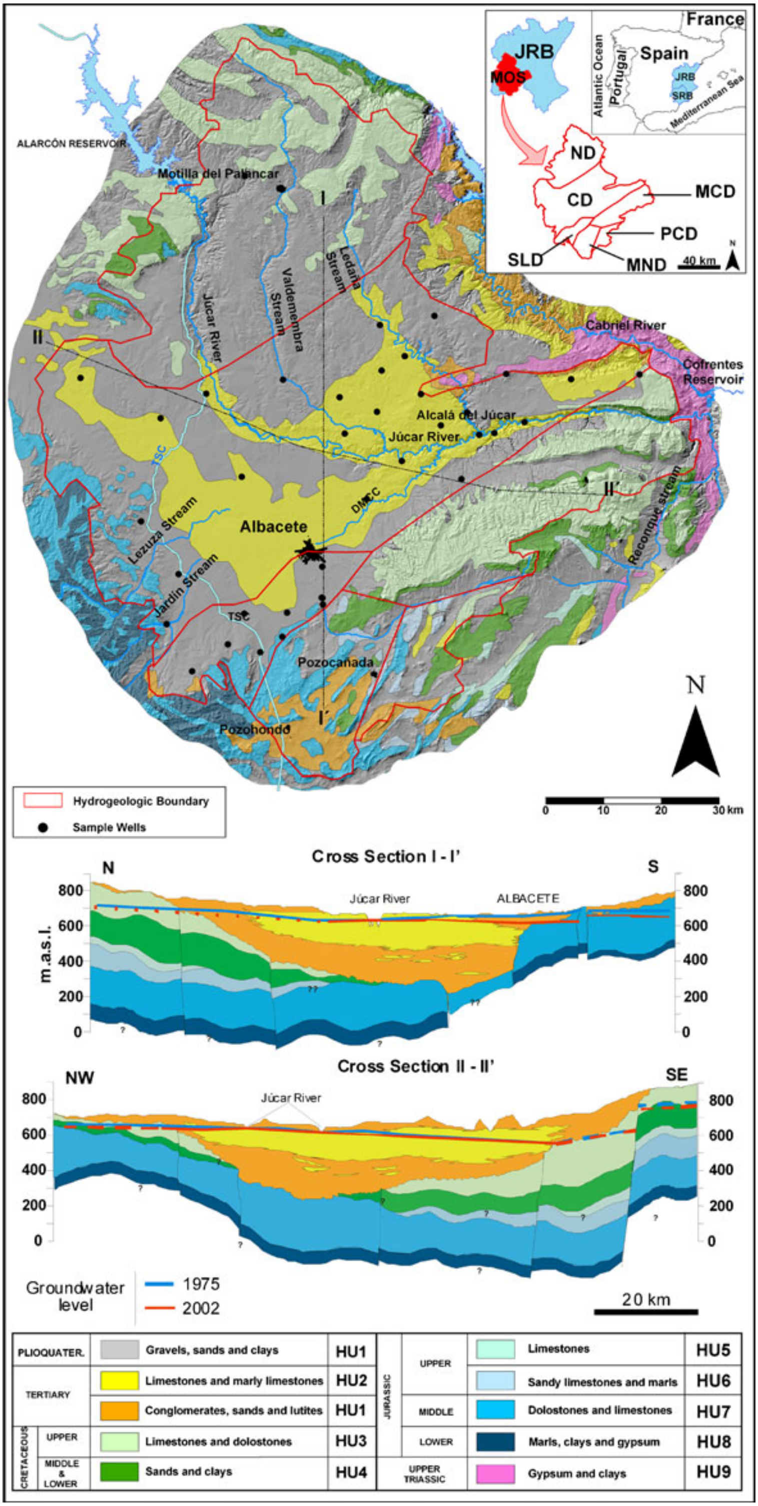

France
Spain
Portugal
Atlantic Ocean
Mediterranean Sea
JRB
SRB
MOS
ND
CD
MCD
PCD
SLD
MND
40 km
N
ALARCÓN RESERVOIR
Motilla del Palancar
Valdemembra Stream
Ledaña Stream
Júcar River
Cabriel River
Cofrentes Reservoir
Alcalá del Júcar
Júcar River
TSC
DMCC
Albacete
Lezuza Stream
Jardín Stream
Reconque stream
Pozocañada
Pozohondo
I
I´
II
II´
Hydrogeologic Boundary
Sample Wells
0 10 20 30 km
Cross Section I - I'
N
S
Júcar River
ALBACETE
m.a.s.l.
800
600
400
200
0
Cross Section II - II'
NW
SE
Júcar River
Groundwater level
1975
2002
20 km
PLIOQUATER. Gravels, sands and clays HU1
TERTIARY Limestones and marly limestones HU2
Conglomerates, sands and lutites HU1
CRETACEOUS UPPER Limestones and dolostones HU3
MIDDLE & LOWER Sands and clays HU4
JURASSIC UPPER Limestones HU5
Sandy limestones and marls HU6
MIDDLE Dolostones and limestones HU7
LOWER Marls, clays and gypsum HU8
UPPER TRIASSIC Gypsum and clays HU9

Table 1 DRASTIC ratings and weights adapted for this study. D: depth to water table; R: net recharge; A: aquifer media; S: soil media; T: topography; I: impact of vadose zone; C: hydraulic conductivity. AOS: Aridisol-Orthid-Salorthid; IOR: Inceptisol-Ochrept-Rhodoxeralf; OXI: Oxisol; PA: Poor developed or absent soil. Generic (G) and Agricultural (A) weights for DRASTIC factors are after Aller et al. (1987)

Ratings													
D (m)	R (mm)	A	S	T (%)	I	C (m/day)							
Range	Rating	Range	Rating	Range	Rating	Range	Rating	Range	Rating	Range	Rating	Range	Rating
<40	10	0–50	1	HU2	6	AOS	4	0–2	10	UH7	10	<4	2
40–80	9	50–100	3	HU3	8	IOR	5	2–6	9	UH2	8	4–12	2
80–120	7	100–180	6	HU7	10	OXI	9	6–12	5	UH1	6	12–28	4
120–160	5					PA	10	12–18	3	UH3 and UH4	4	28–40	6
160–200	2							>18	1	UH4	1	40–80	8
>200	1											>80	10
Weights													
G	5	4		3		2		1		5		3	
A	5	4		3		5		3		4		2	

rising from 148.9 for 1970–1975 to 248.5 mg l^{-1} for 1998–2004. Mean chloride concentrations show subtle variations in these periods but can reach maximum values of about 324.4 mg l^{-1} in the CD, and 57.4 mg l^{-1} in the DSL during the same time span. Minimun chloride background values are around 3.0 mg l^{-1}.

Corine Land Cover 2000 data (IGN 2000) indicates that the principal land use is agriculture, at 5,548 km^2 (75.7%). Natural vegetation covers about 1,661 km^2 (22.6%) and urban land use and surface water bodies only represent about 125 km^2 (1.7%). Dry crops occupy 4,575 km^2 and are mainly represented by cereals (barley and wheat), sunflower, and grapevines. About 973 km^2 is dedicated to irrigated crops, which are classified into summer irrigated crops (SUIC), spring irrigated crops (SIC), alfalfa, double harvest crops (DHC), and grapevines. Major SUIC include corn, sunflower, beet, onion, tomato, and green beans; barley, SIC comprise wheat and garlic; and DHC are a combination of barley and corn. During the crop seasons, nitrogen fertilizers are extensively applied and constitute the largest N-input. Dry crops showed a N consumption of about 14,757 tyr^{-1} for 2000 (Moratalla 2010). Irrigated crops occupy a smaller area but require about 12,894 tyr^{-1} of N from fertilizers. Urban and industrial sources represent 4% of the total N-input (1,359 tyr^{-1}).

3 Methodology

Seven parameters were used to calculate the DRASTIC vulnerability index: Depth to the water table, net Recharge, Aquifer media, Soil media, Topography, Impact of vadose zone, and hydraulic Conductivity (see Aller et al. 1987, for further details). The significant media classifications of each parameter represent the ranges, rated from 1 to 10 based on their relative effects on aquifer vulnerability. The seven parameters were then assigned weights ranging from 1 to 5 to reflect their relative importance in the study area (Table 1). The DRASTIC index was computed by applying a linear combination of the factors considered according to the following expression:

$$\mathrm{DRASTIC\ index} = \mathrm{DrDw} + \mathrm{RrRw} + \mathrm{ArAw} + \mathrm{SrSw} + \mathrm{TrTw} + \mathrm{IrIw} + \mathrm{CrCw} \qquad (1)$$

Where D, R, A, S, T, I, and C are the seven parameters mentioned above, and subscripts r and w are the corresponding ratings and weights, respectively. Subscript r was selected according to the Delphi technique (Aller et al. 1987). Generic (G) and Agricultural (A) weights for DRASTIC factors were taken from Aller et al. (1987) as they were considered appropriate for the MOS. The large quantity of data gathered and the size of the study area made it necessary to use a system for information organization and visualization. Alphanumeric data were georeferenced to the UTM coordinate system, European Datum 30N, Hayford's ellipsoid 1950 (ED50). The data was entered into a Microsoft Office Access® geodatabase linked to GIS (Arc-Gis 9.2®) by means of Structural Query Language. A finite difference grid with a uniform pixel size of 100×100 m^2 was performed for each of the seven parameters (thematic grid maps). The normalized work scale for the DRASTIC index obtained ranged from 1:250000 to 1:500000.

A total of 164 water samples were collected and analysed in 2002, and the Júcar River Basin Authority hydrochemical analyses were used to complete the hydrochemistry database. Nitrate, chloride, and sulphate concentrations were obtained using ionic chromatography following standard methods (APHA-AWWA-WEF 1998). Ion concentration maps were obtained using an ordinary kriging interpolation algorithm. A structural analysis of the variable to be interpolated was constructed by adjusting the experimental semivariograms on a theoretical basis to those with similar behaviour. This procedure serves to calculate a matrix of weights for each item and to estimate the statistical error affecting the interpolation. For all cases, a stationary or spherical theoretical model was chosen. Correlation coefficient and regression analysis was performed to check to what extent the GDi or ADi maps, DRASTIC parameters, and potential pollution load agreed with the spatial distribution of, nitrate, chloride, and sulphate concentrations in groundwater.

4 DRASTIC Vulnerability Index Development

4.1 Depth to Water Table

The depth to the water table is an important factor in any vulnerability model because the amount of attenuation that occurs is directly related to that depth. For example, a contaminant travelling a greater distance through the unsaturated zone is more likely to be sorbed, oxidized, or otherwise degraded below surface-water or recharge-water concentrations. The depth to the water table for 1975 and 2002 was taken from the difference between the ground surface elevation and the potentiometric surface contour lines of Sanz et al. (2009). The potentiometric surface was converted to grid format and the depth to the groundwater table was computed using the raster calculator by subtracting from the elevation map (Fig. 2a, b). The depth to the water table varies from a few metres to more than 200 m. According to this range, the parameter varied from 1 for depths greater than 200 m (least effect on vulnerability) to 10 for shallow depths (<40 m) (most effect on vulnerability) (Table 1).

4.2 Net Recharge

Pollutant transport throughout the unsaturated zone of the aquifer is produced through dissolution in the recharge water. The amount of water that reaches the groundwater table as net recharge is an important factor because the MOS is considered a phreatic multilayer aquifer. The spatial distribution of the net average annual recharge by infiltration for 1940–2002 was obtained from Font (2004). The recharge from the Lezuza and Jardín streams and the DMCC has also been considered. For this purpose, the discharge values were used from gauging stations available on the website of the Júcar River Basin authority (CHJ 2009). Discharge values were incorporated into the net recharge map by distributing these values for each pixel along the river channel.

As Rupert (2001), Stigter et al. (2006), and Assaf and Saadeh (2008) indicate, in semi-arid regions where effective precipitation is low, irrigation return flow must be considered an important component of recharge. The return flow enhances the leaching of agricultural chemicals and soil salinization through the "groundwater recycling processes" reported by Stigter and Carvalho Dill (2001) and Stigter et al. (2002). In this study, in order to obtain a more realistic approach, the spatial distribution of the net recharge related to precipitation has been modified due to irrigation practices (Fig. 2c, d). The analysis of Landsat5-TM and Landsat7-ETM+ satellite images following the methodology of Castaño (1999) and Calera et al. (2001) allows for quantification of the irrigated surface for different crops and estimates groundwater abstractions by the assignment of crop water requirement values (Martín de Santa Olalla et al. 2003). Central pivot irrigation systems and sprinklers have been the most common technologies employed in the MOS since 1990. Irrigation return flow was estimated at between 10 and 15% of applied water (see Tarjuelo 1995). The DRASTIC ratings were modified

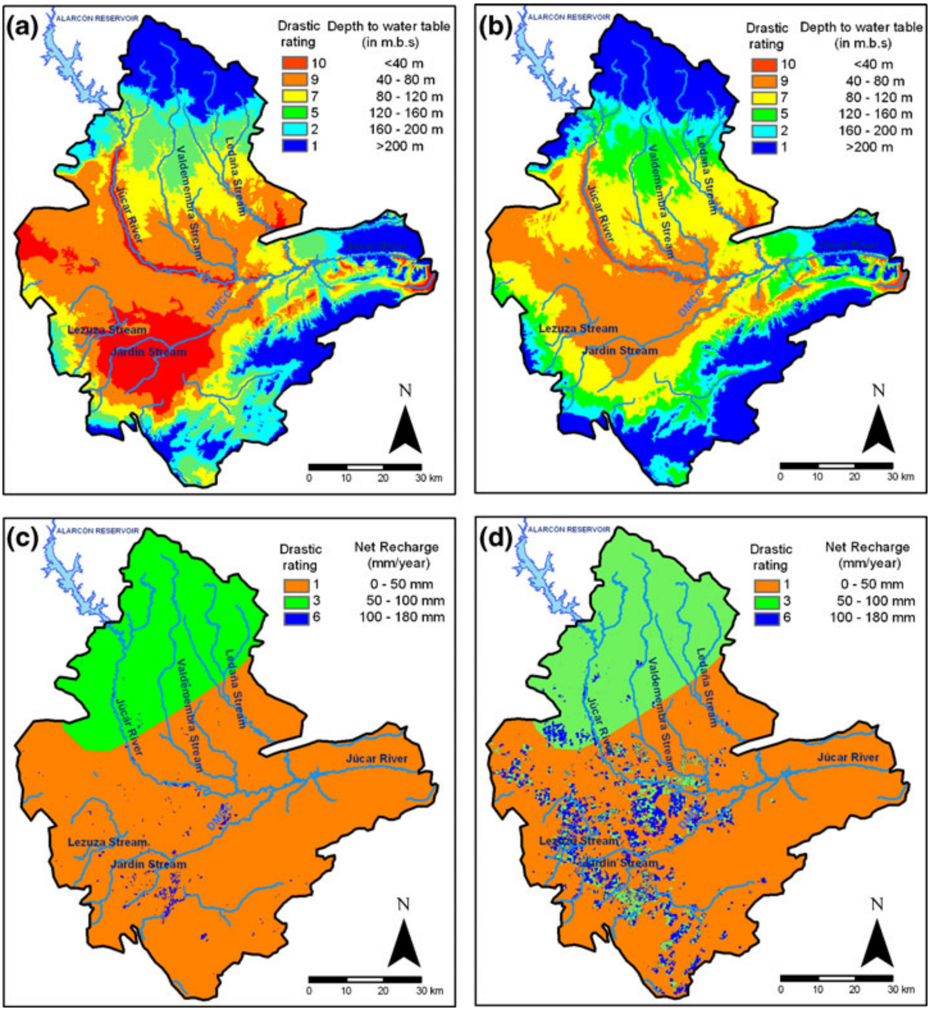


Fig. 2 Depth to groundwater level from ground surface (m.b.s) for: 1975 (**a**), 2002 (**b**) and estimated irrigation return flow for: 1975 (**c**), 2002 (**d**)

accordingly and a rating of 1 was set for low net recharge and a rating of 6 for high net recharge (Table 1).

4.3 Aquifer Media

The aquifer media factor is related to the capacity of the aquifer material to store and transport groundwater pollutants along a flow path. The three-dimensional hydrostratigraphic framework visualization of the MOS constructed by Sanz et al. (2009) was used to represent the spatial extent of the various aquifer types in the MOS vulnerability model. The elevations of the top and bottom of the HU2, HU3, and HU7 (with a total of 516 points) with detailed stratigraphic data have been interpolated through the use of geostatistical methods such as ordinary kriging interpolation. Additional information was obtained by digitizing and georeferencing the synthetic geological map (1:200000), and the analysis of the geological maps of the MAGNA series from the IGME (1:50000). The DRASTIC ratings used in this study were adjusted to account for post-depositional alteration that has occurred in the aquifer hydrogeological units. In this sense, the HU2 is considered a karstified carbonate unit that is poorly fissured. The HU7 aquifer is highly karstified and fissured and so has relatively high hydraulic conductivity and a higher aquifer rating in the DRASTIC vulnerability rating system. Aquifer HU3 is fissured and karstified, but to a lesser extent than HU7. Therefore, it has a moderate aquifer rating relative to the other units. The DRASTIC-modified ratings for aquifer media are 6 (HU2), 8 (HU3), and 10 (HU7) (Table 1; Fig. 3a).

4.4 Soil Media

Soil hydraulic properties control the amount of recharge that can infiltrate downwards. Soils in the study area are immature, with thin, poorly developed horizons (lithosoils, rendsine) heavily influenced by the parent rock. The genesis of these soils is therefore closely associated with the outcrops of carbonate units HU2, HU3, and HU7. In the study area, there are also soils on Plio-Quaternary alluvial or aeolian sedimentary deposits (regosoil). Gleysoils related to recent but inactive swamp environments can be found in areas close to the city of Albacete. Some soils show calcareous concretions of great lateral extent that can reach 1–2 m in thickness. Saline soils can also be found locally. The soil map was obtained from the National Soil Atlas of Spain at a scale of 1: 2000000 (IGN 1992), which uses the natural classification of soils according to the USDA Soil Taxonomy (USDA 1987). This thematic map has been modified based on regional geology. The rating values of the soil layer vary from 4 for those soils grouped into Aridisol-Orthid-Salorthid to 10 for soils associated with a thin or absent soil cover (Table 1; Fig. 3b).

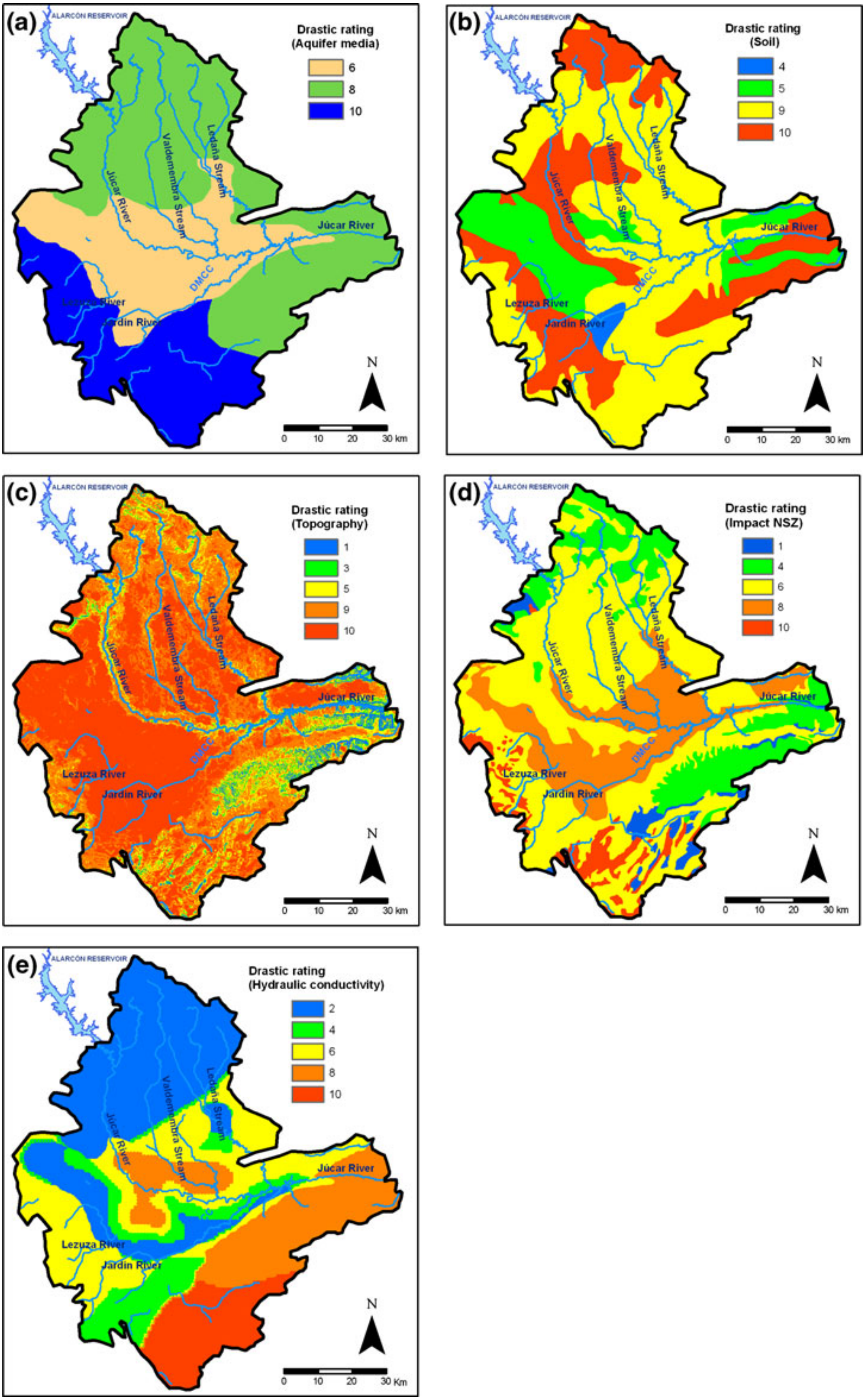
4.5 Topography

The topography governs the flow rate at the surface, which enables contaminant percolation to the saturated zone. The digital elevation model 1:25000 (DEM25) provided by the National Geographic Institute (IGN) of Spain was employed to obtain the slope map. Elevation accuracy was given by the average quadratic error (<3 m). The error in computing surfaces with profiles equals an elevation error of less than 10 m over the entire study area. Ratings were assigned according to the DRASTIC standards postulated by Aller et al. (1987) (Table 2; Fig.3c).

4.6 Impact of the Vadose zone

The properties of materials constituting the unsaturated zone may exert significant control on the transport and attenuation of pollutants to the saturated zone. The geologic map layer used for aquifer media rating has been employed for contouring the map of the impact of the vadose zone in the MOS. In consequence, the rating assigned was adapted to the materials of the study area. DRASTIC-modified ratings range from 1 (low permeability, HU4) to 10 (high permeability, HU7) (Table 1; Fig. 3d).

Fig. 3 DRASTIC ratings: aquifer media (**a**), soil (**s**), topography (**c**), impact of vadose zone (**d**), and hydraulic conductivity (**e**)


(a)
ALARCÓN RESERVOIR
Drastic rating
(Aquifer media)
6
8
10
Júcar River
Valdemembra Stream
Ledaña Stream
DMCC
Lezuza River
Jardín River
N
0 10 20 30 km
(b)
ALARCÓN RESERVOIR
Drastic rating
(Soil)
4
5
9
10
Júcar River
Valdemembra Stream
Ledaña Stream
DMCC
Lezuza River
Jardín River
N
0 10 20 30 km
(c)
ALARCÓN RESERVOIR
Drastic rating
(Topography)
1
3
5
9
10
Júcar River
Valdemembra Stream
Ledaña Stream
DMCC
Lezuza River
Jardín River
N
0 10 20 30 km
(d)
ALARCÓN RESERVOIR
Drastic rating
(Impact NSZ)
1
4
6
8
10
Júcar River
Valdemembra Stream
Ledaña Stream
DMCC
Lezuza River
Jardín River
N
0 10 20 30 km
(e)
ALARCÓN RESERVOIR
Drastic rating
(Hydraulic conductivity)
2
4
6
8
10
Júcar River
Valdemembra Stream
Ledaña Stream
Lezuza River
Jardín River
N
0 10 20 30 Km

4.7 Hydraulic Conductivity

The transport and fate of pollutants within the groundwater system depend greatly on the system capability for mass transfer. The magnitude and spatial distribution of hydraulic conductivity (C) is a key parameter to estimate pollutant transport time. C is directly related to the transmissivity (T) through the saturated aquifer thickness (b). Transmissivity (T) and specific yield (q) data were gathered in 587 boreholes employed to obtain T-q data pairs (see Sanz et al. 2009 for further details). Saturated aquifer thickness data (b) were acquired by subtracting the elevations of the top and the bottom of HU2, HU3, and HU7. Statistical analysis of the T values estimated from the empirical relationship between log-T and log-q, and estimated b values, allow for the derivation of hydraulic conductivities. DRASTIC-modified ratings range from 2 (low hydraulic conductivity, HU4) to 10 (high hydraulic conductivity, HU7) (Table 1; Fig. 3e).

5 Results and Discussion

5.1 Consequences of Irrigated Cropland Expansion on DRASTIC Vulnerability Index

According to the GDi, the vulnerability map for 1975 showed that the high, very high, and extremely high DRASTIC index values covered an area of about 3,056 km^2 (42.1%) (Table 2; Fig. 4a). The lowest values (low to very low DRASTIC index) corresponded to a total area of about 1,811 km^2 (24.9%) (Table 2; Fig. 4a). Moderate vulnerability extended over 2,395 km^2 (33.0%). The ADi results in 1975 revealed a very large increase in vulnerability (Table 2; Fig. 4c): high, very high, and extremely high values increased to 6,142 km^2 (84.6%). 181 km^2 (2.5%) of the total area was occupied by low to very low vulnerability zones. The area of moderate risk occupied about 939 km^2 (12.9%). The vulnerability maps obtained for 2002 show the result of land use changes on the ADi (Fig. 4b, d). Vulnerable areas from high, very high, and extremely high DRASTIC index values comprise 2,407 km^2 (33.2%) for the GDi and 5,977 km^2 (82.3%) for the ADi (Table 2; Fig. 4b, d). Areas classified as low to very low vulnerability represent 2,186 km^2 (30.2%) and 198 km^2 (2.7%) for the GDi and ADi, respectively (Table 2; Fig. 4b, d). For the GDi, the moderately vulnerable areas covered over 2,668 km^2 (36.7%). For ADi, areas of moderate potential risk covered 1,088 km^2 (15%).

Table 2 Classification of DRASTIC index map in terms of vulnerability for 1975 and 2002

		1975 (GDi)		2002 (GDi)		1975 (ADi)		2002 (ADi)	
Class	Range	Area (km^2)	Coverage (%)	Area (km^2)	Coverage (%)	Area (km^2)	Coverage (%)	Area (km^2)	Coverage (%)
Extremately high	>180	20	0.3	21	0.3	2018	27.8	1429	19.7
Very high	160–180	434	6.0	449	6.2	2478	34.1	2534	34.9
High	140–160	2602	35.8	1937	26.7	1646	22.7	2014	27.7
Moderate	120–140	2395	33.0	2668	36.7	939	12.9	1088	15.0
Low	100–120	1089	15.0	1376	19.0	166	2.3	181	2.5
Very Low	<100	722	9.9	810	11.2	15	0.2	17	0.2
Total		7262	100.0	7262	100.0	7262	100.0	7262	100.0

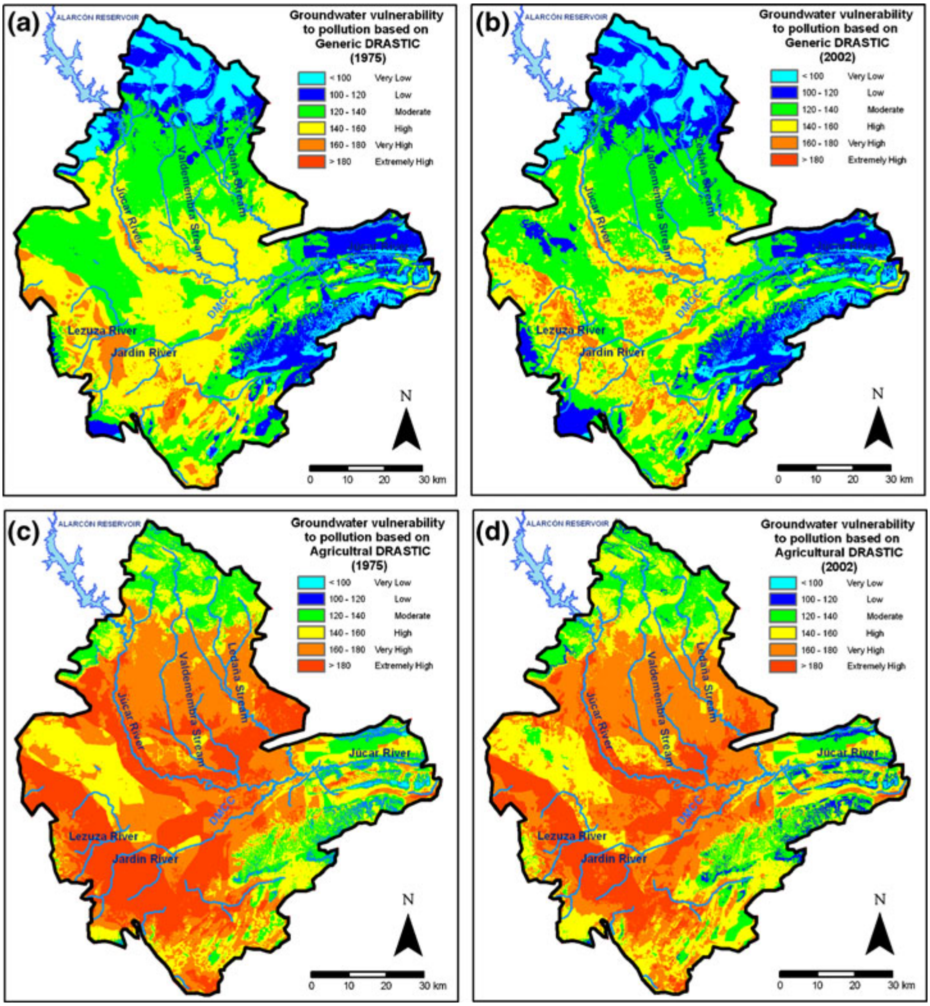


Fig. 4 MOS vulnerability maps for: General Drastic index in 1975 (**a**), General Drastic index in 2002 (**b**), Agricultural Drastic index in 1975 (**c**), and Agricultural Drastic index in 2002 (**d**)

Although the extent of the calculated vulnerability areas is different for the ADi and GDi models, it is noticeable that the location of the vulnerable areas does not change significantly between the two. As can be seen, in both models the most vulnerable zones correspond to the central and southern domains (CD, SLD, and MND) (Fig. 4). The comparison between the DRASTIC models considered reveals the high sensitivity to changes in weighting scores, but not in the distribution of the relative potential areas to groundwater degradation.

The correlation coefficient analyses between the GDi and ADi and the DRASTIC parameters allowed the vulnerability maps to be interpreted more objectively in terms of which of the hydrogeological features controlling groundwater pollution better explain the DRASTIC index distribution in the MOS (Table 3). The DRASTIC index (GDi and ADi models) is closely related to the depth of the potentiometric level, regardless of the year considered. The coefficient of this media varies from 0.77 in 1975 to 0.73 in 2002 for the

Table 3 Correlation coefficients between the Generic DRASTIC index (GDi) and Agricultural DRASTIC index (ADi) and factors for 1975 and 2002

	D	R	A	S	T	I	C
GDi 1975	0.77	−0.32	0.08	−0.04	0.35	0.64	0.25
GDi 2002	0.73	0.11	−0.07	−0.09	0.34	0.63	0.23
ADi 1975	0.74	−0.13	0.09	0.19	0.53	0.51	0.03
ADi 2002	0.69	0.20	0.00	0.17	0.52	0.53	0.03

GDi and 0.74 in 1975 to 0.69 in 2002 for the ADi. The lithology of the vadose zone is the second factor in importance and the coefficients range from 0.64 in 1975 to 0.63 in 2002 for the GDi. Topography and hydraulic conductivity can be ranked in third place when considering their influence on the GDi. In the ADi, the soil media occupies third position. The negative coefficient found between the net recharge and the DRASTIC index is significant in 1975: -0.32 for the GDi and −0.13 for the ADi. These results underscore the high vulnerability of the natural system in the central and southern domains, coinciding with the proximity to the ground surface of both the potentiometric surface and the HU2 and HU7. In these domains, there are no significant topographic differences, and the hydraulic soil properties are more favourable for contaminant percolation (Fig. 4). The system is less vulnerable to groundwater pollution in the northern sector of the ND and MCD, where the potentiometric surface is below 160 m, the HU7 is located at the greatest depths, and there is considerable topographic relief. The DRASTIC index shows moderate vulnerability coinciding with the existence of extensive HU1 deposits.

The differences in the quantified surfaces between the years considered indicate that the Generic DRASTIC model shows a regional decrease in MOS vulnerability. Nonetheless, this difference is not as notable when applying the agricultural DRASTIC scores, where high, very high, and extremely high DRASTIC index values represent a difference of 165 km^2 in the study period. Conceptually, the effect of the change in the “net recharge” on the groundwater vulnerability of the system, considering the increase in the irrigation return flow from the irrigation surface, is in opposition to the groundwater table drop. Thus, the decrease of the MOS in vulnerability to groundwater pollution due to the lowering of the groundwater level is partially compensated for by the increase in recharge, resulting in subtle changes in the ADi for the 1975 and 2000 vulnerability maps. In consequence, the MOS vulnerability increases locally in areas where the irrigated cropland has expanded due to the effect of increased irrigation return flow (Fig. 4d). Obviously, the consideration of irrigation returns as part of the net Recharge parameter in arid or semi-arid areas has considerable consequences on the resulting vulnerability maps. Therefore, net recharge is of great relevance to vulnerability modelling, as other authors have pointed out (see Murray and McCray 2005), mostly when it is not homogenously distributed in the hydrogeological system.

5.2 Testing Integrated DRASTIC Vulnerability Maps for 2002

An evaluation of the agricultural and generic DRASTIC indices has been carried out in the Central and El Salobral-Los Llanos domains for 2002, wherein the nitrate, chloride, and sulphate datasets are more complete and data points are spatially well distributed (Fig. 5). In 2002, nitrate concentrations ranged from 0.3 to 264.0 mg l^{-1} (mean value of 33.0 mg l^{-1}). Chloride contents in groundwater ranged from 5.0 to 166.4 mg l^{-1} (mean value of

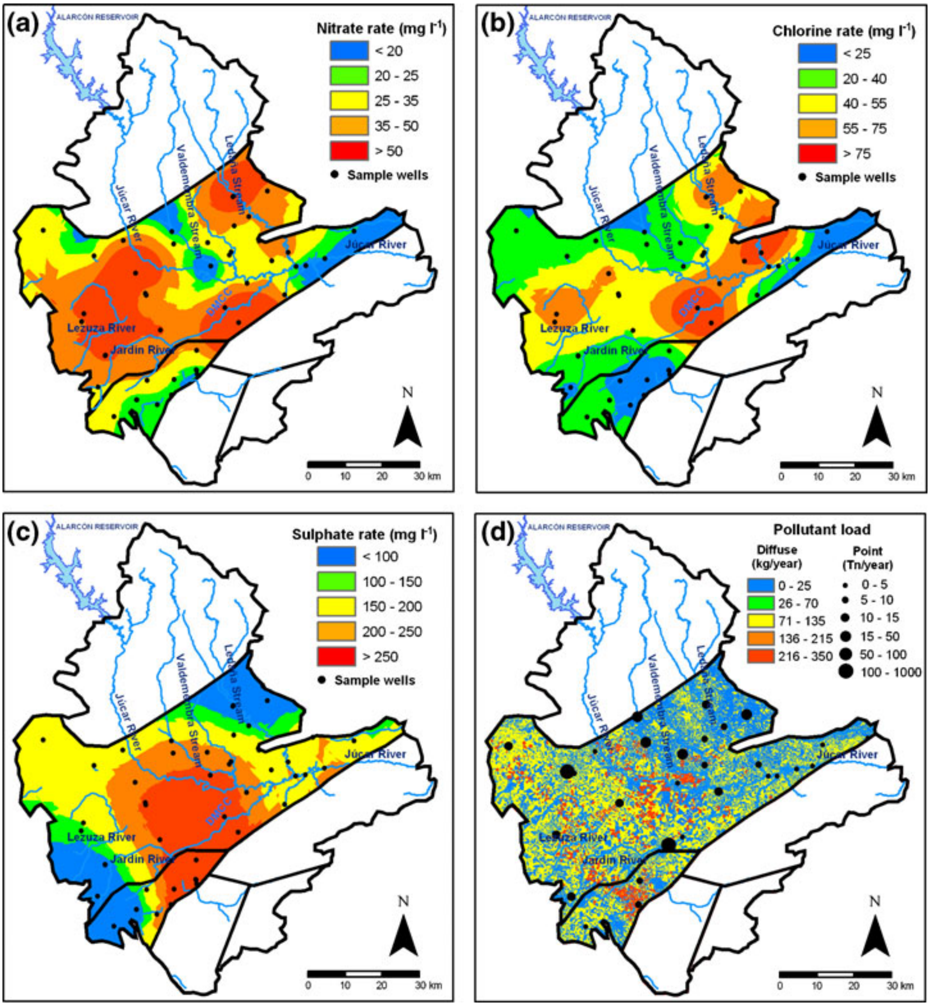


Fig. 5 Groundwater contamination maps (mg l^{-1}) for 2002 in the CD and SLD for: nitrate (**a**) chloride (**b**), sulphate (**c**), and pollutant load map (**d**)

48.5 mg l^{-1}). As for sulphate, concentrations varied from 7.3 to 1081.2 mg l^{-1} (mean value of 210.4 mg l^{-1}).

Although visual examination of the maps seems to validate the spatial distribution of the ions and vulnerability zones considered, the statistical correlation among these variables is poor even though the p-value is lower than 0.001 (Table 4). The multiple regression analyses considering the ADi or GDi and the potential pollution load (after Moratalla 2010) as independent variables, and nitrate contents in groundwater as the dependent variable, also shows p-values lower than 0.001. The R-square values indicate that the model only accounts for about 5.67% of the total nitrate groundwater content variability for the ADi and 1.08% for the GDi (Table 5). Nonetheless, it is possible that the approach carried out may not capture the probabilistic nature or the uncertainty of groundwater vulnerability, thus making the validation difficult.

Table 4 Results of the simple regression analysis between the Agricultural DRASTIC index (ADi) and Generic DRASTIC index (GDi) and nitrate, sulphate and chloride contents in the Central Domain and El Salobral-Los Llanos Domain for 2002. Std. Error: Standard error

Predictor	Coefficient	Std error	p-value	Coefficient	Std error	p-value
	ADi versus Nitrate contents			GDi versus Nitrate contents		
Constant	3.17	1.23	0.0103	22.50	1.36	<0.001
DRASTIC index	0.21	0.007	<0.001	0.12	0.001	<0.001
Square-R (%)	5.57			1.05		
	Nitrate=3.17+0.21 ADi			Nitrate=22.50+0.12 GDi		
	ADi versus Sulphate contents			GDi versus Sulphate contents		
Constant	60.68	5.19	<0.001	−34.69	5.19	<0.001
DRASTIC index	0.74	0.04	<0.001	1.56	0.04	<0.001
Square-R (%)	4.22			11.11		
	Sulphate=60.68+0.74 ADi			Sulphate=−34.70+1.56 GDi		
	ADi versus Chloride contents			GDi versus Chloride contents		
Constant	16.49	1.25	<0.001	25.53	1.37	<0.001
DRASTIC index	0.22	0.01	<0.001	0.20	0.01	<0.001
Square-R (%)	5.79			2.90		
	Chloride=16.49+0.22 ADi			Chloride=25.53+0.20 GDi		

In order to identify the most effective parameter for explaining groundwater vulnerability and the calculated nitrate, chloride, and sulphate spatial distribution layer, a correlation coefficient analysis was performed (Table 6). Statistical results suggest that the best positive correlation is between the GDi and the sulphate spatial distribution (correlation coefficient=0.31). With regard to nitrate and chloride distribution, the ADi shows a higher correlation than the GDi (Table 6). The impact of the water table depth is the most influential parameter on groundwater sulphate (correlation coefficient=0.46) and chloride (correlation coefficient=0.31) spatial distribution in the MOS. This positive correlation can be explained by the effect of the intensive groundwater abstractions, which lead to a shift to sulphate and chloride hydrofacies. Nitrates and chloride show a positive correlation

Table 5 Results of multiple regression analysis between nitrate concentration, Agricultural DRASTIC index (ADi) or Generic DRASTIC index (GDi) and pollution load (pl) in the Central Domain and El Salobral-Los Llanos Domain for 2002. Std. Error: Standard error

Predictor	Coefficient	Std error	p-value
Constant	2.20	1.69	0.081
ADi	0.22	0.01	<0.001
pl	−0.01	0.07	<0.001
Square-R (%)	5.67		
Nitrate=2.20+0.22 ADi-0.01 Land use			
Constant	24.45	1.69	<0.001
GDi	−0.14	0.01	<0.001
pl	0.17	0.07	<0.001
Square-R (%)	1.08		
Nitrate=30.50+2.29 GDi+0.17 Land use			

Table 6 Correlation matrix between Agricultural DRASTIC index (ADi), Generic DRASTIC index (GDi) DRASTIC factors and ion contents in groundwater in the Central Domain and El Salobral-Los Llanos Domain for 2002. D: depth to water table; R: net recharge; A: aquifer media; S: soil media; T: topography; I: impact of vadose zone; C: hydraulic conductivity

	GDi	ADi	D	R	A	S	T	I	C	Nitrate	Chloride
ADi	0.94										
D	0.63	0.54									
R	0.55	0.51	0.22								
A	0.04	0.15	−0.47	−0.04							
S	0.32	0.57	−0.12	0.05	0.37						
T	0.20	0.33	0.21	0.13	−0.01	−0.08					
I	0.30	0.15	0.09	0.01	−0.20	−0.10	−0.14				
C	0.26	0.18	−0.14	−0.04	0.15	0.20	−0.08	−0.15			
Nitrate	0.20	0.24	0.27	0.06	−0.07	0.08	0.19	0.00	−0.11		
Chloride	0.20	0.24	0.31	0.05	−0.27	0.17	0.05	0.11	−0.13	0.57	
Sulphate	0.31	0.21	0.46	0.21	−0.35	−0.21	0.10	0.18	−0.06	0.10	0.20

(correlation coefficient=0.57), suggesting common processes governing their transport and fate in the hydrogeological system, probably correlated with recharge patterns. Nitrate spatial distribution also shows a poor but positive relationship with topography (correlation coefficient=0.19). MOS irrigated crops occupy areas of flat relief. These areas concentrate the irrigation return flow, which leads to nitrate leaching in the unsaturated zone.

The DRASTIC index measures the potential of the MOS to facilitate downward seepage of pollutants to underlying groundwater, but does not take into account the transport and attenuation of system pollutants. MOS dilution, diffusion, and dispersion mechanisms may drive changes in hydrochemistry, evolving in the direction of groundwater flow. As Glynn and Plummer (2005, pp. 265) point out, “flow patterns in regional aquifers inferred from mapping hydrochemical facies and zones can indicate flow directions that occurred over time scales considerably greater than the time scale over which present-day, or even predevelopment water levels were established”. In consequence, the volume of aquifer system affected by pollution does not necessarily correspond to the location of the diffuse- or point-pollution sources on the land surface.

The attenuation of contaminants by natural processes may also disturb the predictable distribution of pollutants in the aquifer system. For example, denitrification processes convert harmful nitrates into innocuous nitrogen through the consumption of available electron donors such as organic matter, pyrite, or ferrous iron (Böttcher et al. 1990; Korom 1992). The aquifer-river relationships play an important role when considering the input-output mass of pollutants to the system. When the Júcar is a losing river, some pollutants can be introduced into the system. Contaminated surface waters infiltrating the aquifer may also lead to pollutant dilution. In contrast, when the Júcar is a gaining river, some pollutants can be incorporated to the discharge flow and nitrogen is exported to neighbouring hydrogeologic systems. In the MOS, a rough calculation of the nitrogen mass added to the discharge flow gives a figure of about 402 tyr^{-1}. The Jardín and Lezuza streams import about 335 t nitrogen yr^{-1} to the MOS from the nearby Jardín River Basin. On the other hand, the DRASTIC approach does not account for the effect of urban or industrial point-pollution sources (landfill sites, septic tanks, wastewater effluent discharge to channels).

6 Conclusions

This study employed both Integrated Generic DRASTIC Index and Agricultural DRASTIC models to determine the vulnerability of groundwater to contamination in the multilayer Mancha Oriental System (MOS). Results indicate that the MOS, which is under deteriorating quality conditions, is highly vulnerable to groundwater pollution. Integrated Generic DRASTIC Index and Agricultural DRASTIC Index maps show that changes in land uses can cause transient conditions that affect the spatial and temporal distribution of the DRASTIC vulnerability index in the MOS. At a regional scale, a comparison of DRASTIC maps for 1975 and 2002 indicates that the increase in irrigated cropland surface has reduced the extent of the high to extremely high vulnerability areas. Nonetheless, in areas with irrigated cropland, the DRASTIC index increased locally in 2002. It is noteworthy that the contribution of net recharge to vulnerability increased in 2002 when the irrigation return flow was significant. Statistical analyses by means of GIS tools indicate no correspondence between the spatial distribution of nitrate, chloride, and sulphate contents in groundwater with the vulnerability maps.

Although DRASTIC-derived vulnerability maps are a useful tool as a general guide for vulnerability assessment, it is evident that the method is not suitable to assess the distribution of MOS contaminants resulting from interpolating concentrations by geostatistical methods. The DRASTIC method has limitations because the robustness of the DRASTIC index depends heavily on user capabilities in representing the real system, and the numerical probabilistic occurrence of the raster layers. In addition, when considering the depth to groundwater table and the net recharge as parameters in highly modified systems, the approach is not conceptually valid since contaminants change in distribution in accordance with the flow regime and biogeochemical conditions. The poor statistical correspondence between the DRASTIC indices and the spatial distribution of chemicals suggests that the map should be validated by groundwater flow and hydrogeochemical models before interpretation and incorporation into the decision-making process. Water authorities have to take into account that vulnerability maps alter in consonance with changes in the hydrogeological system, above all concerning land-use changes, groundwater flow patterns, recharge regimes, and the natural attenuation of pollutants.

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