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# Assessment of a newly implemented irrigated area (Lerma Basin, Spain) over a 10-year period. II: Salts and nitrate exported

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ABSTRACT

Irrigated agriculture impacts the quality of water bodies receiving irrigation return flows. The leaching of salts and nitrate from Lerma Basin (7.38 $km^2$), a newly-implemented pressurized irrigated area in the Ebro Basin (Spain), was assessed in this study for the hydrological years 2004–2013, covering years before (2004–2005), during (2006–2008) and after (2009–2013) the implementation of irrigation. The concentration of salts and nitrate were measured for all the components of the water balance and the amounts of these pollutants coming from the irrigated surface (352 ha) were estimated. Besides, salt and nitrate contamination indices were computed. Under unirrigated conditions, the studied area exported 1.89 Mg $ha^{-1}$ $year^{-1}$ and 11.4 kg $ha^{-1}$ $year^{-1}$ of salts and nitrate-nitrogen, respectively. These amounts increased to 3.51 Mg $ha^{-1}$ $year^{-1}$ for salts and 30.8 kg $ha^{-1}$ $year^{-1}$ for nitrate-nitrogen after the implementation of irrigation. Salt and nitrate contamination indices (SCI and NCI, respectively) increased by a factor of three from unirrigated to irrigated conditions, reaching values of 0.96 [Mg $ha^{-1}$ $year^{-1}$]/[dS $m^{-1}$] and 0.12, respectively. Despite these values being well under the threshold considered for more sustainable irrigated areas (SCI < 2.0 [Mg $ha^{-1}$ $year^{-1}$]/[dS $m^{-1}$] and NCI < 0.2), it would be advisable to improve irrigation management to increase water use and decrease leaching. Complementary measures such as adjusting fertilization rates to temporal crop necessities or the use of catch crops may prove to be useful.


## 1. Introduction

Agriculture is recognized as the main source of diffuse pollution for both ground and surface water. Externalities of modern agriculture include downstream impacts on water quality through off-site migration of agrochemicals (e.g., Oliver and Kookana, 2012). In particular, irrigated agriculture causes enhanced impacts on water bodies receiving irrigation return flows, for surface water (Barros et al., 2012a,b; Causapé et al., 2004a; García-Garizábal et al., 2012, 2014) and ground water (Arauzo et al., 2011; García-Garizábal, 2012; Stigter et al., 2006). The leaching of salts and nitrate are of special interest when considering the main environmental impacts.

The leaching of salts is a requirement of irrigated agriculture (Letey et al., 2011), as the accumulation of salts in soil profiles is deleterious for plants, producing productivity losses and even, in extreme situations, forcing the abandonment of cultivation. Thus, there are cases where an excess of irrigation water is necessary to leach salts. Irrigation return flows with high levels of salts can impact both water supply systems and ecosystems (Duncan et al., 2008; Nielsen et al., 2003). The actual amount of leached salts depends on several factors such as climate, hydrogeological conditions or irrigation management (Causapé et al., 2004a; García-Garizábal et al., 2012). However, in newly-developed irrigated saline soils, a significant decrease in soil salinity after several years of irrigation has been reported (Wang et al., 2012). Thus it can be expected that, with adequate irrigation management, salt leaching decreases over time in newly-irrigated saline soils.

Nitrate pollution is a major concern in irrigated areas as high nitrate concentrations have been regarded as a threat for human health and ecosystems (Höring and Chapman, 2004; Sutton et al., 2011). Nitrogen is usually applied to crops in excessive amounts and leached during main rain events, although even in adequate amounts, irrigation water can also produce nitrogen leaching (García-Garizábal et al., 2012; He et al., 2012). Additionally, water availability implies in changes in the strategies of farmers, who tend to grow more productive crops and, thus, increase

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fertilization rates (Gaydon et al., 2012). Leached nitrogen not only supposes a loss of efficiency for the farmer's investment, but also implies in an increase of nitrate concentration in irrigation return flows (Arauzo et al., 2011; Merchán et al., 2013). Considerable increases in nitrate concentration of ground and surface waters have been linked to irrigated agriculture around the world (e.g., Stigter et al., 2006; Thayalakumaran et al., 2008; Lassaletta et al., 2009). As a consequence, it is expected that the implementation of irrigation increases nitrogen leaching until a new equilibrium between inputs, processes and outputs is reached.

Leaching of salts and nitrate has been studied in several irrigated areas of the Ebro River Basin (Spain). In these studies, differences have been reported between traditional flood (Causapé et al., 2004a) and modern pressurized (Tedeschi et al., 2001) irrigation systems. Besides, offsite pollution levels in rainfed areas can be close to those of irrigated systems in areas with high rainfall (Casalí et al., 2008). Even though the results of these studies are reported in unitary terms, differences in climate, hydrological conditions, and irrigation and fertilization management hinder comparisons. However, studies carried out in a same area eliminate differences, and allow for differences in offsite pollution to be related to agricultural management, given a certain grade of variety due to climatic factors.

In a basin where the main land use is irrigated agriculture, long term monitoring of irrigated agriculture is necessary to adequately assess the evolution of salts and nitrate leaching. In this context, the objective of this study was to assess the salt and nitrogen pollution induced by a newly-irrigated area during the implementation and consolidation phases of irrigated agriculture. This was carried out based on water balances performed in a companion paper for a basin meeting these requirements (Merchán et al., 2015b)

## 2. Description of study area

The complete climatic, geographical, geological and agronomical description of the study area (Lerma Basin, Zaragoza, Spain) can be found in the companion paper that presents the first part of this study (Merchán et al., 2015b). This section provides additional information on soil salinity and nitrogen fertilization, specific to the objective of this work.

### 2.1. Soil salinity

Maps of the apparent electrical conductivity ($EC_a$) were generated (Urdanoz et al., 2008; Causapé et al., 2009) to obtain better information on the spatial variability of the soil salinity in the basin. The maps showed that the average apparent electrical conductivity in horizontal configuration ($EC_{ah}$) of the Lerma Basin soils (Fig. 1) was low (0.20 dS m$^{-1}$), although depressed areas at the bottom of the Tertiary valleys presented $EC_{ah}$ values above 5.5 dS m$^{-1}$. $EC_{ah}$ and electrical conductivity of the soil saturation extract ($EC_e$) (until 1 m soil depth) were correlated and it was verified that 92% of soils of the study area up to this depth presented $EC_e$ below 4 dS m$^{-1}$, which represents a soil salinity that does not affect the performance of crops in the area.

The average apparent electrical conductivity in vertical configuration ($EC_{av}$) for the Lerma Basin (0.34 dS m$^{-1}$) was nearly twice that of $EC_{ah}$ (0.20 dS m$^{-1}$), indicating the predominance of normal profiles where soil salinity increases with depth, which is coherent with the proximity to Tertiary saline substrate (Plata, 2011). However, inverted profiles were observed in 19% of the basin surface, especially close to the gullies and in uncultivated areas. Therefore, despite the fact that most of the soils present low salinity, the presence of Tertiary materials substrata (alternating marls, clays and gypsum, ITGE, 1988) constitutes the main source of salts.

### 2.2. Nitrogen fertilization

Surveys were conducted during the years 2007, 2008 and 2012 (by phone and face-to-face enquiries) with all farmers included in Lerma Basin. The fertilization of the main crops of the study area (Merchán et al., 2015b) is presented in this section. Fertilization of maize represented an average of 352 kg N ha$^{-1}$ year$^{-1}$. The general agronomic management consisted of sowing fertilization with compound fertilizers (NPK, mainly 8-15-15 and 15-15-15), followed by multiple applications of liquid fertilizers (32% N: 16% urea and 16% ammonium nitrate) by "fertigation". Winter cereals (wheat and barley) received 143 kg N ha$^{-1}$ year$^{-1}$, distributed between the first applications as compound fertilizers a few days before sowing and the remainder as urea in early spring. Fertilization of tomato was characterized by frequent applications of small rates throughout the cycle, with the aim of overcoming the different nutritional requirements of each vegetative stage. Compound fertilizers and liquid fertilizers (fertigation through drip irrigation systems) were mainly used, with an average annual contribution of 124 kg N ha$^{-1}$ year$^{-1}$. Finally, sunflower received an average of 104 kg N ha$^{-1}$ year$^{-1}$ through two applications, the first before sowing and a side-dressing in early June.

The remaining crops constituted a minority in the Lerma Basin, associated in part with double cropping of a plot in the same year (Merchán et al., 2015b). The main associations were peas with maize or sunflower, and winter cereal with sunflower, maize or other summer crops. When winter cereal was followed by maize cultivation in the same year and plot, N fertilization reached 410 kg N ha$^{-1}$ year$^{-1}$. Although existent, the application of organic fertilizers such as manure or pig slurry was minimal and not used systematically (only in a few plots and punctually). In addition, there was no isotopic evidence of organic origins for nitrate dissolved in either ground or surface water (Merchán et al., 2014).

## 3. Methodology

### 3.1. Salt balance and exported nitrate

Salt balances were carried out and the nitrate exported was quantified during the period 2004–2013 for the irrigable area (352 ha). Salt and nitrate concentrations were assigned to the water balance components (Merchán et al., 2015b). Null concentrations were assigned to evapotranspiration, wind drift and evaporation losses from sprinkler irrigation and storage in the soil, because these components have negligible concentrations or represent negligible contribution to salt balance and nitrate exported.

The salt and nitrate concentrations of precipitation ($P$) were determined from the average of eleven samples collected by a pluviometer with an applied sheet of liquid paraffin, in the study area. Average precipitation total dissolved solids ($P_{TDS}$) and nitrate-nitrogen ($P_{NO_3-N}$) concentrations were 84 and 0.1 mg L$^{-1}$, respectively. The salt and nitrate concentrations of irrigation water were determined from the average of six samples taken from the irrigation canal ($I_{TDS}$ = 275 mg L$^{-1}$; $I_{NO_3-N}$ = 0.5 mg L$^{-1}$), which presented low variability (coefficient of variation: $CV_{TDS}$ = 9%, $CV_{NO_3-N}$ = 7%).

A great sampling effort was made to determine the concentration of salt and nitrate in the water exported from the Lerma Gully. This gully was initially sampled on a monthly basis until construction of the gauging station in 2005 (Fig. 1), when an automatic sampler (ISCO Model 3700) allowed for daily sampling. Water samples collected in the gully were transported to the laboratory, where the electrical conductivity of water corrected to 25 °C (EC) and nitrate-nitrogen concentration ($NO_3$—N) were determined. An Orion-5 Star conductivimeter was utilized for the conductivity

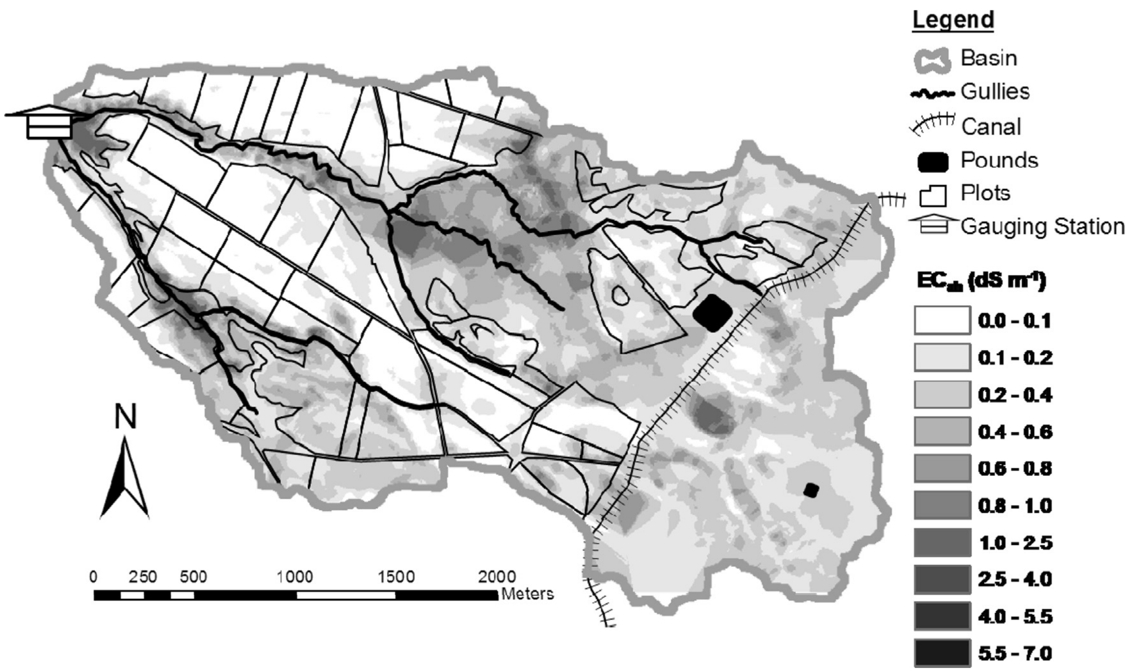


**Fig. 1.** Apparent electrical conductivity in horizontal configuration ($EC_{ah}$) for the soils of the Lerma Basin. Selected hydrological, irrigation and monitoring features.

measurements, and an AutoAnalyzer 3 equipment was used to determine $NO_3$—N by colorimetry. In July 2010 the daily sample procedure was replaced by an automatic water quality station which recorded, in 10-min time steps, EC and $NO_3$—N through a conductivimeter and a Hach-Lange Nitratax equipment connected to an ad hoc designed data-logger.

Seventeen water samples were selected within the range of variation of EC in the Lerma Gully, from which the concentration of bicarbonate ($HCO_3^-$) and dry residue (DR) were determined. From these concentrations, the total dissolved solids (TDS) were calculated from the following equation (Custodio and Llamas, 1983):

$$\mathrm{TDS} = \mathrm{DR} + \frac{1}{2}\mathrm{HCO}_3^- \tag{1}$$

For these water samples, the relationship between EC (dS $m^{-1}$) and TDS (mg $L^{-1}$) was used to convert EC into TDS through the following equation:

$$\mathrm{TDS} = 712.22 \times \mathrm{EC} - 104.83; \quad R^2 = 0.99; \quad p < 0.01 \tag{2}$$

Eq. (2) was also used to estimate the total dissolved solids in the incoming water flows from the unirrigated area and water in the aquifer. The concentration of salt and nitrate of these waters was estimated from water samples manually collected at a gully before entrance into the irrigable area and in a net of piezometers. Incoming water flows from the unirrigated area were sampled monthly, and the salts and nitrate concentrations were assigned to the entire month. This component had relatively low influence on the water balance (6% of inputs) and therefore its influence on salts and nitrate balances is expected to be marginal. Variations of salt and nitrate stored in the aquifer were estimated through annual samples taken on October 1st, at the closure of water balances, and the variation in the saturated thickness in three representative piezometers (Merchán et al., 2015b), located in each of the Quaternary materials units (lowest salinity surfaces in Fig. 1).

The product of concentrations and water volumes yields the mass of salt and nitrate for each component of the water balance. Note that herein the mass of nitrogen is estimated in the form of nitrate and not total nitrogen. However, it is considered a reasonable approach since nitrate accounted for 90% of total nitrogen in the Lerma Basin waters (Merchán et al., 2014). Nevertheless, this estimation should be considered as a minimum, and the actual amount of N leaving the basin may be slightly higher.

The difference between inputs ($P$: precipitation, $I$: irrigation, IWF: incoming water flows from the unirrigable area included in the basin), outputs (LG: Lerma Gully) and storage ($A$: aquifer) of the salinity balance corresponds mainly to the end result of the set of dissolution-precipitation processes. The influence of fertilizers was not considered in salt balances, since dissolution of soluble salts was the main salinization process and independency between nitrate pollution and salinization processes was reported in the study area (Merchán et al., 2015a). Plant uptake of salts was considered negligible in this study.

$$\mathrm{Input} - \mathrm{Outputs} - \mathrm{Storage} = (P + I + \mathrm{IWF}) - (\mathrm{LG}) - (A) \tag{3}$$

In the case of nitrate, the objective was not necessarily to close a balance but to quantify the masses exported as well as the contribution from precipitation, irrigation and the unirrigated area in the basin. The difference between inputs, outputs and storage can be attributed to the combination of nitrogen cycle components not taken into account (e.g., fertilization inputs, crop exports, volatilization of ammonia in mineral fertilizers, nitrification and denitrification).

### 3.2. Contamination indices

The unit masses of exported salts and nitrate from the irrigable area ($D_S$ and $D_N$, respectively) were obtained after discounting the contaminants introduced by the incoming water flows from the unirrigated area. Two indices proposed by Causapé (2009) were applied to further analyse the agro-environmental impact produced: salt contamination index (SCI) and nitrate contamination index (NCI):

$$\mathrm{SCI} = \frac{D_S\left[\mathrm{Mg\ ha^{-1}\ year^{-1}}\right]}{\mathrm{EC_{NI}}\left[\mathrm{dS\ m^{-1}}\right]} \tag{4}$$

$$\mathrm{NCI} = \frac{D_N\left[\mathrm{kg\ ha^{-1}\ year^{-1}}\right]}{\mathrm{FN}\left[\mathrm{kg\ ha^{-1}\ year^{-1}}\right]} \quad (5)$$

where $EC_{NI}$ is the average electrical conductivity of drainage during the unirrigated season of every year and represents the salinity of the geological materials of a specific irrigated land. FN represents the fertilization needs, which were annually estimated from the surfaces occupied by crops, harvest production, and the nitrogen extracted by the crops (Orús and Sin, 2006).

These two indices allow for the comparison of the impacts generated in different areas by irrigation. Comparison is possible as the indices adjust the exported unit masses by natural factors such as salinity from the local geology in the case of SCI, and climatic and socioeconomic factors (e.g., crops planted and respective nitrogenous fertilization requirements) in the case of NCI. Therefore, these indices are more permissive to unfavourable irrigated areas, presenting limits that well-managed irrigation areas with different characteristics should not exceed.

## 4. Results and discussion

### 4.1. Salts

#### 4.1.1. Salts balance

The main entrance of salts into the system corresponded to the inputs through irrigation water, representing 45% of total inputs over the study period, despite the low salinity of irrigation water ($EC < 0.4\ dS\ m^{-1}$) (Table 1). However, water flows from the unirrigable zone accounted for 43% of inputs. Precipitation presented the lowest salinity ($EC < 0.1\ dS\ m^{-1}$) of all components considered, contributing with 12% of the salt inputs. Note that other salt entrances, such as those carried by fertilizers, have not been considered herein since the salinization of water has been mainly attributed to natural ongoing dissolution processes (Merchán et al., 2015a).

The only output of salts considered was the water drained by the Lerma Gully (Table 1). Before the irrigation period, an average of $3.17\ Mg\ ha^{-1}\ year^{-1}$ was being exported whereas with the implementation of irrigation the output of salts increased up to $4.35\ Mg\ ha^{-1}\ year^{-1}$. Such an increase was mainly due to an increase in discharges (Merchán et al., 2015b) as the water salinity in the Lerma Gully decreased with the implementation of irrigation (Merchán et al., 2013).

As occurred for the water balances previously performed in Lerma (Merchán et al., 2015b) in the years when estimation of aquifer storage could be carried out, there were both negative and positive values, covering a range from $-0.50$ to $+0.48\ Mg\ ha^{-1}\ year^{-1}$. A clear pattern was not apparent and the value of each year was dependent on the specific hydrological conditions of that year. The average storage for the entire study period was close to zero and therefore negligible in the long term balances.

The results of the annual balance show a predominance of dissolution processes over precipitation processes (IN–OUT–ST < 0, Table 1), with the only exception in 2007 when the aquifer storage was not estimated. Under unirrigated conditions, the dissolved mass of salts ranged between $-1.03\ Mg\ ha^{-1}\ year^{-1}$ ($p_{2005} = 212$ mm) and $-2.04\ Mg\ ha^{-1}\ year^{-1}$ ($p_{2004} = 627$ mm), whereas in irrigated conditions the dissolved mass of salts reached up to $-5.54\ Mg\ ha^{-1}\ year^{-1}$. Before irrigation, net dissolved masses reached $-1.53\ Mg\ ha^{-1}\ year^{-1}$, whereas at the end of the balance net dissolved masses reached $-1.76\ Mg\ ha^{-1}\ year^{-1}$, showing the increased dissolution caused by the addition of irrigation water.

Causapé et al. (2004b) had already identified dissolution processes as responsible for increases in salt contents in another irrigated area close to the study zone, with similar geological

**Table 1**
Salt masses. Annual and cumulative inputs (IN) [precipitation (*P*), irrigation (*I*), incoming water flows (IWF) from the unirrigable area]; outputs (OUT) [water drained through the Lerma Gully (LG)]; and storage (ST) [in the aquifer (ΔA)] and differences between inputs, outputs and storage (IN–OUT–ST) for the years 2004–2013. Cumulative values are presented with annual figures to facilitate comparison.

| Year | IN | | | | | | | | OUT | | ST | | IN-OUT-ST | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | *P* | | *I* | | IWF | | Σ In | | LG | | ΔA | | | |
| | Annual | Cumulative | Annual | Cumulative | Annual | Cumulative | Annual | Cumulative | Annual | Cumulative | Annual | Cumulative | Annual | Cumulative |
| | $Mg\ ha^{-1}\ year^{-1}$ | | | | | | | | | | | | | |
| 2004 | 0.53 | 0.53 | 0.00 | 0.00 | 1.86 | 1.86 | 2.40 | 2.40 | 4.47 | 4.47 | – | – | −2.04 | −2.07 |
| 2005 | 0.19 | 0.36 | 0.00 | 0.00 | 0.71 | 1.28 | 0.90 | 1.65 | 1.88 | 3.17 | – | – | −1.03 | −1.53 |
| 2006 | 0.37 | 0.37 | 0.49 | 0.16 | 1.35 | 1.31 | 2.21 | 1.84 | 3.22 | 3.19 | – | – | −0.95 | −1.35 |
| 2007 | 0.34 | 0.36 | 1.24 | 0.43 | 1.18 | 1.27 | 2.76 | 2.07 | 2.68 | 3.06 | – | – | 0.04 | −0.99 |
| 2008 | 0.31 | 0.35 | 1.48 | 0.64 | 1.08 | 1.24 | 2.87 | 2.23 | 3.68 | 3.19 | – | – | −0.81 | −0.96 |
| 2009 | 0.32 | 0.34 | 1.54 | 0.79 | 1.16 | 1.22 | 3.02 | 2.36 | 8.45 | 4.06 | 0.24 | 0.04 | −5.54 | −1.74 |
| 2010 | 0.26 | 0.33 | 1.55 | 0.90 | 1.00 | 1.19 | 2.81 | 2.42 | 5.41 | 4.25 | −0.20 | 0.01 | −2.57 | −1.84 |
| 2011 | 0.28 | 0.33 | 1.56 | 0.98 | 1.01 | 1.17 | 2.86 | 2.48 | 4.13 | 4.24 | −0.48 | −0.06 | −0.63 | −1.71 |
| 2012 | 0.20 | 0.31 | 1.70 | 1.06 | 0.69 | 1.12 | 2.59 | 2.49 | 2.82 | 4.08 | −0.13 | −0.06 | −0.16 | −1.53 |
| 2013 | 0.40 | 0.32 | 1.45 | 1.10 | 1.49 | 1.15 | 3.35 | 2.58 | 6.72 | 4.35 | 0.50 | −0.01 | −3.96 | −1.76 |

**Table 2**
Annual and cumulative mass of exported salts from the irrigable area ($D_S$), average electrical conductivity of drainage during the unirrigated period ($EC_{NI}$) and salt contamination index (SCI) during the study years (2004–2013). Cumulative values are presented with annual figures to facilitate comparison.

| Year | $D_S$ | | $EC_{NI}$ | | SCI | |
|---|---|---|---|---|---|---|
| | Annual | Cumulative | Annual | Cumulative | Annual | Cumulative |
| | $Mg\ ha^{-1}\ year^{-1}$ | | $dS\ m^{-1}$ | | $[Mg\ ha^{-1}\ year^{-1}]/[dS\ m^{-1}]$ | |
| 2004 | 2.60 | 2.60 | 4.89 | 4.89 | 0.53 | 0.53 |
| 2005 | 1.17 | 1.89 | 5.45 | 5.17 | 0.21 | 0.36 |
| 2006 | 1.87 | 1.87 | 3.79 | 3.79 | 0.49 | 0.49 |
| 2007 | 1.50 | 1.68 | 3.78 | 3.79 | 0.40 | 0.44 |
| 2008 | 2.60 | 1.99 | 4.24 | 3.93 | 0.61 | 0.51 |
| 2009 | 7.29 | 3.32 | 4.59 | 4.10 | 1.59 | 0.81 |
| 2010 | 4.41 | 3.53 | 3.13 | 3.91 | 1.41 | 0.90 |
| 2011 | 3.12 | 3.46 | 2.81 | 3.72 | 1.11 | 0.93 |
| 2012 | 2.13 | 3.27 | 3.15 | 3.64 | 0.68 | 0.90 |
| 2013 | 5.23 | 3.51 | 3.86 | 3.67 | 1.36 | 0.96 |
| Trend[a] | ns | ** | ns | *** | ns | * |
| Slope[a] | – | +0.23 | – | −0.11 | – | +0.06 |

ns: Non significant trend ($p > 0.1$).
[a] Significant trend and slope with Mann–Kendall test and Sen's slope (Helsel and Hirsch, 2002).
* $p < 0.1$.
** $p < 0.05$.
*** $p < 0.01$.

settings. In addition, Merchán et al. (2015a) identified the main processes in the same study area through an extensive geochemical sampling. They concluded that dissolution of halite and gypsum were the main reason for the observed increase in salinity.

### 4.1.2. Salt contamination index

The unirrigated years presented the lowest value of masses of salts exported in the dry year 2005, 1.17 $Mg\ ha^{-1}$ (Table 2) and 2.60 $Mg\ ha^{-1}$ in the wet year 2004, covering a range of unirrigated conditions. These values were lower than others reported in unirrigated agricultural areas with similar precipitation but more saline groundwater in Western Australia (2.9 $Mg\ ha^{-1}\ year^{-1}$, Salama et al., 1993). With irrigation implementation, exports values up to 7.29 $Mg\ ha^{-1}$ were reached in 2009. Before irrigation, 1.89 $Mg\ ha^{-1}\ year^{-1}$ was exported whereas during the irrigated period salts exported increased up to 3.51 $Mg\ ha^{-1}\ year^{-1}$.

Duncan et al. (2008) reviewed several studies of salt leaching from irrigated areas in Australia, Spain and United States, reporting ranges of salt loads between 0.04 and 40.9 $Mg\ ha^{-1}\ year^{-1}$. This range was narrowed in the irrigated area of the Ebro Basin from 0.67 $Mg\ ha^{-1}\ year^{-1}$ in low salinity pressurized irrigated areas (Andrés and Cuchí, 2014) to 19.8 $Mg\ ha^{-1}\ year^{-1}$ in high salinity flood irrigated soils (Barros et al., 2012a). As seen, irrigated agriculture presents a wide range of salt leaching depending on differences in hydrological settings, irrigation and drainage management, or climatic trends (Duncan et al., 2008).

Increasing salt loads in Lerma Basin contrast with the decrease of salts available in the study zone, indicated by the electrical conductivity of the water during the unirrigated period, with a significant decreasing trend of -0.11 $dS\ m^{-1}\ year^{-1}$ (Table 2). The reported increases in irrigation drainage with the implementation of irrigation (Merchán et al., 2015b) explain the rising values of the mass of salts exported (+0.23 $Mg\ ha^{-1}\ year^{-2}$) as well as the salt contamination index (annual +0.06 $[Mg\ ha^{-1}\ year^{-1}]/[dS\ m^{-1}]$). In fact, implementation of irrigation supposed an increase from 0.36 (unirrigated conditions) to average values of 1.23 $[Mg\ ha^{-1}\ year^{-1}]/[dS\ m^{-1}]$ in the period in which irrigation was consolidated (2009–2013). These values were in the range of other pressurized irrigated areas of the Ebro Basin, where SCI values between 0.3 and 1.7 $[Mg\ ha^{-1}\ year^{-1}]/[dS\ m^{-1}]$ were obtained for low saline and very high saline soils, respectively (Table 3, Andrés and Cuchí, 2014; Tedeschi et al., 2001). Flood irrigated areas with low and high natural salinity presented higher SCI, reaching 4.3 and 10.7 $[Mg\ ha^{-1}\ year^{-1}]/[dS\ m^{-1}]$, respectively (García-Garizábal et al., 2014; Barros et al., 2012a).

Since the irrigated surface and irrigation rates have stabilized, it is expected that the system has reached a new steady state, and variations in the different years are due to variation in water availability (both by precipitation and irrigation rates). However, confirmation in the long term is strongly recommended (Thayalakumaran et al., 2007) as changes in drainage water salinity have been observed in other saline irrigation areas after more than 50 years of irrigation implementation (Barros et al., 2012a).

## 4.2. Nitrate

### 4.2.1. Exported nitrate

Among the components directly estimated in this study, the most significant nitrate input in the Lerma Basin during the study period was irrigation water (corresponding to 57%), followed by incoming flows from unirrigable areas (33%) and rain water (10%, Table 4). Annually, these percentages varied depending on the volume of rainfall and irrigation applied: the greatest components were rainwater and incoming flows from the unirrigated area at the beginning of the study (2004–2006) and then major component was irrigation (2007–2013). These water balance components presented low concentrations of nitrate (average values of 0.4 $mg\ L^{-1}$ for precipitation water, 2 $mg\ L^{-1}$ for irrigation water and 8 $mg\ L^{-1}$ for incoming flows from unirrigable areas).

Before implementation of irrigation, Lerma Gully transported 14.26 $kg\ NO_3{-}N\ ha^{-1}\ year^{-1}$ in the wet year of 2004 and 9.27 $kg\ NO_3{-}N\ ha^{-1}\ year^{-1}$ in the dry year 2005. These values were higher than those observed for the first years of irrigation (2006–2007) due to remaining N from when the basin was rainfed-cultivated only (stored in soils and aquifer of the basin before irrigation implementation). The absence of fertilization coupled with the lack of rain caused a decrease in the nitrate exported in 2005 (9.1 $kg\ NO_3{-}N\ ha^{-1}$) and 2006 (6.5 $kg\ NO_3{-}N\ ha^{-1}\ year^{-1}$).

The gradual implementation of irrigation, which began in 2006, led to a progressive increase in the use of fertilizers in the area, increasing the nitrogen applied as well as the $NO_3{-}N$ loads in the gully. The greatest mass of nitrate flowing in Lerma Gully was recorded in 2009 (56.61 $kg\ NO_3{-}N\ ha^{-1}\ year^{-1}$), right after high fertilization needs (294 $kg\ N\ ha^{-1}$) were recorded in 2008. These dynamics suggest a delay of 1–2 years in the response to N inputs.

**Table 3**
Irrigation efficiency (IE), mass of salts exported by the irrigated area ($D_S$), electrical conductivity of the outlet water during the non irrigated period ($EC_{NI}$), salt contamination index (SCI), mass of nitrate-nitrogen exported by the irrigated area ($D_N$), fertilization needs (FN) and nitrate contamination index (NCI) for several irrigation districts in the Ebro Basin.

| Study area Irrig. district | Irrig. system | Study period | Area (ha) | IE (%) | $D_S$ (Mg ha$^{-1}$ year$^{-1}$) | $EC_{NI}$ (dS m$^{-1}$) | SCI [Mg ha$^{-1}$ year$^{-1}$]/[dS m$^{-1}$] | $D_N$ (kg $NO_3$—N ha$^{-1}$ year$^{-1}$) | FN (kg N ha$^{-1}$ year$^{-1}$) | NCI (–) |
|---|---|---|---|---|---|---|---|---|---|---|
| DXIX[a] Bardenas I | Flood | 2001 | 100 | 56 | 4.5 | 1.1 | 4.3 | 101 | 115 | 0.88 |
| | (improved) | 2005–2008 | | 83 | 1.3 | 1.1 | 1.2 | 51 | 76 | 0.67 |
| La Violada[b] | Flood | 1996–1998 | 4k | 47 | 19.8 | 1.8 | 10.7 | 106 | 242 | 0.44 |
| Monegros II | (improved) | 2006–2008 | | 58 | 10.2 | 2.0 | 5.0 | 22 | 115 | 0.19 |
| Lasesa[c] Alto Aragón | Pressurized mature | 2010 | 1.4k | 76 | 0.5 | 1.5 | 0.3 | 49 | 288 | 0.17 |
| DIX & DXI[d] | Pressurized | 1997–1999 | 494 | 80 | – | – | – | 18 | 225 | 0.08 |
| Monegros II | | 1997–1998 | 470 | | 14.0 | 8.4 | 1.7 | 49 | 223 | 0.22 |
| Lerma Basin Bardenas II | Pressurized recent | 2009–2013 | 352 | 76 | 4.4 | 3.5 | 1.2 | 41 | 267 | 0.16 |

[a] García-Garizábal et al. (2012, 2014).
[b] Barros et al. (2012a,b).
[c] Andrés and Cuchí (2014).
[d] Tedeschi et al. (2001) (salts data), Cavero et al. (2003) (nitrate data).

**Table 4**
Nitrate masses. Annual and cumulative inputs (IN) [precipitation ($P$), irrigation ($I$), incoming water flows (IWF) from the unirrigable area]; outputs (OUT) [water evacuated through the Lerma Gully (LG)]; and storage (ST) [in the aquifer ($\Delta A$)] and differences between inputs, outputs and storage (IN–OUT–ST) for the years 2004–2013. Cumulative values are presented with annual figures to facilitate comparison.

| Year | IN | | | | | | | | OUT | | ST | | IN–OUT–ST | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | $P$ | | $I$ | | IWF | | $\Sigma$ IN | | LG | | $\Delta A$ | | Annual | Cumulative |
| | Annual | Cumulative | Annual | Cumulative | Annual | Cumulative | Annual | Cumulative | Annual | Cumulative | Annual | Cumulative | | |
| | kg $NO_3$—N ha$^{-1}$ year$^{-1}$ | | | | | | | | | | | | | |
| 2004 | 0.53 | 0.53 | 0.00 | 0.00 | 0.50 | 0.50 | 1.03 | 1.03 | 14.26 | 14.26 | – | – | −13.23 | −13.23 |
| 2005 | 0.19 | 0.36 | 0.00 | 0.00 | 0.21 | 0.36 | 0.40 | 0.72 | 9.27 | 11.77 | – | – | −8.87 | −11.05 |
| 2006 | 0.37 | 0.36 | 0.80 | 0.27 | 0.38 | 0.36 | 1.55 | 0.99 | 6.93 | 10.15 | – | – | −5.38 | −9.16 |
| 2007 | 0.34 | 0.36 | 2.04 | 0.71 | 0.33 | 0.36 | 2.71 | 1.42 | 11.89 | 10.59 | – | – | −9.18 | −9.16 |
| 2008 | 0.31 | 0.35 | 2.44 | 1.06 | 0.31 | 0.35 | 3.05 | 1.75 | 22.34 | 12.94 | – | – | −19.29 | −11.19 |
| 2009 | 0.32 | 0.34 | 2.53 | 1.30 | 0.33 | 0.34 | 3.18 | 1.99 | 56.61 | 20.22 | −6.0 | −1.01 | −47.38 | −17.22 |
| 2010 | 0.26 | 0.33 | 2.54 | 1.48 | 0.33 | 0.34 | 3.13 | 2.15 | 40.50 | 23.11 | −3.6 | −1.38 | −33.77 | −19.59 |
| 2011 | 0.28 | 0.32 | 2.57 | 1.61 | 0.29 | 0.34 | 3.14 | 2.27 | 36.86 | 24.83 | −0.8 | −1.31 | −32.89 | −21.25 |
| 2012 | 0.20 | 0.31 | 2.79 | 1.74 | 0.21 | 0.32 | 3.20 | 2.38 | 24.21 | 24.76 | −7.2 | −1.96 | −13.85 | −20.43 |
| 2013 | 0.40 | 0.32 | 2.39 | 1.81 | 0.41 | 0.33 | 3.20 | 2.46 | 49.42 | 27.23 | 10.0 | −0.77 | −56.20 | −24.00 |

Until the remaining N was leached, Lerma Gully presented concentrations comparable to those recorded during the irrigated period, approximately 80 mg $L^{-1}$ (Merchán et al., 2013). The delay in the hydrological behaviour of N depends on predominant hydrological flow paths and the scale of the study zone, and thus is very site specific. For instance, Howden et al. (2011) reported a delay of ≈37 years in the N hydrological response in a highly groundwater-controlled, small scale, headwater basin in the Thames River Basin with an average of 40 m unsaturated zone. In the Lerma Basin, with shallow aquifers and a high degree of connection between the aquifers and the gullies, the delay is minimal as exposed previously.

In the years where storage could be estimated, storage ranged from −7.2 to 10.0 kg $NO_3$—N $ha^{-1}$ $year^{-1}$. Its accumulated value was −0.77 kg $NO_3$—N $ha^{-1}$ $year^{-1}$, which is a significant value when compared to other considered inputs.

The sum of considered inputs accounted for 12% of nitrate exported from the Lerma Gully (27.23 kg $NO_3$—N $ha^{-1}$ $year^{-1}$, Table 4), giving a IN-OUT-ST of −24.00 kg $NO_3$—N $ha^{-1}$ $year^{-1}$. This pattern was recorded every year, indicating that, of the components not taken into account, inputs (e.g., nitrogen fertilizers) were higher than outputs (e.g., exported N in harvest, volatilization or denitrification).

### *4.2.2. Nitrate contamination index*

Under unirrigated conditions (2004–2005), the mass of nitrate exported was, on average, 11.4 kg $NO_3$—N $ha^{-1}$ $year^{-1}$ (Table 5, cumulative value for 2005). In small basins with contrasting land uses, which were monitored for ≈10 years in other areas of the Ebro River Basin, N loads of 5 kg $ha^{-1}$ $year^{-1}$ for forests, 12 kg $ha^{-1}$ $year^{-1}$ for pastures and between 17 and 37 kg $ha^{-1}$ $year^{-1}$ for rainfed agriculture areas have been reported (Casalí et al., 2008, 2010). Similar figures were found for rainfed agriculture in other countries, such as Sweden, where the N leaching in 21 agricultural catchments with similar precipitation ranged from 6 to 32 kg N $ha^{-1}$ $year^{-1}$ (Kyllmar et al., 2014).

With irrigation, the mass of exported nitrate in Lerma Basin increased (2.6 kg $NO_3$—N $ha^{-1}$ $year^{-2}$, Table 5) until reaching average values of 30.8 kg $NO_3$—N $ha^{-1}$ $year^{-1}$, with a maximum of 56.3 kg $NO_3$—N $ha^{-1}$ $year^{-1}$ for the year 2009. This average value was approximately three times the values recorded under unirrigated conditions. Other pressurized irrigated areas exported similar values (between 18 and 49 kg $ha^{-1}$ $year^{-1}$; Cavero et al., 2003; Andrés and Cuchí, 2014; Table 3). However, flood irrigated areas of the Ebro River Basin presented values over 100 kg $NO_3$—N $ha^{-1}$ $year^{-1}$ (García-Garizábal et al., 2012; Barros et al., 2012b).

The comparison with the net N load from cultivated areas does not take into account specific characteristics of the area, such as fertilization needs of different crops. Fertilization needs increased (8.8 kg N $ha^{-1}$ $year^{-2}$, Table 5) mainly during the first three years (2006–2008), as a consequence of the progressive transformation into irrigated land. Fertilization needs reached an average of 251 kg N $ha^{-1}$ $year^{-1}$ across the irrigated period, with a maximum of 302 kg N $ha^{-1}$ $year^{-1}$ for the last year (2013). These values were one of the highest reported in the Ebro Basin irrigated areas (Table 3) due to the high proportion of crops with high fertilization needs (e.g., maize).

Nitrate contamination index gradually increased with the transformation of the basin, from 0.05 in 2006 to a maximum of 0.21 in 2009 (Table 5). The average for the entire irrigated period was 0.12 and its accumulated values were practically constant since 2010. Therefore, the implementation of irrigation supposed an increase of 0.01 $year^{-1}$. However, this NCI trend is lower than that observed for the exported mass of nitrate (2.6 kg $NO_3$—N $ha^{-1}$ $year^{-1}$), as a consequence of the higher increasing trend in FN (8.8 kg N $ha^{-1}$ $year^{-2}$). In other pressurized irrigated areas of the Ebro Basin, similar figures were recorded (Table 3): 0.17, Andrés and Cuchí, 2014; 0.08 to 0.22, Cavero et al., 2003. Flood irrigated areas with lower FN (115–242 kg $NO_3$—N $ha^{-1}$ $year^{-1}$) presented higher NCI (from 0.44 to 0.88, García-Garizábal et al., 2012; Barros et al., 2012b).

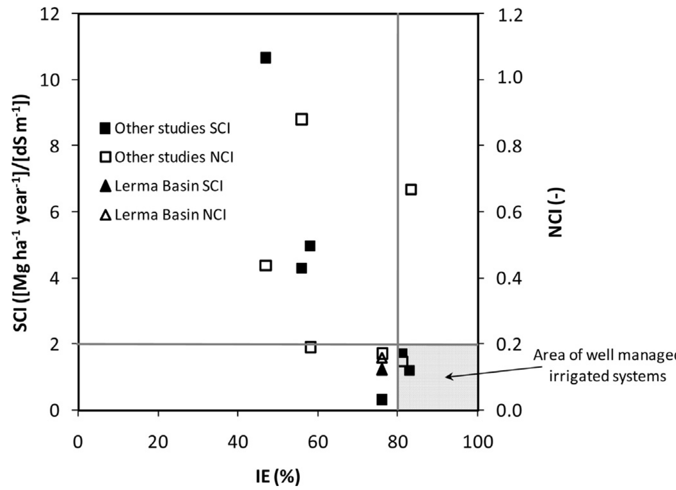


**Fig. 2.** Relation between irrigation efficiency (IE, %), salt contamination index (SCI, [Mg $ha^{-1}$ $year^{-1}$]/[dS $m^{-1}$]) and nitrate contamination index (NCI, –). Data from the studies included are presented in Table 3. More sustainable irrigated areas are within the zone with IE > 80%, ICS < 2 [Mg $ha^{-1}$ $year^{-1}$]/[dS $m^{-1}$] and NCI < 0.2, thresholds indicated by grey lines and shaded area.

Despite the apparent stabilization in NCI, it is necessary to continue monitoring the area, especially considering the expected changes in fertilization management imposed by the designation of the study zone as a vulnerable zone to nitrate pollution (BOA, 2009).

### *4.3. Agro-environmental assessment*

The values obtained for exported mass of salts and nitrate in the Lerma Basin are within those observed in other areas of the world. However, comparison of actual figures is difficult as a consequence of the different natural and agronomic conditions of each area. Causapé (2009) compared SCI and NCI values of different irrigated areas of the Ebro Basin and related them to irrigation management, concluding that SCI values <2.0 [Mg $ha^{-1}$ $year^{-1}$]/[dS $m^{-1}$] and NCI values <0.2 are indicative of an acceptable water use (irrigation efficiency >80%), with a relatively small mass of exported salts and nitrate for the natural and agronomic conditions of the irrigated system.

In this sense, pollution indices once the irrigated area was consolidated ($SCI_{09-13}$ = 1.2 [Mg $ha^{-1}$ $year^{-1}$]/[dS $m^{-1}$] and $NCI_{09-13}$ = 0.16) were below the exposed thresholds, and may be probably reduced with higher irrigation efficiency, which did not reach 80% (Merchán et al., 2015b). Although increases were verified in the masses of salts and nitrate exported during the first years of irrigation, the values obtained were still low considering the salinity and fertilization needs of the area.

Other irrigated areas with more favourable conditions (low salinity and fertilization needs) presented higher pollution indices, which could be related to the management of the irrigated system. Data reported for several irrigation areas (Table 3) show how higher efficiencies in irrigation provided lower pollution indices (Fig. 2), being these key factors to minimize agro-environmental impacts. However, this relationship was not as clearly observed in the case of the nitrate pollution index as in the salt pollution index. Nitrate pollution is affected by several other factors, both natural (e.g., soil texture) and anthropogenic (e.g., fertilization management), and therefore is more variable.

**Table 5**
Annual and cumulative mass of nitrate exported from the irrigable area ($D_N$), fertilization needs (FN) and nitrate contamination index (NCI) during the study years (2004–2013). Cumulative values are presented with annual figures to facilitate comparison.

| Year | $D_N$ | | FN | | NCI | |
|---|---|---|---|---|---|---|
| | Annual | Cumulative | Annual | Cumulative | Annual | Cumulative |
| | kg $NO_3$—N ha$^{-1}$ year$^{-1}$ | | kg N ha$^{-1}$ year$^{-1}$ | | | |
| 2004 | 13.8 | 13.8 | – | – | – | – |
| 2005 | 9.1 | 11.4 | – | – | – | – |
| 2006 | 6.5 | 6.5 | 123 | 123 | 0.05 | 0.05 |
| 2007 | 11.6 | 9.1 | 255 | 189 | 0.05 | 0.05 |
| 2008 | 22.0 | 13.4 | 294 | 224 | 0.07 | 0.06 |
| 2009 | 56.3 | 24.1 | 267 | 235 | 0.21 | 0.10 |
| 2010 | 40.2 | 27.3 | 219 | 232 | 0.18 | 0.12 |
| 2011 | 36.6 | 28.9 | 266 | 237 | 0.14 | 0.12 |
| 2012 | 24.0 | 28.2 | 281 | 244 | 0.09 | 0.12 |
| 2013 | 49.0 | 30.8 | 302 | 251 | 0.16 | 0.12 |
| Trend[a] | ns | ** | ns | ** | ns | * |
| Slope[a] | – | +2.6 | – | +8.8 | – | 0.01 |

ns: Non significant trend ($p > 0.1$).
[a] Significant trend and slope with Mann–Kendall test and Sen's slope (Helsel and Hirsch, 2002).
* $p < 0.05$.
** $p < 0.01$.

As an example of how improved management can reduce pollution indices, García-Garizábal et al. (2012, 2014) demonstrated how small improvements in irrigation management incremented irrigation efficiencies in flood irrigation areas (from 56 to 83%, Table 3) and decreased SCI in 70% (reaching levels of 1.2 [Mg ha$^{-1}$ year$^{-1}$]/[dS m$^{-1}$]) and NCI in 24% (reaching values of 0.67). In a similar way, decreases of approximately 50% in SCI and 56% in NCI were observed after structural and management improvements in another flood irrigation system, reaching levels of SCI and NCI of 5.0 [Mg ha$^{-1}$ year$^{-1}$]/[dS m$^{-1}$] and 0.19, respectively (Barros et al., 2012a, 2012b).

These observations were in line with the general thinking that the adaptation of water application to crop needs is the most effective measure to reduce N leaching (Quemada et al., 2013). Thus, irrigation management appears as a key factor to improve agro-environmental quality. Improvements in irrigation management for the Lerma Basin were deeply discussed in the first part of this study (Merchán et al., 2015b).

Apart from irrigation, there are other management factors affecting the leaching of salts and nitrate from irrigated agriculture. For instance, significant reductions in salts (from 7.5 to 4.7–5.3 Mg ha$^{-1}$ year$^{-1}$) and nitrate (from 99 to 37–70 kg $NO_3$—N ha$^{-1}$ year$^{-1}$) leaching from maize plots were observed in a treatment with two different cover crops (Gabriel et al., 2012a,b).

Moreover, the application of recommended fertilization rates decreased nitrate leaching. However, significant reductions in yield were observed in several trials, which could be avoided with optimal application times (Quemada et al., 2013). In fact, the application of nitrogen fertilizers over a threshold value does not increase yield significantly. Instead, nitrate leaching is increased (e.g., Arbat et al., 2013; He et al., 2012), with even decreasing yields being reported when extreme fertilization rates were applied to some crops (Castellanos et al., 2013).

Other factors, out of the control of farmers, are also relevant to nitrate pollution. In a long term study carried out in similarly managed irrigated soils of the Ebro Basin, no leaching in fine soils and high leaching in coarse soils were reported (Arauzo and Valladolid, 2013). This corroborates observations made in other countries, with lower leaching values for clay-textured soils than for sandy soils (Kyllmar et al., 2014).

Therefore, the optimization of water and fertilization managements along with, when needed, some additional measures (such as the usage of cover crops) should be implemented in order to decrease agro-environmental impacts on water bodies receiving irrigation return flows.

## 5. Conclusions

Dissolution processes were the main source of exported salts in the Lerma Basin, and these processes predominated over precipitation (1.76 Mg ha$^{-1}$ year$^{-1}$) in the salt balance. These exported salts evolved from 1.89 Mg ha$^{-1}$ year$^{-1}$ under unirrigated conditions to 3.51 Mg ha$^{-1}$ year$^{-1}$ under irrigated conditions, increasing with a rate of 0.23 Mg ha$^{-1}$ year$^{-2}$. The salt contamination index was almost three times higher during the irrigated period (0.96 [Mg ha$^{-1}$ year$^{-1}$]/[dS m$^{-1}$]) in comparison with the unirrigated period (0.36 [Mg ha$^{-1}$ year$^{-1}$]/[dS m$^{-1}$]). Due to progressive salt washing of the study zone, lower increases were verified in comparison with the exported salts.

Regarding nitrate exports, increases were verified from 9.1 kg $NO_3$—N ha$^{-1}$ year$^{-1}$ under unirrigated conditions to 30.8 kg $NO_3$—N ha$^{-1}$ year$^{-1}$ during the irrigated period. Nitrate contamination indices have increased with irrigation implementation until reaching a stable value of 0.12, being this value more than double of that obtained in the first years of irrigation.

Despite SCI and NCI are within the threshold considered for the more sustainable irrigated areas (SCI < 2.0 [Mg ha$^{-1}$ year$^{-1}$]/[dS m$^{-1}$] and NCI < 0.2), it would be advisable to improve irrigation management to increase water use and decrease leaching. Additionally, complementary measures such as adjusting fertilization rates to temporal crop necessities or the use of catch crops may prove useful.

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