Agricultural Water Management 186 (2017) 40–50

Contents lists available at ScienceDirect

Agricultural Water Management

journal homepage: www.elsevier.com/locate/agwat

Soil vs. groundwater: The quality dilemma. Managing nitrogen leaching and salinity control under irrigated agriculture in Mediterranean conditions

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ARTICLE INFO

Article history:
Received 14 December 2015
Received in revised form 13 January 2017
Accepted 22 February 2017

Keywords:
Intensive cropping system
Nitrogen fertilisation
Nitrate leaching
Groundwater quality
Saline irrigation water
Irrigation management

ABSTRACT

A 3-year field trial was carried out in southern Italy on an agricultural farm close to the seacoast of Manfredonia Gulf (Apulia Region) where crop irrigation with saline water is standard practice. Seawater intrusion into the groundwater, and the consequent soil salinization represent a serious environmental threat. Each year, two crop cycles were applied, in spring-summer and autumn-winter seasons, respectively. The crop pairing over the three years was tomato and spinach; zucchini and broccoli; pepper and wheat. Cultivation was performed in a field-unit characterised by three adjacent plots. At the centre of each plot, a hydraulically insulated drainage basin was dug (0.70 m depth) to collect the draining water. The crops were irrigated with saline water and leaching treatments were applied with saline or fresh water whenever soil salinity reached a predetermined electrical conductivity threshold. Since soil salinity control might increase nitrate leaching, operational criteria should optimize the trade-off between the application of higher water volumes to reduce soil salinity and lower water volumes to protect groundwater quality from nitrate leaching. The amount of nitrogen leached from the soil root-zone was considerable (on average, 156 kg N ha^{-1} $year^{-1}$) and higher in autumn-winter than spring-summer (72 vs. 28% of the average annual value). In autumn-winter season, nitrogen losses were mainly due to plentiful nitrogen fertilisation and high rainfall. In spring–summer, extra irrigations promoted salt leaching together with nitrogen losses. To manage both irrigation and nitrogen fertilisation a "decoupling" strategy is recommended. This strategy suggests applying leaching preferably at the end of the spring-summer growing season, soon after crop harvesting or at the beginning of the autumn-winter season, before second crop cycle starting. In autumn-winter season, proper nitrogen supplies and timely top-dressing applications, still allow salts to be discharged by rainfalls but prevent nitrogen losses, thus preserving groundwater quality.

1. Introduction

Intensively managed crops usually take advantage of large agrotechnical inputs, such as irrigation water and fertilisers, together with pesticides and herbicides. Serious environmental burdens might result from these inputs so that intensive agriculture practices are recognized as major non-point contamination sources of aquifers (Antonopoulos, 2001; Chowdary et al., 2005; Candela et al., 2008; Moss, 2008; Poch-Massegú et al., 2014).

Nitrogen leaching, namely the downward transport of nitrate-nitrogen (NO_3^--N) out of the root zone by water percolating through the soil profile, is among the most common groundwater form of contamination (Oyarzun et al., 2007). NO_3^--N pollution occurs when nitrogen fertiliser inputs greatly exceed the amount of nitrogen needed by the crops (Asadi et al., 2002; Thompson et al., 2007; Zhu et al., 2005). Nitrate form of nitrogen (NO_3^-) is highly soluble, easily mobile within the soil water solution and poorly adsorbed by the soil particles (Shamrukh et al., 2001). If the rate of NO_3^- uptake by the crop is not great enough, it accumulates into the root zone and is easily leached by irrigation water and rainwater in the deeper soil layers, finally reaching groundwater (Yuan et al., 2000). The greater the nitrogen surplus, the greater the risk of NO_3^-loss from the soil (Gheysari et al., 2009). The amount of leached NO_3^-, however, is directly related to the deep drainage process (Sánchez-Pérez et al., 2003). Several factors could influence this process: the textural characteristics of the vadose zone (Arauzo and Valladolid, 2013), the crop growth features, with specific regard

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http://dx.doi.org/10.1016/j.agwat.2017.02.019

to nitrogen uptake by the root system (Askegaard et al., 2011; Zhou and Butterbach-Bahl, 2014), climatic and weather conditions, nitrogen biochemical transformations in the soil (Schmidtke et al., 2004), and management practices (Ruidisch et al., 2013; Virgílio Cruz et al., 2013; Meisinger et al., 2015). In particular, nitrogen fertilisation (Sánchez-Pérez et al., 2003) and irrigation (Rajput and Patel, 2006; Gheysari et al., 2009; Jia et al., 2014; Wang et al., 2014) are very influential.

NO_3^--N leaching results in the loss of a mineral nutrient source for the plants, widely ranging from 2 kg N to 100 kg N ha $^{-1}$ year $^{-1}$ (Echeverría and Sainz Rozas, 2007) or even more, thus representing a significant economic issue for farmers. Moreover, when groundwater is used as drinking water, high NO_3^- concentrations represent a health concern, because of the associated risk of diseases, like methaemoglobinaemia or blue-baby syndrome in infants, and gastrointestinal cancer in adults (Merrington et al., 2002; Pavoni, 2003; Wolfe and Patz, 2002). To prevent this potential human health hazard, the World Health Organisation (WHO) established a contaminant level that should not exceed 50 mg NO_3^- l^{-1} (11.5 mg NO_3^--N l^{-1}) in drinking water (WHO, 2004). Nevertheless, groundwater NO_3^- concentrations exceeding or approaching this fixed standard have been observed in several countries (Koh et al., 2007; Liu et al., 2005; Min et al., 2002).

In 1991, the European Union (EU) adopted the Nitrates Directive 91/676/CEE, with the aim to protect water quality by preventing nitrate leaching from agricultural activities and promoting the adoption of a code for "Good Agricultural Practices". The Directive includes the definition of Nitrate Vulnerable Zones (NVZs) as areas of land that drain into polluted waters, whereby farmers are required to comply with specific limits of inorganic fertiliser and organic slurry application rates. Although the Nitrates Directive has been implemented in all EU Member States, the problems of nitrate pollution of aquifers still persists throughout Europe (EEA, 2012). Across many European monitoring stations of groundwater quality, 14.4% showed over 50 mg NO_3^- l^{-1} concentrations, and 5.9% showed between 40 and 50 mg NO_3^- l^{-1} from 2008 to 2011 (EC, 2013).

The detrimental consequences of nitrate leaching affecting groundwater quality are much more evident in intensive agricultural areas and, particularly, in arid and semi-arid regions (Jalali, 2005; Ibrikci et al., 2015). In such areas, the use of saline water is a further likely option to meet crop water requirements, with the better quality water primarily allocated to civil uses (Feikema et al., 2010; Verma et al., 2012).

Irrigation with saline water often causes soil salinization (Schoups et al., 2005; Tedeschi and Dell'Aquila, 2005; Wang et al., 2011). Periodic applications of water volumes in excess of crop evapotranspiration are required to leach out soluble salts that have accumulated in the root zone (Ayers and Westcot, 1985). This leaching application results in high volumes of drainage water that are often enriched not only in salts, but also in nutrients (Jalali and Merrikhpour, 2008), including nitrogen (Feng et al., 2005). Therefore, under irrigation conditions where saline water is used, soil salinity management might increase the risk of nitrogen losses from the active soil profile.

In most Mediterranean coastal areas where irrigated agriculture is possible only using brackish water, soil salinization is widely spread. Furthermore, several countries in the Mediterranean basin are affected by non-point source nitrate pollution of aquifers (Zalidis et al., 2002), which frequently occurs in areas of intensive agriculture, such as horticulture, floriculture and citriculture (De Paz and Ramos, 2004; Ramos et al., 2002). Salinization and nitrate leaching are two of the leading threats of the European Mediterranean regions: two sides of the same coin, so to speak. Nevertheless, only a few studies have addressed these two related problems of simultaneous salt and nitrogen leaching (Causapé et al., 2006; Merchán et al., 2015).

Considering the experimental area (Apulian Tavoliere plain – Southern Italy), crop irrigation with saline water is a standard practice, and soil salinization represents a serious environmental threat. To control soil salinity, salt leaching is required but, in this way, groundwater contamination by nitrate becomes very likely. A double bound (in the form of dilemma) is constraining farmer operational choices: less or more irrigation water? On this respect, an optimization strategy is needed.

The here applied experimental trial was already reported in a previous paper (Libutti and Monteleone, 2012), but it was focused on the salt-leaching process only. The present work has turned its attention on nitrate leaching. The specific objectives of this study were to assess the amounts of nitrogen removed from the soil profile through drainage water under saline irrigation conditions, to identify the periods during which nitrogen leaching is most likely to occur, and to suggest agricultural management practices to reduce nitrogen losses and to minimise the risk of nitrate pollution from agricultural sources. At the same time, salt accumulation into the active soil layer should be effectively prevented.

2. Materials and methods

2.1. Site description and experimental layout

A three-year period of continuous field experiments was carried out from spring 2007 to spring 2010 in a Mediterranean area in the north-eastern part of the Apulia Region (southern Italy), at San Giovanni Rotondo in the Foggia district. The experimental field (41°34′N, 15°43′ E; altitude, 15 m a.s.l.) was located on an agricultural farm producing cereals and vegetables. The farm is 15 km from the coast of Manfredonia Gulf (Adriatic Sea), and only a few hundred meters far from the San Severo NVZ, one of the nine Apulian NVZs.

In autumn 2006, a special experimental set-up (Fig. 1) was arranged. This included three adjacent and identical plots of approximately 100 m^2 (6.4 m wide, 15.6 m long). At the centre of each plot, an artificial draining basin was dug, with the removal of the soil to create a trench of approximately 50 m^2 (3.2 m wide, 15.6 m long), with a depth of 0.7 m. The bottom of the trench had a slope gradient of 0.5%. The vertical walls and the bottom of each trench were covered with a plastic sheet to hydraulically isolate the basin and to prevent lateral fluxes and percolation of water. A set of 52-mm-diameter corrugated draining pipes was installed at the bottom of the trench, over the plastic cover, to collect the percolating water. Each plot had six draining pipes that were longitudinally arranged into two groups per trench (three draining pipes per group) and covered with a polypropylene textile. At one end of the trench, the three draining pipes of each group were respectively connected to an unperforated PVC pipe and finally assembled to a connection pipe to channel the percolating water into a 1000-l tank. Each plot had two tanks (i.e., one tank per group of draining pipes) that were buried at the 'downstream' end of the plots. The trenches were then filled with the same soil previously dug out, as near as possible with correct reproduction of the original soil stratification. As a result of this experimental set-up, the natural hydraulic gradient of the soil was disrupted, while a water-saturated zone was formed at the bottom of each draining basin, before the water drained away. This condition mimicked the presence of a shallow water table at a depth of 0.7 m.

During natural or intentional water-leaching processes (i.e., caused by precipitation or irrigation, respectively), the water that percolated along the soil profile was entirely collected in the tanks. The drainage water was then discharged from each tank using an

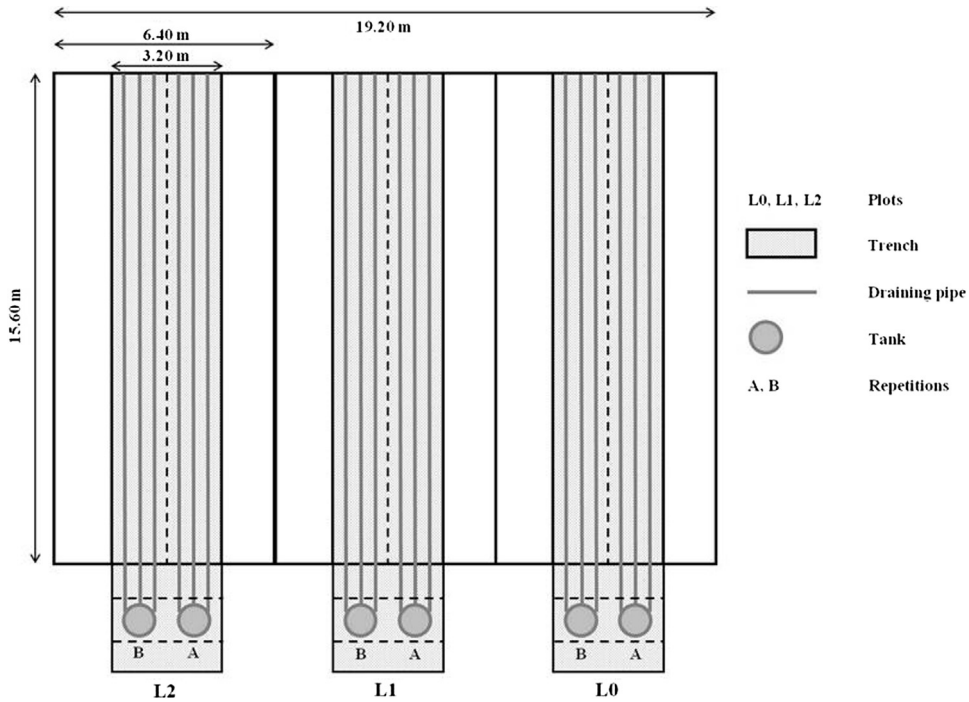

Fig. 1. Layout of the experimental field.

electric pump, with the amounts measured using a water meter placed at the head of the pump. The use of this experimental set-up allowed the water volumes percolating along the 0.70 m-deep soil profile to be recovered and measured, through the whole three-year study period.

2.2. Climate, crop rotation and agronomic conditions

The climate of the study area is Mediterranean, with a long-term (1921–2003) mean annual precipitation of 360 mm. Fig. 2a illustrates the overall time-course of monthly precipitations. Fig. 2b shows the rainfall regime as the superposition of two (and only two) periodic curves obtained through Fourier decomposition analysis. The first curve shows a semi-amplitude oscillation (B_1) of approximately 13 mm, according to a phase shift (J) that corresponds to the 81th day of the year (DOY 81) and coinciding with the spring equinox (21 March). The second curve shows a semi-amplitude oscillation (B_2) of approximately 9 mm, and it can be considered as intra-seasonal modulation, indicating increased rain during spring with respect to summer, and during autumn with respect to winter. The annual alternation of a dry (spring–summer, S1) and wet (autumn–winter, S2) growing season is clearly noticeable. According to this long-term pattern, S1 is the period when precipitations are lower than the annual average value: it starts at the spring equinox (21 March, DOY 81). S2, correspondingly, is the period when precipitations are higher than the annual average value: it starts at the autumn equinox (21 September, DOY 263). On average, 63% of the total annual precipitation is distributed during S2 (227 mm), while the remaining 37% during S1 (133 mm).

The agriculture activity in the study area is highly intensive with the application of a double-cropping system (i.e. two crops per year). Both S1 and S2 included a first phase of crop cultivation and a second phase of fallow, after crop harvesting, with the soil left completely bare. During the whole experimental period, across S1 and S2, the cultivated crops were, respectively (Fig. 3): tomato (*Solanum lycopersicum* L.; cultivar [cv.] 'Perfect peel') and spinach (*Spinacia oleracea* L., cv. 'Falcon') in 2007–2008 (Y1); zucchini (*Cucurbita pepo* L., cv. 'President') and broccoli (*Brassica oleracea* L. var. *italica* Plenk, cv. 'Marathon') in 2008–2009 (Y2); pepper (*Capsicum annuum* L., cv. 'Akron 1′) and wheat (*Triticum durum* L., cv. 'Aureo') in 2009–2010 (Y3).

Crop rotation and cultivar choice followed the options usually applied by farmers in the study area. Cropping operations, including fertilisation, weed and pest control, were carried out according to the standard local farming techniques. The nitrogen fertilisers applied in the experimental field were urea (600 and 100 kg ha^{-1} to spinach and wheat, respectively), ammonium nitrate (200 and 300 kg ha^{-1} to broccoli and wheat, respectively) and ammonium sulphate (100, 400 and 150 kg ha^{-1} to tomato, zucchini and pepper, respectively).

The field experiments were carried out on a loam soil (USDA classification). According to the FAO international standard taxonomic soil classification system "World Reference Base for Soil Resources", the soil of the area belongs to the *haplic calcisol* reference group that corresponds to a *typic calcixerept* in the USDA soil taxonomy. The main physico-chemical properties of the 0.7 m soil surface layer, measured before trial started, are reported in Table 1.

2.3. Experimental treatments

Crop irrigation was regularly performed using saline water and the leaching treatments with saline or fresh water were applied when soil salinity (in terms of electrical conductivity, EC) reached predetermined critical values. A soil electrical conductivity threshold (EC_T) was set for each crop, according to the model proposed by Mass and Hoffman (1977) and assuming a 20% reduction in the estimated crop yield. The corresponding EC_T values were: 6.8 dS m^{-1} for zucchini, 5.0 dS m^{-1} for tomato and broccoli, 4.6 dS m^{-1} for spinach, 3.0 dS m^{-1} for pepper, and finally 11.0 dS m^{-1} for wheat. Three leaching treatments were applied (Fig. 1): no leaching (L0); leaching with saline water (L1); leaching with fresh water (L2).

Irrigation scheduling was performed according to the evapotranspiration criterion with waterings carried out every time the available soil moisture was depleted to the threshold value of 50%. The reference evapotranspiration was calculated daily according to the FAO Penman-Monteith equation (Allen et al., 1998). For

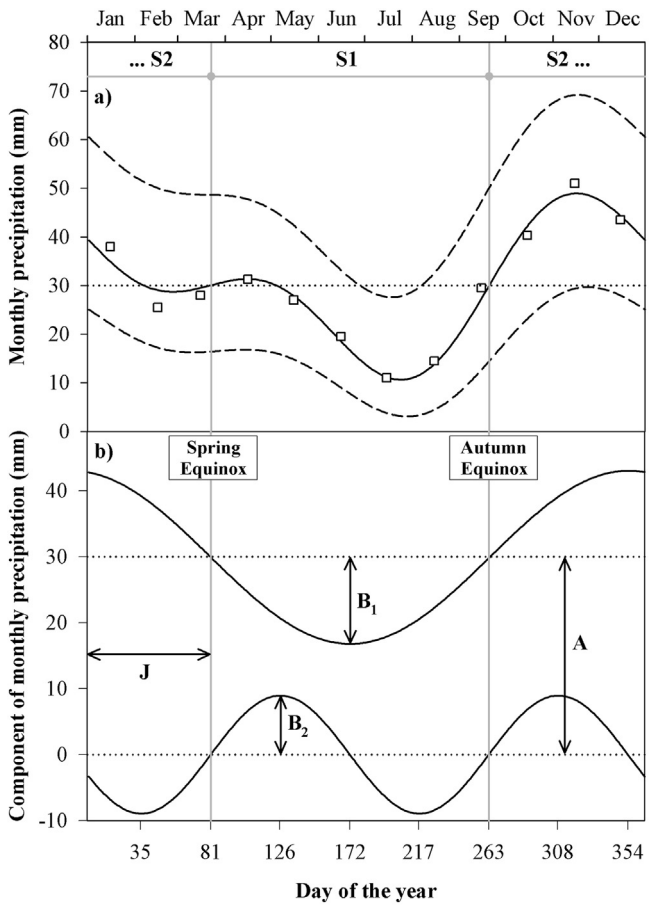

Fig. 2. Monthly precipitation over the long-term period from 1921 to 2003 for the study area; a) the open symbols are the monthly median values (50th percentile) fitted by the solid curve; the two dashed lines indicate the 25th (lower) and 75th (upper) percentiles, respectively. The horizontal dotted line shows the overall monthly mean (A = 30 mm); b) the two curves were obtained through Fourier decomposition analysis and their superposition explains the overall rainfall regime (B_1, B_2 and J being parameters of the curves as explained in the text).

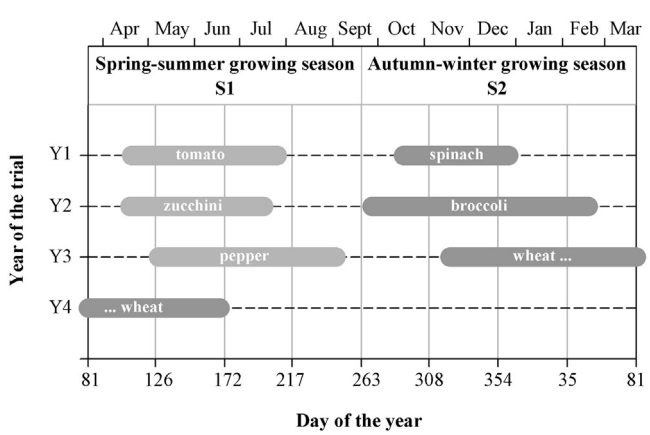

Fig. 3. Time arrangement and length of each cropping cycle with respect to the two growing seasons (S1 and S2) and the three years of the trial (Y1, Y2, Y3). Note that the wheat cropping cycle goes beyond year 3, as it also involved year 4.

each considered crop, the maximum crop evapotranspiration (ET_C) was estimated daily according to the classical 'two-step' procedure, i.e. by multiplying the reference evapotranspiration by the crop coefficients as proposed by the FAO Irrigation and Drainage Paper N. 56 (Allen et al., 1998). To calculate the daily reference evapotranspiration and to daily update the water balance, the considered meteorological variables were: maximum and minimum air temperature (°C) and humidity (%), wind speed (m s^{-1}), and total precipitation (mm). They were logged every 10 min by a weather station placed close to the experimental field; the data were then averaged and recorded every 30 min using a data-logger (Campbell Scientific, USA). Each watering restored the full ET_C losses, with the soil water content taken back to field capacity.

Table 1
Means ± standard errors of the main physico-chemical properties of the 0.7 m soil surface layer.

Soil property	Mean ± standard error
Clay (%)	19.87 ± 0.81
Silt (%)	34.28 ± 0.74
Sand (%)	45.85 ± 0.74
Field capacity (% dw)	29.86 ± 0.31
Wilting point (% dw)	17.41 ± 0.19
Bulk density (Mg m^{-3})	1.31 ± 0.21
Organic matter (%)	1.61 ± 0.07
Total nitrogen (‰)	1.08 ± 0.05
Olsen P_2O_5 (mg kg^{-1})	62.35 ± 0.74
pH	7.58 ± 0.06
EC (dS m^{-1})	2.45 ± 0.17
Na^+ (mg kg^{-1})	200.19 ± 8.38
Ca^{2+} (mg kg^{-1})	178.02 ± 17.38
Mg^{2+} (mg kg^{-1})	25.88 ± 3.16
NO_3^--N (mg kg^{-1})	10.78 ± 0.36
SAR of the satured soil paste extract	3.75 ± 0.12

Table 2
Means ± standard errors of the main chemical parameters measured on saline and fresh water, during the whole experimental period.

Water parameter	Water type	
	Saline water	Fresh water
pH	7.69 ± 0.03	8.04 ± 0.06
EC (dS m^{-1})	5.23 ± 0.03	0.71 ± 0.02
Na^+ (mg l^{-1})	693.79 ± 4.99	84.90 ± 3.87
Ca^{2+} (mg l^{-1})	217.33 ± 1.81	85.54 ± 2.45
Mg^{2+} (mg l^{-1})	170.29 ± 1.35	27.97 ± 1.32
K^+ (mg l^{-1})	133.22 ± 1.95	23.65 ± 1.58
NO_3^--N (mg l^{-1})	40.52 ± 0.52	1.63 ± 0.15
Cl^- (mg l^{-1})	1807.43 ± 2.04	110.98 ± 2.24
SO_4^- (mg l^{-1})	98.37 ± 1.96	74.53± 1.15

As soon as the soil salinity exceeded the predetermined crop EC_T, salt leaching was performed by increasing the irrigation volume according to a leaching fraction (LF) calculated following van Hoorn and van Alphen (1994):

$$LF(\%) = 100 \times EC_W/(2 \times EC_T) \quad (1)$$

where EC_W is the electrical conductivity of the irrigation water.

Saline water was pumped from groundwater. It was characterised by an electrical conductivity that ranged from 4.7 to 5.8 dS m^{-1} during the summer irrigation period, with a mean pH of 7.7. The NO_3^--N concentration in the groundwater varied from 28.5 to 45.6 mg N l^{-1}. Conversely, fresh water was derived from the local water distribution network. This water source was characterised by an electrical conductivity of 0.7 dS m^{-1} and a mean pH of 8.0. The NO_3^--N concentration of this fresh water varied from 1.2 mg N l^{-}1 to 2.5 mg N l^{-1}. Table 2 reports the main chemical parameters measured on saline and fresh water, during the whole experimental period. A drip irrigation system was used, with twelve dripping lines per plot, placed at a distance of 0.5 m. Emitters were installed every 0.4 m along the lines, with a flow rate of 3 l h^{-1}. The amounts of water supplied were measured using a water meter inserted at the head of the irrigation system.

2.4. Sampling and analysis

Each of the three experimental plots was considered as divided into two identical sampling areas (replicates), A and B, respectively (Fig. 1). Soil samples were collected at two-week intervals from the two replicates area of each plot. Each replicate, in turn, consisted of three soil cores taken at 20, 40, and 60 cm soil depths, using a 50-mm-diameter soil auger. Therefore, 18 samples in total were obtained every sampling date. Samples were stored and maintained frozen before determining the NO_3^--N concentration by soil extraction with 2 M KCl, followed by spectrophotometric analysis of the extract (Keeney and Nelson, 1982).

Before trial started, a soil physico-chemical characterization had been carried out. Soil samples were air-dried, crushed, passed through a 2-mm sieve and analysed. The particle-size distribution was determined using the pipette-gravimetric method. The water holding capacities at −0.03 MPa and −1.5 MPa were obtained using a pressure-plate apparatus (Soilmoisture Equipment Corp.), with their difference providing the maximum crop-available water. The bulk density was measured by extracting three intact soil cores at 20, 40, and 60 cm soil depths, with an 'undisturbed' soil-core sampler (Model 0200, Soilmoisture Equipment Corp.). The pH and electrical conductivity were measured on 1:2.5 (w/v) aqueous soil extracts and saturated soil paste extracts respectively. The available phosphorus was determined by the sodium bicarbonate method (Olsen et al., 1954), and the total organic carbon by the Walkley and Black (1934)) acid dichromate digestion technique. The total nitrogen was obtained according to the Kjeldahl method (Bremner 1996).

Saline, fresh and drainage water were sampled over the cropping cycles and fallow periods. Brackish water from the well and fresh water from the local water distribution network were sampled in triplicates every time they were used for crop irrigation and leaching application. Drainage water was sampled in triplicates from the tanks whenever drainage occurred. The water samples were collected in sterile 50-ml polyethylene containers, and transported to the laboratory in refrigerated bags. They were then kept in a refrigerator at +4 °C, and examined within 24 h of collection. The chemical analysis of the brackish, fresh and drainage water included the determination of pH, electrical conductivity (dSm^{-1}) and NO_3^--N content (mg N l^{-1}). The pH was measured using a GLP 22+ pH and Ion Meter (Crison Instruments, Barcelona), and the electrical conductivity using a GLP 31+ EC-Meter (Crison Instruments, Barcelona). For the determination of NO_3^--N concentration, the samples were analysed using an ion chromatograph (Dionex ICS 1100, Dionex Corporation, Sunnyvale, CA, USA).

2.5. Experimental data: source and processing

The present study focused on the water and NO_3^--N fluxes throughout the 0.70 m soil profile. Therefore, the considered data were water volumes into and out of the soil profile and the corresponding NO_3^--N amounts, recorded through each crop cycle and fallow period, across the two growing seasons. Starting from these data, a full set of water and nitrogen soil-balance components were calculated. Table 3 lists them, their measurement units, and their sources.

Considering the *soil water balance*, the seasonal water supply (W_{IN}) was the sum of water amount from precipitation (W_P) and irrigation (W_I). The latter included the water volumes used to irrigate the crop (W_{IW}) and the 'extra' water volumes used to perform salt leaching (W_{IL}). Moreover, the relative drainage (W_R) was obtained as the percentage of drainage water (W_{OUT}) over the seasonal water supply (W_{IN}).

Concerning the *soil nitrogen balance*, the seasonal NO_3^--N supply (N_{IN}) was the sum of NO_3^--N amount from fertiliser (N_F) and irrigation (N_I). The latter included the NO_3^--N amount from the water volumes used to irrigate the crops (N_{IW}) and from the 'extra' water volumes used to perform salt leaching (N_{IL}). The NO_3^--N amounts taken out of the soil (N_{OUT}) were those removed by the drainage water. The variation in the NO_3^--N soil content (ΔN) was calculated as the difference between the total N_{IN} and the leached N_{OUT}. The NO_3^--N soil load (N_{LD}) was calculated as the sum of the initial NO_3^--N soil content (N_0) and the total N_{IN} during the growing season. The NO_3^--N ratio (N_R) was then calculated as the percentage N_{OUT} by drainage over the soil N_{LD}.

The assumption regarding nitrate losses from the soil profile was that the seasonal amount N_{OUT} should be the consequence of both the NO_3^--N concentration in the drainage water (C_N) and the water volume actually drained (W_{OUT}) during the same season, according to the following equation:

$$N_{OUT} = C_N * W_{OUT} \quad (2)$$

In order to identify how much the two factors (C_N and W_{OUT}, respectively) affected the resulting NO_3^--N losses (N_{OUT}), the multiplicative relationship (2) was converted into an additive one (3) through a natural logarithm (Ln) transformation. Therefore:

$$Ln(N_{OUT}) = Ln(C_N) + Ln(W_{OUT}) \quad (3)$$

Table 3
Components of the water and nitrogen balance.

Parameter	Unit	Abbreviation	Source
Water balance			
Precipitation water	$m^3\ ha^{-1}$	W_P	Measured
Irrigation water [a]	$m^3\ ha^{-1}$	W_I	Measured
Seasonal water supply	$m^3\ ha^{-1}$	W_{IN}	$W_P + W_I$
Drainage water	$m^3\ ha^{-1}$	W_{OUT}	Measured
Relative drainage	%	W_R	$100*W_{OUT}/W_{IN}$
Nitrogen balance			
NO_3^--N supplied by fertilisers	kg N ha^{-1}	N_F	Measured
NO_3^--N supplied by irrigation water [b]	kg N ha^{-1}	N_I	Measured
Seasonal NO_3^--N supply	kg N ha^{-1}	N_{IN}	$N_F + N_I$
NO_3^--N removed by drainage water	kg N ha^{-1}	N_{OUT}	Measured
Variation of NO_3^--N soil content	kg N ha^{-1}	ΔN	$N_{IN}–N_{OUT}$
Initial NO_3^--N soil content	kg N ha^{-1}	N_0	Measured
NO_3^--N soil load	kg N ha^{-1}	N_{LD}	$N_0 + N_{IN}$
NO_3^--N ratio	%	N_R	$100*N_{OUT}/N_{LD}$

[a] Sum of irrigation water (W_{IW}) and 'extra' irrigation water (W_{IL}).
[b] Sum of NO_3^--N supplied by irrigation water (N_{IW}) and by 'extra' irrigation water (N_{IL}).

Allometry properly defines the scaling relationship between the two factors and the resultant variable (N_{OUT}) according to the following regression lines:

$$(4a)\mathrm{Ln}(C_N) = a_1 + b_1\mathrm{Ln}(N_{OUT}) \quad (4a)$$

$$\mathrm{Ln}(W_{OUT}) = a_2 + b_2\mathrm{Ln}(N_{OUT}) \quad (4b)$$

In order to comply with Eq. (2) the following conditions are also necessary:

$$a_1 + a_2 = 0 \quad (5a)$$

(5b) $b_1 + b_2 = 1$

Considering the *allometric exponents* b_1 and b_2, respectively, together they account for the total (100%) of the variability observed in N_{OUT}. As much as the b_i value is close to 1, the higher is the degree the corresponding i-factor is affecting the N_{OUT} outcomes. Conversely, the influence of the i-factor on N_{OUT} will be very poor as much as its corresponding b_i value is close to zero.

Seasonal N_{OUT} and ΔN were linearly regressed with respect to the seasonal drainage water (W_{OUT}) while N_R was linearly regressed as a function of the seasonal relative drainage (W_R), according to a covariance analysis (ANCOVA). A full factorial statistical ANCOVA model was applied taking into account the *Season* (*S1*, *S2*), the *Year* (*Y1*, *Y2*, *Y3*) and the *Leaching treatment (L0, L1 and L2)* as experimental factors. The first two linear models (N_{OUT} vs. W_{OUT} and ΔN vs. W_{OUT}) express water and NO_3^--N fluxes in absolute terms and were useful to define how much the experimental factors and their combinations affected NO_3^--N leaching. Differently, the third linear model (N_R vs. W_R) expresses water and NO_3^--N fluxes in relative terms and was intended to define a generalised and comprehensive nitrate leaching pattern that is not influenced by any variable except the relative drainage (taken as the statistical regressor). The ANCOVA analysis was performed using the JMP software package, version 8.1 (SAS Institute Inc., Cary, NC, USA). All of the figures were obtained using the SigmaPlot software (Systat Software, Chicago).

3. Results and discussion

3.1. Water supply and drainage

Precipitations (W_P) observed during S1 and S2 over the three years showed values higher than or at least equal to the median long-term seasonal values (Fig. 4a). Of note, there was an extraordinary level of precipitation during S2 on Y2, when rainfall was equal to 449 mm (the double of the median long-term seasonal value).

With regard to irrigation water (W_{IW}), greater volumes were applied during S1 (dry season) than S2 (wet season) (Fig. 4b). During S1, the irrigation water volumes needed to restore ET_C was 544, 356 and 479 mm, for tomato (Y1), zucchini (Y2) and pepper (Y3), respectively. During S2, the autumn precipitation completely satisfied the ET_C of spinach (Y1), while broccoli (Y2) was irrigated only during the first 40 days after transplanting with 101 mm of water. Wheat (Y3), conversely, was totally rainfed. Considering the amount of 'extra' irrigation water requiredfor salt leaching (W_{IL}), greater volumes were applied in S1 than S2 (Fig. 4b) due to the irrigation with

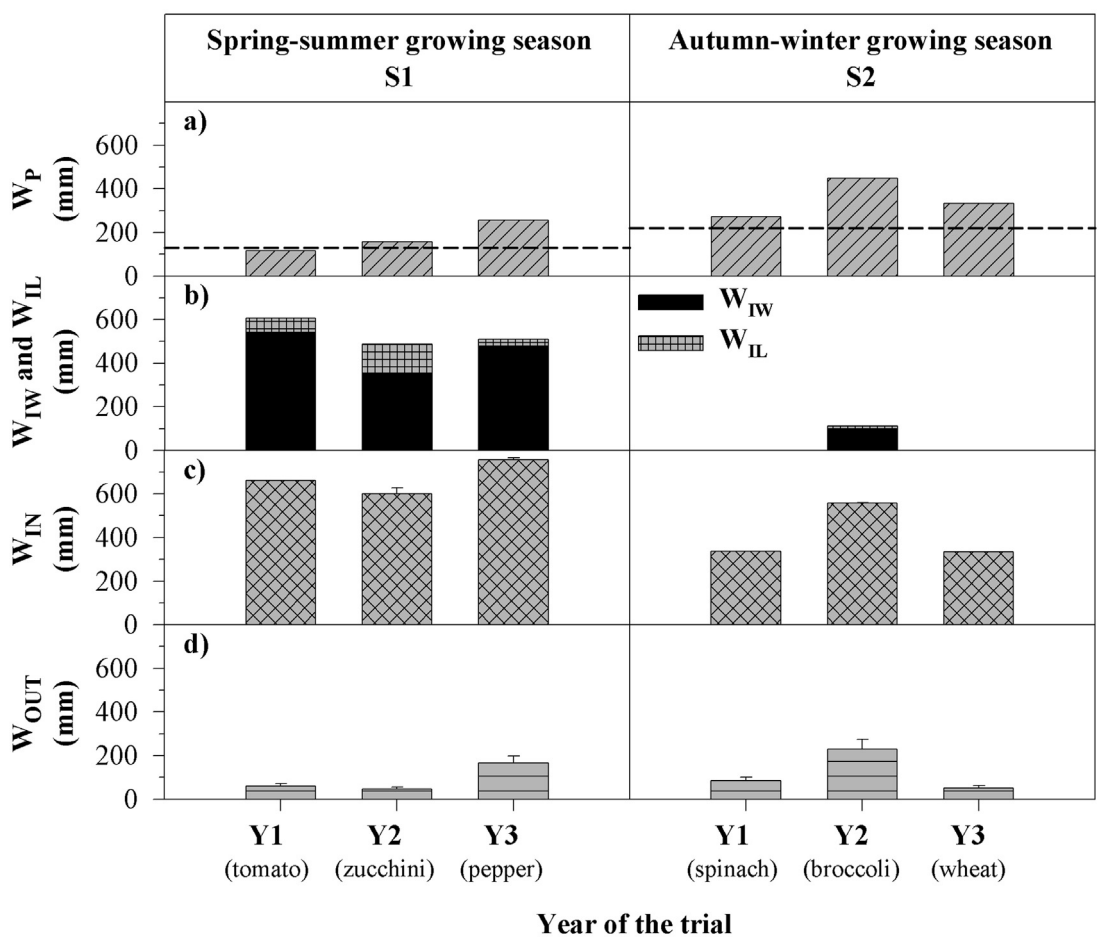

Fig. 4. Mean seasonal amounts of rainwater (a; W_P), irrigation water (b; W_{IW}),'extra' irrigation water (b; W_{IL}), total water supply (c; W_{IN}), and drainage water (d; W_{OUT}), during the two growing seasons (S1, S2) of the three experimental years (Y1, Y2, Y3). In (a), the horizontal dashed lines show the median long-term seasonal precipitation. Vertical bars indicate standard errors of the means.

saline water and the consequent soil salt accumulation. During S1, leaching water volumes equal to 63 and 130 mm were applied in the fallow periods after the tomato (Y1) and zucchini (Y2) harvesting, respectively. Indeed, soil salinity reached the predetermined EC_T values only at the end of the two growing cycles and leaching water application was needed to ensure the establishment of the following autumn-winter crops. Unlike tomato and zucchini, during the pepper cropping cycle (Y3) the predetermined EC_T was reached 40 days after transplanting. Starting from this period and until the end of the crop cycle, a leaching water amount, on average equal to 30.5 mm, was applied. During S2, a leaching water application was needed soon after broccoli transplanting (Y2), since the EC_T was reached, and a mean of 10.5 mm of water was applied. As the autumn precipitation then occurred, there was no need of applying more 'extra' irrigation water.

The total water supply (W_{IN}) was on average higher during S1 (672 ± 46 mm) than S2 (409 ± 74 mm), due to the higher evapotranspiration demand of the spring-summer growing season and the consequent higher irrigation needs of the crops (Fig. 4c).

Differently, drainage water (W_{OUT}) was on average higher in S2 (122 ± 54 mm) than S1 (90 ± 38 mm) (Fig. 4d). S2 accounted for a higher W_{OUT} in Y2 (228 mm). During S1, higher W_{OUT} was observed in Y3 (165 mm). These larger drainage volumes were mainly the result of larger precipitations occurred in these two periods.

3.2. Nitrogen supply and leaching

NO_3^--N amounts supplied by fertilisers (N_F) are reported in Fig. 5a. During S1 similar N_F levels were provided to the crops over the three years (on average, 91 ± 8 kg N ha^{-1}). On the contrary, during S2 the highest N_F was applied to spinach in Y1 (276 kg N ha^{-1}). Spinach crop is indeed greatly responsive to nitrogen fertilisation (Cantliffe, 1992; Magnifico et al., 1992) and it is one of the highest NO_3^- accumulator plants (Maynard et al., 1976).

NO_3^--N supplied by irrigation water (N_{IL}) was higher during S1 than S2 (Fig. 5b). In S1, tomato, zucchini and pepper regularly received irrigation water, thus, a further NO_3^--N supply (on average, 163 ± 7 kg N ha^{-1}) apart fertilisation was applied. During S2, only the broccoli (Y2) was irrigated, thus accounting for further 41 kg N ha^{-1}. Fig. 5b also shows the nitrogen supplied by leaching water applications (N_{IL}). The highest level (69 kg N ha^{-1}) was observed during S1 in Y3, applied to pepper. S2 only accounted for 17 kg N ha^{-1}, observed in Y2, during the broccoli cropping cycle. Comparing Fig. 5a with Fig. 5b, specifically considering the S1 growing season, is quite evident that the "hidden" nitrogen supply provided with irrigation is much higher than the nitrogen supplied with fertilisation (on average 202 ± 9 and 91 ± 8 kg N ha^{-1}, respectively). This condition greatly increases the risk of groundwater pollution by nitrates, because farmers are often not aware of the great nitrogen content of the water used to irrigate crops, thus unintentionally over-fertilising.

The total NO_3^--N supply (N_{IN}) was, on average, higher during S1 (260 ± 8 kg N ha^{-1}) than S2 (168 ± 63 kg N ha^{-1}), due to the higher nitrogen input from irrigation (Fig. 5c).

The average amount of NO_3^--N lost by drainage (N_{OUT}) was higher in S2 (112 ± 54 kg N ha^{-1}) than S1 (44 ± 19 kg N ha^{-1}) over the three years (Fig. 5d). The high NO_3^--N leaching during S2 in Y1, that reached 132 kg N ha^{-1}, was due to the high nitrogen fertilisation applied to spinach. Differently, the very high N_{OUT} value observed during S2 in Y2 was the consequence of heavy and exceptional autumn-winter precipitations that resulted in the loss of 193 kg N ha^{-1} from the soil profile. Considering the autumn-winter

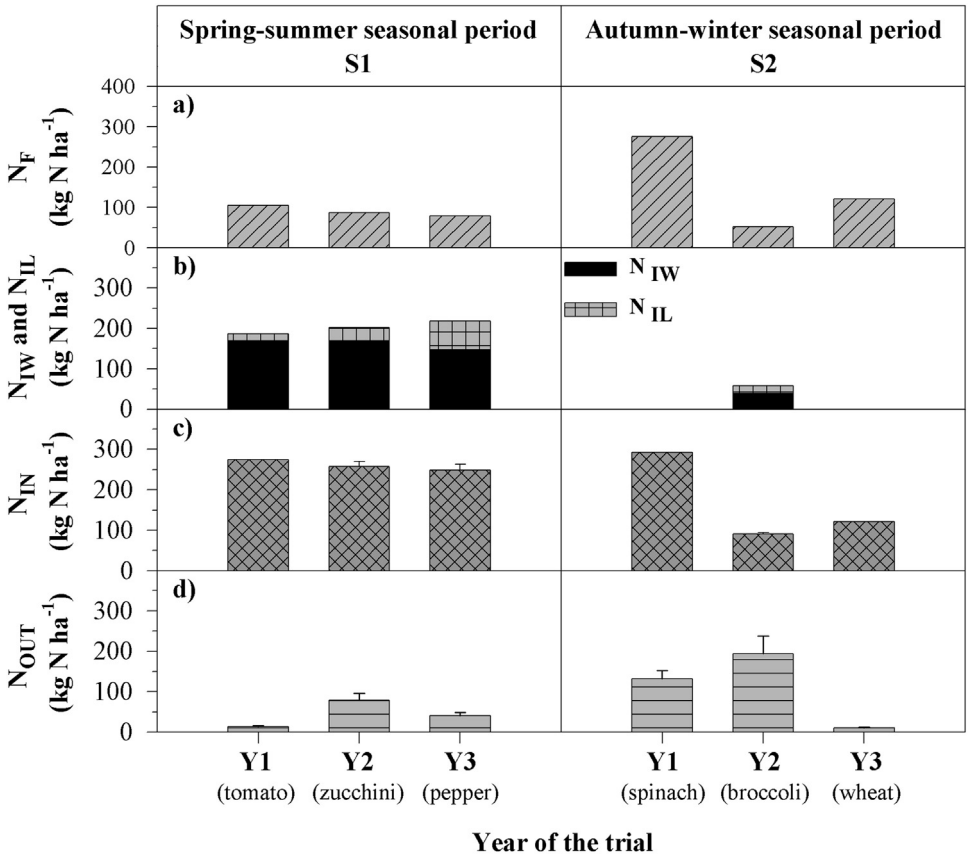

Fig. 5. Mean seasonal amounts of NO_3^--N from fertilisers (a; N_F), irrigation water (b; N_{IW}), 'extra' irrigation water (b, N_{IL}), total NO_3^--N supply (c; N_{IN}), and NO_3^--N removed by drainage (d; N_{OUT}), during the two growing seasons (S1, S2) of the three experimental years (Y1, Y2, Y3). Vertical bars indicate standard errors of the means.

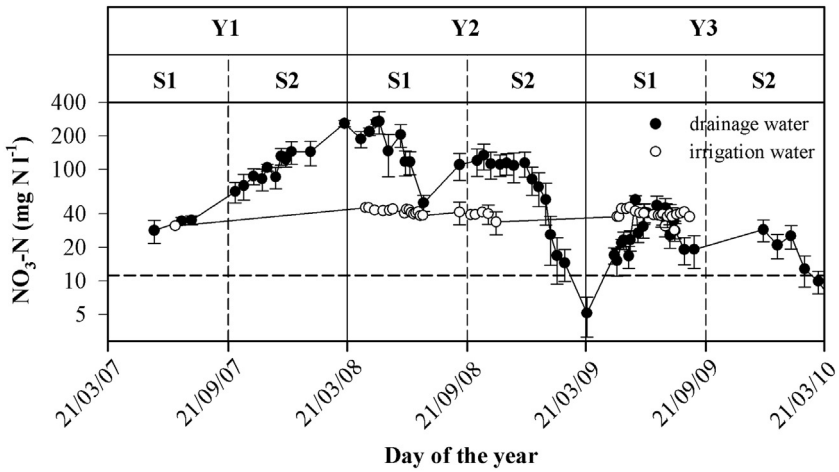

Fig. 6. Time-trend of NO_3^--N concentration (logarithmic scale) in the drainage and irrigation water, during the two growing seasons (S1, S2) of the three experimental years (Y1, Y2, Y3). Vertical bars indicate the standard errors of the means. Horizontal dashed line indicates the limit of NO_3^--N concentration in groundwater set by Nitrate Directive 91/676/CEE (11.3 mg NO_3^--N l^{-1}, as 50 mg NO^{3-} l^{-1}).

seasons (S2), large N-fertilising applications, lesser seasonal ET_C, and abundant rainwater with a correspondingly high drainage rate, were the conditions that facilitated nitrogen leaching. The recorded NO_3^--N losses are roughly in agreement with data obtained in other experimental trials where intensive vegetable production systems, characterised by higher nitrogen application rates, also showed similar nitrogen leaching potential (Zhao et al., 2010). Another study showed that in a wheat-rapeseed rotation under a Mediterranean climate, nitrate leaching increased with nitrogen application (Gallejones et al., 2012). A good data agreement was also noticed with the observation of Arregui and Quemada (2006). They showed that in Mediterranean climate conditions, nitrogen leaching mostly occurred in the autumn-winter season, when the crop nitrogen demand was low, precipitation exceeded crop evapotranspiration, and a considerable drainage took place along the soil profile.

During S1, the higher N_{OUT} (79 kg N ha^{-1}) was observed in Y2, due to the leaching water applied in the fallow period, after zucchini harvesting and before broccoli transplanting. At this time, residual amounts of NO_3^--N not utilized by the plants were easily leached out from the bare soil. Similarly, the leaching water applied during the pepper crop cycle in Y3, originated a NO_3^--N loss of 41 kg N ha^{-1}. Therefore, considering the spring–summer growing season (S1), the most important factor involved in the loss of nitrogen from the soil was the leaching water application. A study conducted in northern Italy confirmed that high nitrate leaching frequently occurred when high nitrogen concentration in the soil is associated to high summer irrigation amounts greatly exceeding the crop transpiration demands thus originating high drainage volumes (Perego et al., 2012).

The NO_3^--N concentration in the irrigation water from the aquifer and the drainage water from the field (Fig. 6) were always above the limits of the Nitrate Directive 91/676/EEC for surface water and groundwater (11.5 mg NO_3^--N l^{-1} equal to 50 mg N l^{-1}). Particularly high were the NO_3^--N levels in drainage water. They varied from a minimum of 5 mg N l^{-1} (at the end of S2 in Y2 and Y3) up to a maximum of 260 mg N l^{-1} (at the beginning of S1 in Y2). The need to control and mitigate the negative impact of nitrogen leaching on groundwater quality is therefore still urgent, despite that Directive 91/676/EEC was issued and implemented long ago.

3.3. Modelling nitrogen leaching

The outcome of the ANCOVA provided a set of linear regressions. Estimates of the linear coefficients (intercepts and slopes, respectively), together with their corresponding statistical significance, are reported in Table 4. The ANCOVA models accounted for the effect of each experimental factor (i.e., *Year*, *Season*, and *Leaching treatment*) as well as for their factorial combination, thus applying a "full factorial" design. The *Leaching treatment* never resulted significant in the ANCOVA and for this reason it was discarded from the regression models. Considering that "*Year*" should be regarded as a "random" factor (due to the unpredictability weather conditions affecting each year), while *Season* is clearly a "fixed" factor, only the *S1 vs. S2* effects are reported in Table 2 and will be discussed.

Column 1 and 2 of Table 2 show the allometric exponents of the regression lines (4a) and (4b), that is Ln (C_N) and Ln (W_{OUT}) both as a function of Ln (N_{OUT}). It is clear that NO_3^--N leaching (N_{OUT}) was fully dependent on the value of W_{OUT} (slope value not statistically different from one), while it was completely insensitive to the value of C_N (slope value not statistically different from zero). This result is an indirect confirmation of the excellent outcomes of the applied ANCOVA models based on the choice of the absolute value of W_{OUT} as a regressor (column 3 and 4 of Table 2) or its relative amount W_R (column 5 of Table 2).

*Column 3 of Table 2 explains the variation of NO3–N soil content (ΔN) as a function of drainage water (WOUT). Fig. 7a represents this relationship (the two upper solid lines). The intercept of the regression line can be interpreted as the total amount of NO3–N added to the soil by fertilisation and irrigation throughout the growing season (NIN). The slope, on the other hand, accounts for the amount of NO3–N discharged from the active soil profile by each unit volume of drainage water (WOUT). Statistically, the model was highly significant (R2 = 0.97**) also showing a good level of precision (root mean square error [RMSE] = 22 kg N ha−1; coefficient of variance [CV] = 15%). A significant influence was showed by both *Year* and *Season*, as well as their interaction, on the value of the intercept. Particularly, the spring-summer season (S1) showed a higher intercept than the autumn-winter season (S2), which highlights the higher total amount of NO3–N applied to the soil in S1 than in S2 (Fig. 7a). These data can be explained by considering that, over the three-year period, fertilisation resulted in a higher nitrogen application (NIN) during S1 than S2 (232 *vs.* 141 kg N ha−1, respectively, according to the intercept values). Moreover, no sig-

Table 4
Statistical results of the ANCOVA models. The parameter estimates (intercepts and slopes of the regression lines) and their corresponding probabilities are reported. The summaries of fits are also showed in the lower part of the Table.

	1		2		3		4		5	
	Ln (C_N) vs. Ln (N_{OUT})		Ln (W_{OUT}) vs. Ln (N_{OUT})		ΔN vs. W_{OUT}		N_{OUT} vs. W_{OUT}		N_R vs. W_R	
Term	Estimate	P	Estimate	P	Estimate	P	Estimate	P	Estimate	P
Intercept	−2.600	**	2.600	**	186.582	**	15.404	n.s.	1.449	n.s.
S1	−0.533	n.s.	0.533	n.s.	45.309	**	−9.900	n.s.	−0.948	n.s.
S2	0.533	n.s.	−0.533	n.s.	−45.309	**	9.900	n.s.	0.948	n.s.
Slope	−0.055	n.s.	1.055	**	−0.055	**	−0.055	**	0.492	**
S1	0.050	n.s.	−0.050	n.s.	0.017	n.s.	−0.001	n.s.	0.111	n.s.
S2	−0.050	n.s.	0.050	n.s.	−0.017	n.s.	0.001	n.s.	−0.111	n.s.
Full factorial model (Y, S)										
R^2	0.950	**	0.931	**	0.969	**	0.959	**	0,956	**
RMSE	0.262		0.262		22.495		16.026		2.327	
CV (%)	9.034		3.944		15.153		23.184		23.969	

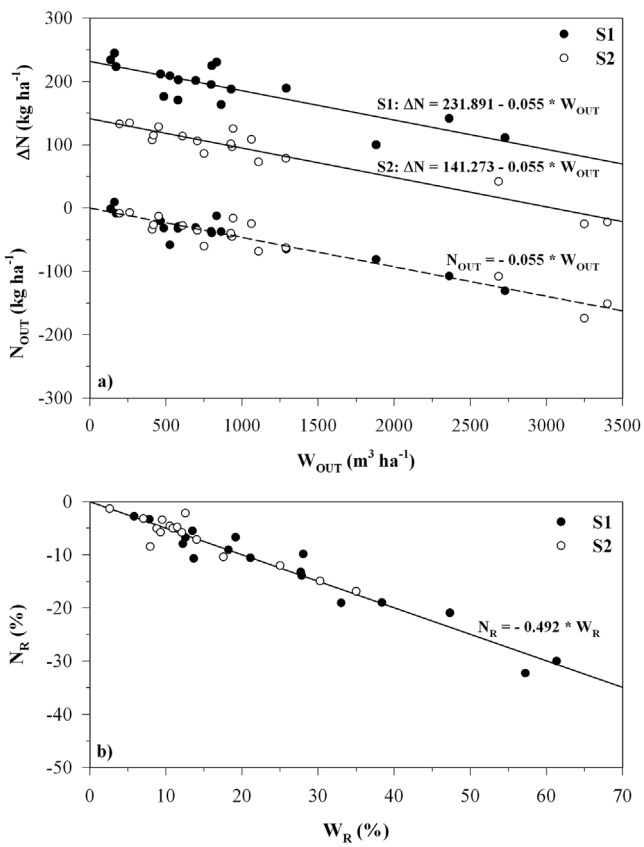

Fig. 7. Linear regressions obtained from the ANCOVA statistical analysis: (a) Variation in the NO_3^--N soil content (ΔN) and NO_3^--N removed by drainage (N_{OUT}) as a function of the drainage water volume (W_{OUT}); (b) NO_3^--N ratio (N_R) as a function of the relative drainage volume (W_R).

nificant effect was played by both *Year* and *Season* on the value of the slope coefficient, therefore an equal slope for the two regression lines should be assumed. According to the slope value and considering a soil cultivated surface of 1 ha, a reduction in soil NO3–N content equal to 5.5 kg N have been observed from the active soil layer per each 100 m3 of drainage water discharged.

Column 4 of Table 2 explains the NO_3^--N loss from the active soil layer (N_{OUT}) as a function of drainage water (W_{OUT}). Fig. 7a represents also this relationship (dashed lower line). With no drainage, NO_3^--N leaching is absent; coherently, the regression line shows a zero intercept value (the intercept value of 15.4 reported in Table 2 is not statistically different from zero) and directly springs from the axis origin. The slope of the regression line explains the NO_3^--N loss rate per unit of drainage volume. Statistically, the model was highly significant ($R^2 = 0.96^{**}$) also showing a good level of precision (RMSE = 16 kg N ha^{-1}; CV = 23%). As expected, *Year* and *Season* showed a negligible statistical influence on the intercept, as well as their interaction ($Y \times S$). The slope of the regression line was significantly affected by the *Year* factor but not by *Season*. Again, the two growing seasons (S1 and S2) showed a common behaviour, this time with respect to NO_3^--N losses. It is worth to note that the slope value of N_{OUT} was not statistically different from the one previously estimated considering ΔN (see column 3 of Table 2), thus explaining that the loss of NO_3^--N by leaching (N_{OUT}) was ruled by the same physical law governing the decrease of NO_3^--N within the active soil layer (ΔN).

NO_3^--N losses are potentially affected by both the initial amount of NO_3^--N in the soil (N_0) and the quantity of NO_3^--N actually supplied by fertilisation and irrigation (N_{IN}). Indeed, the higher is the NO_3^--N availability in the soil, the higher could be the risk of NO_3^--N losses by leaching. The sum of these two NO_3^--N amounts corresponds, as already reported, to the NO_3^--N soil load (N_{LD}). Therefore, N_{OUT} could be effectively expressed in relative terms as a fraction of N_{LD} by considering the NO_3^--N ratio ($N_R = N_{OUT}/N_{LD}$). On this respect, the last model (Column 5 of Table 2 and Fig. 7b) explains the NO_3^--N ratio (N_R) as a function of the relative drainage (W_R). Statistically, the model was highly significant ($R^2 = 0.96^{**}$) also showing a good level of precision (RMSE = 2.3%; CV = 24%). As expected, no significant influence by *Season* and *Year* on either the intercept or the slope coefficients was detected. Therefore, a single regression line passing through the axis origin is featured. The slope value quantitatively expresses the relative NO_3^--N losses from the soil due to leaching: a 10% increase in drainage water corresponded to a 4.9% increase in NO_3^--N loss with respect to the total NO_3^--N amount in the active soil layer.

3.4. The nitrogen-salinity managing according to a "decoupling" strategy

The critical points detected as a result of the trial can be summarized as follow:

1) In the spring-summer growing season (S1), leaching water application could be quite hazardous considering that large drainage volumes and increased nitrogen losses from the soil-plant system were observed. Also considering salt management (Libutti and Monteleone, 2012), salt leaching could be very demanding in terms of water supply and relative inefficient in order to control soil salinization.
2) In the autumn-winter growing season (S2), a larger nitrogen fertiliser supply to the crop, together with the usually

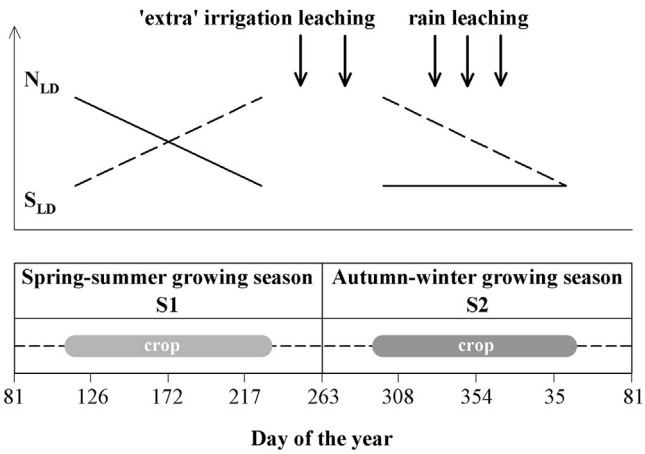

Fig. 8. The proposed "dissociation strategy" as a trade-off mechanism between application of more water for soil salinity control and less water to protect groundwater quality from nitrate leaching.

intense precipitations, resulted in significant nitrogen losses into groundwater due to natural leaching.

In reply to these critical points, a "decoupling" strategy to manage jointly soil salinity (requiring leaching to remove excess salts) and nitrogen fertilisation (not requiring leaching to prevent nitrate loss from the soil) is proposed. Fig. 8 shows the basic principles of this strategy:

1) The application of leaching water should be avoided during the spring-summer crop cycles (S1), simply postponing this operation at the end of the growing season, once the crops are harvested (i.e. the fallow period). The salinity buildup into the soil (S_{LD}), at the end of the irrigation season, can ensure an effective leaching action of salts if compared with too early interventions in the course of the crop cycle. At the same time, the crop progressively exploits the available nitrogen in the soil profile, therefore N_{LD} is largely depleted by the crop uptake and its concentration at the end of the growing season remains relatively low, thus not representing a contamination risk to groundwater. At the same time, the extra water amount required to leach out the salts from the soil profile is limited because, in the absence of the crop, no irrigation water should be accounted for. As a rule, when N_{LD} is high, W_{OUT} should be kept extremely low; correspondingly, also N_R should be kept very low by minimizing W_R.
2) During the autumn-winter growing season (S2), rainfalls play an essential role in the discharge of salts from the soil, but could be very dangerous in terms of nitrate losses. In these conditions, nitrate-fertilising supply should be performed with extreme caution. Following the "code of good agronomic practice", basal dressing should be totally avoided, while top dressing should be progressively applied in tune with the crop growth dynamic. Following this kind of prescription, the N_{LD} level into the soil is kept relatively low, thus taking groundwater nitrate contamination effectively under control. With a relatively low N_{LD}, N_R could be increased in order to sustain a correspondingly high W_R, very effective in terms of salt leaching.

4. Conclusions

Intensive agriculture activity with a close succession of spring-summer and autumn-winter crops, plenty fertilised and irrigated, brings about the serious threat of groundwater contamination by nitrate leaching, together with soil salinization when saline water is used to irrigate. Indeed, one of the outcomes of the present study emphasizes the high vulnerability of the aquifer underlying the experimental site. This pollution risk is further increased when unsuitable practices of nitrogen fertilisation and irrigation are applied. Strategies intending to reduce nitrate contamination of groundwater but ignoring the complex dynamic relationships with other management factors are likely to fail (Letey and Vaughan, 2013). The conventional concept of a surplus of both irrigation and fertilisation, still frequently applied in intensive farming, need to be overcome. The new approach should improve efficiency and reduce cultivation costs but, accordingly, avoid large environmental burdens due to non-point groundwater polluting sources.

The proposed "decoupling" strategy of salt leaching and nitrogen fertilisation management can achieve the objective of reducing nitrate losses into groundwater, while keeping the soil free from excess salt accumulation, by timely applying leaching water volumes only when soil salinity is high but soil nitrate content is low. This specific circumstance matches the inter-cropping fallow period, at the end of the spring-summer growing season (S1), after the crop harvesting; alternatively, at the beginning of the autumn-winter season (S2), before the second annual crop cycle is started.

Such a strategy needs to be accompanied by the correct calibration of the amounts and timing of nitrogen fertilisation. The actual nitrogen needs by the crop and the actual nitrogen content of the soil at the beginning of the crop cycle should lead to the assessment of correct fertilisation plans. Moreover, the amount of nitrogen supplied to the crop by irrigation water, which was revealed to be very remarkable, also needs to be accounted. Finally, a consistent part of the proposed strategy also pertains to the choice of the crops. To avoid the detrimental effects of salt accumulation on crop yield during the spring-summer season, salt-tolerant crops should be preferably cultivated.

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