UNIVERSITÀ DEGLI STUDI DI SASSARI
SCUOLA DI DOTTORATO DI RICERCA
Scienze e Biotecnologie
dei Sistemi Agrari e Forestali
e delle Produzioni Alimentari

Indirizzo Produttività delle Piante Coltivate

Ciclo XXVI

SUSTAINABLE MANAGEMENT OF NITROGEN FERTILIZATION IN IRRIGATED FORAGE SYSTEMS IN NITRATE VULNERABLE ZONES

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Summary

The research was conducted in an intensive dairy farming Nitrate Vulnerable Zone (NVZ) under irrigated Mediterranean conditions focusing on the irrigated maize-Italian ryegrass cropping system. The aims of study were i) quantification of the cropping system nitrogen use efficiency (NUE) of organic fertilizers in relation to the prescriptions imposed by the Nitrate Directive (91/676/CEE), ii) assessment of the nitrate leaching dynamics under the maize-ryegrass cropping system in relation to contrasting fertilization systems, iii) assessment of the relations, at territorial scale, between N surplus and NO_3 concentration in groundwater.

Four fertilization systems were compared at field scale: slurry+mineral, slurry, farmyard manure and mineral at a target rate of 315+130 kg ha-1 N for maize and ryegrass respectively. Fertilization rates of organic fertilizers varied according to their variable N content. NUE of organic fertilizers was not significantly different from that of mineral for maize if N rates were near to target. Organic fertilizers had very low NUE for the ryegrass.

The soil water nitrate concentration was very variable in time with minimum in spring and maximum in autumn-winter. At district area, the average N surplus was closely correlated with the N input.

The study provided sufficient evidence that N pollution of groundwater occurs in autumn-winter and that there are margins to increase the crop N use efficiency by reducing the actual N inputs without significant reduction of crop yield, but with increased costs for the disposal of effluents outside the NVZ.

Introduction

Until the 90s the objectives of the European Community and national policies have had as main objective the increase agricultural competitiveness, through the intensification of cropping systems, the modernization of the sector and the specialized company. This has resulted in a significant increase in the number of animals bred mainly in central and northern Europe, but not only, resulting in the development of the use of double-cropping maize-ryegrass especially in intensive systems of dairy cattle (Moreira, 1994). This cropping system has allowed important management options to increase the production of forage (Helsel and Wedin, 1981; Hughes, 1985) needed for animal feeding. These farming systems are characterized by keeping animals sheltered on barns equipped with concrete gratings, and they produce considerable amounts of slurry and manure widely available to any crop, but especially for maize (Zavattaro et al. 2012 a). In Italy dairy cattle livestock is represented by about 1.8 million animals (ISTAT, 2011) and maize is the most widespread cereal crop after wheat. The cultivated land increased by the end of 1980 until 2004, then, there was a decrease of about 400,000 hectares until 2009 but in recent years it has increased again (ISTAT, 2012), also in relation to the use of silage maize for the production of biogas. In fact the biogas plants increased from 273 to 521 in 2010-2011 (CRPA, 2010). Farmers often do not have enough land on which to dispose farm effluents, therefore this has led to the over-application of organic fertilizers in addition to the mineral fertilizers that are used to achieve high production of silage maize (20-25 t ha^{-1}) and grass hay (7-10 t ha^{-1}) (Giola et al., 2011; Trindate et al., 2008; Trindate et al., 1997). In this type of system, maintaining a high N use efficiency is crucial to obtain high yields and the nutritive value of forage (Neeteson, 2000; Aarts, 2003) but also to limit N losses in environment (Gourley et al., 2012; VandeHaar and St-Pierre, 2006; Fageria and Baligar, 2005) with a consequent impact on water resources underground (Monaghan et al, 2007; Schroder et al 2004; Trindade et al., 1997; Strebel et al., 1989).

According to Giles (2005) nitrogen (N) pollution is the third largest threat to our planet after biodiversity loss and climate change. Specifically, nitrogen losses from agriculture are considered a major cause of pollution of surface and groundwater water (Ten Berge, 2002 Behrendt et al., 2003; Delgado et al., 2008). The efficient N fertilization Nitrogen use efficiency is considered one of the most important management strategies for sustaining or increasing crop yield and quality, and improving nitrogen use efficiency (NUE). The NUE defined as the ratio between the amount of fertilizer N applied and the amount of N removed with the harvest as reported by Brentrup and Palliere, 2010. Management strategies, adequate rate, appropriate source and timings of fertilizer application during crop growth cycle play an important role in reducing costs of production and environmental pollution (Abbasi et al., 2012; Fageria et al., 2006; Ruiz-Diaz and Sawyer, 2008). Such practices not only can increase yield but also contribute to the reduction of the cost of production and environmental pollution. Many studies have focused on split application of fertilizers as a way to reduce N losses and to improve NUE (Liu and Wiatrak, 2011; Sainz Rozas et al., 2004; Garrido-Lestache et al., 2005).

A nitrate directive (ND, 1991/676/CEE) was issued by the European Union in 1991 concerning the prevention of diffuse nitrate pollution of water bodies. The underlying objective of the ND is to reduce water pollution caused by nitrates from agricultural sources and to further prevent such pollution. The implementation mechanisms of the ND involve, among other things, the designation of nitrate-vulnerable zones (NVZs), comprising of areas that drain into polluted or vulnerable waters and which potentially contribute to nitrate pollution. Designation of NVZs is determined in relation to the achievement of a quality state in ground and surface water. A threshold was set for nitrate concentration in ground and surface water (50 mg L^{-1}) to designate NVZs. Barnes et al. (2009) show that NVZ's in Europe account for 38% of the total agricultural area, on which there are various constraints including the regulation of the maximum rate of organic fertilizers to be applied on agricultural land (170 kg N ha^{-1} $year^{-1}$). The ND doesn't distinguish the territorial aspect and the determinants of the problem, but more and more studies reported that the Mediterranean area showed that pedo-climatic contexts and socio-economic development are markedly different and distinct, compared to the northern and central Europe areas. In some studies it can be stated that it is not sufficient only to limit the amount of total N from animal manure because nitrate leaching depends on the speed and the strategy of application of all forms of N (Van Der Meer, 2001) from climate and soil conditions and management practices (Meisinger e Delgado, 2002; Havlin, 2004). Furthermore, the attention only on nitrogen inputs, as well as a uniform threshold, has been criticized because it doesn't take into account the differences in N use or the fate of any N surplus (Schröder et al., 2004). Wallot et al., 2009 highlighted that the lack of objective success for NVZ designation suggests that nitrate pollution control strategies based on input management need to be rethought.

Following a scenario analysis and the definition of the potential environmental impact of dairy operations, Zavattaro et al. (2012 a) concluded that the expansion of a low-input grass-based farming type was the only solution that could significantly decrease N fluxes and achieve water quality targets. Moreover, the same authors stated that to define the scale of survey study more effective for understanding the system is highly complex: the choice of the scaling is critical because environmental processes are often non-linear and hence plot scale processes may not be useful to understand the catchment scale dynamics. When non-linearity and spatial heterogeneity are not taken into account, a loss of information and/or bias in the results may occur (Oenema and Heinen, 1999; Scoones and Toulmin, 1998).

Therefore, scaling requires an understanding of the dynamics of the processes (Haila, 2002). According to Pelosi et al. (2010), a discrepancy in the spatial scale limits the effectiveness of agri-environmental policies and mitigation practices (Kleijn et al, 2004;. Concepción et al., 2008). Unfortunately, even after decades of studies, when mitigation of environmental impact of N use in agriculture is considered, the scaling issue is still unresolved (van Delden et al., 2011). Studies at the basin scale can allow to calculate the nitrogen surplus and quantify the importance of different input and output sources from the system to estimate the N losses (Leach et al., 2003). Since the 70s, the apparent N balance method was applied at different scales in France (Coppenet 1975), but this approach is applicable when sufficient information is available to reconstruct the flows in and out at farm, field or area scales. The method was designed to offer support to the farmer to calculate the optimal dose of nitrogen fertilizer to be distributed and was subsequently used to estimate the pollution potential of the crop (Simon and Le Corre, 1992). The apparent N balance method is currently applied at farm level to assess the potential for nitrate pollution of surface and ground water resources of different cropping or farming systems (Ventura et al., 2008; Simon, 1995). Balance sheets are widely used to study the flow of nutrients in agro-ecosystems, such as agro-environmental indicators for the efficiency of use of nutrients, as well as tools in support of legislative policy (Oborn et al., 2003; Oenema et al., 2003). Several studies demonstrate that only holistic approaches can identify mitigation strategies that consider the whole farm organization and sustainability and address both water and air quality protection (Zavattaro et al., 2012b; Collins et al. 2007). This involves the integration of different skills and levels of knowledge, both scientific and local, in order to contextualize effective actions. Nguyen et al. (2013) showed that the generation of 'hybrid' knowledge (i.e. the integration of local and scientific knowledge), where integration of multiple views and methods in the framework of a carefully designed social learning process, can effectively contribute to better management of the nitrate issue. From the results of this study, we can argue that a prerequisite to trigger a new process towards the sustainable management of the N cycle in intensive dairy farming is the emergence of agreement on concerted actions among stakeholders.

In addition to the complexity of the nitrate issue, European countries, and in particular southern Europe, are exposed to climate change (CC) impacts on geochemical cycles, with increased risk of nutrient leaching, soil salinization, soil loss water erosion (AEA, 2007), and on the hydrologic cycle. This will expose the ecosystems and human communities to substantial changes in the availability and the quality water (as well as the reduction of crop productivity, increase the risk of desertification and neglect of marginal lands) with consequences also on human health (Townsend et al., 2003; Powlson et al., 2008). The CC can increase the risk of nitrate leaching unless preventive measures are taken. Warmer temperatures and higher CO_2 concentrations may lead to higher demands for nitrogen fertilizer (Olesen and Bindi, 2002; Olesen et al., 2007), but extreme weather events, such as heavy storms or droughts, will make fertilizer recommendations less reliable than under stable climate. Besides, a warmer climate may result in increased turnover of soil organic matter especially during winter (Olesen et al., 2004b), which may further increase the risk of nitrate leaching. Cost-efficient strategies to tighten the N cycle are therefore urgently needed. Such strategies should aim to minimize N availability when there is minimal root uptake and risk of percolation.

The areas affected by intensive cropping systems are the areas most prone to this type of vulnerability and risk. Therefore, in order to identify nitrogen fertilization management options applicable and sustainable in irrigated forage systems in Nitrate Vulnerable Zones is necessary.

General hypothesis

The identification of sustainable options for efficient nitrogen use and nitrate pollution mitigation in NVZ require an holistic approach, i.e. considering the agronomic, ecological and social dimensions as intertwined. In this research, new scientific knowledge was developed through a co-researching approach with stakeholders since the design of the experiments. A field scale experiment and a district scale survey were the core business of this research approach aiming to the deconstruction of the nitrate issue and the identification of new options for a more sustainable use of organic fertilizers. The data emerging from field experiments and surveys reported in this thesis were part of a wider process involving stakeholders in a learning process that led to the definition of the treatments to be compared in the field experiment and the integration of scientific and lay knowledge in the interpretation of results (Nguyen et al., 2013). This thesis reports only the results from the field experiment and territorial survey and related interpretations given by the researchers.

Objectives

The aim of the research was to collect data useful for the design of sustainable cropping systems in areas vulnerable to nitrate pollution in the context of intensive irrigated Mediterranean cropping systems, through the quantification of the nitrogen use efficiency of organic fertilizers and taking into account the specifications imposed by Nitrates Directive (91/676/CEE). The potential impact of nitrate leaching of an intensive cropping system is particularly relevant in areas where intensive dairy farming is practiced and excess farm waste is available for N fertilization.

The experimental research was framed in the context of international literature with regard to relations between agricultural activity and environmental quality with specific reference to the processes that control the nitrogen cycle in agro ecosystems. The study has been conducted at two different scales of investigation in a dairy farming context under Mediterranean conditions in which 80% of the agricultural area is occupied by a double cropping of maize and Italian ryegrass:

1. field scale: to evaluate the influence of the fertilization system on crop yield, nitrogen use efficiency of fertilizer and nitrate leaching potential, in order to support decisions on sustainable management strategies to be adopted;

2. territorial scale: to quantify the field gate N surplus as proxy indicator of the leaching potential of nitrate leaching in relation to the actual groundwater pollution.

The thesis was divided into three chapters, that were designed as independent papers that are intended to be submitted to a peer reviewed international journal.

I. The first chapter of the thesis is focused on the assessment of the cropping system N use efficiency of different fertilization systems in intensive forage systems for dairy cattle, in order to test if the ND prescriptions are consistent to the objective of maximizing NUE and minimizing nitrate pollution under Mediterranean conditions.

II. The second chapter is focused on the assessment of the nitrate leaching dynamics under the maize-ryegrass cropping system, in relation to contrasting fertilization systems.

III. The third chapter is focused on the assessment of the field gate Nitrogen surplus as an indirect estimator of the nitrate leaching potential at district scale, determinate through apparent N balance in the context of the ryegrass-maize cropping system.

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Chapter 1
Nitrogen use efficiency of fertilizer in intensive forage systems under Mediterranean conditions

Abstract

Agricultural sustainability relies on Nitrogen Use Efficiency (NUE) where efficient N fertilization, soil and appropriate crop practices are among the most important management strategies for increasing crop yield and water quality either surface or groundwater. The experiment was conducted between June 2009 and September 2011 in a private farm. It was evaluated the effect of different types of N fertilization (cattle slurry, cattle manure, mineral and slurry plus mineral) on the DM yield of herbage and N use efficiency in intensive double-cropping forage systems for dairy cattle in Nitrate Vulnerable Zones (NVZs) in Mediterranean environment. The results of the field scale showed that in maize in the environmental conditions considered, the organic fertilization can achieve levels of NUE comparable to or slightly lower than those of mineral fertilizer. The Italian Ryegrass is exposed to massive leaching if fertilized before seeding and it showed a low efficiency of organic fertilizers.

Keywords: Nitrogen fertilization, Silage maize, Italian ryegrass, intensive dairy farming, Nitrate Vulnerable Zones, dairy cattle effluents.

Introduction

The Europe Union (EU) governmental policies of last 20 years about livestock farming, have contributed to increase the number of animals bred, in particular in dairy farms. The EU is a major player on world markets for most dairy goods and produces the largest single share of the global market. Dairying is one of the most profitable sectors of EU agriculture in particular in Germany, UK, France, Netherlands and North Italy.

However, the high number of livestock has led mainly to the production of large amounts of cattle waste , available to any crop. Farmers often do not have enough land on which to dispose the waste products in the farm, therefore, this has led to the over-application of these organic fertilizers in addition to those of synthesis.

Excessive N feeding in this system can decrease the Nitrogen Use Efficiency (NUE) and this is crucial to maintain high yields, the nutritive value of forage necessary for animal feed (Neeteson, 2000; Aarts, 2003), to reduce production costs and to limit N losses in the environment (Gourley et al., 2012; VandeHaar and St-Pierre, 2006; Fageria and Baligar, 2005) with a consequent impact on underground water resources (Delgado et al., 2008; Monaghan et al, 2007; Schröder et al 2004; Trindade et al., 1997; Strebel et al., 1989). Moreover, the increase of fertilizer and feed costs, the product prices reduction and the implementing regulations environmental pollution reduction have created new pressures to improve nutrient elements use in agricultural production.

In fact, the EU issued the Nitrates Directive (ND) (91/676/CEE), with the objective of reducing water pollution caused or induced by nitrates and to identify the Nitrate Vulnerable Zones (NVZ) (CE, 1991). Barnes et al., 2009 showed that the NVZ in Europe account for 38% of the total agricultural area, on which there are various constraints including the regulation of organic fertilizers (170 kg N ha^{-1} applied to agricultural land each year).

Agronomic management practices on the use of organic fertilizers may transform the target from a waste to a resource product. To improve N efficiency in agriculture, N management strategies that take into consideration the fertilizer improvement under soil and crop management practices, the application of adequate N doses and, the source and timing of fertilizer application during the crops growth cycle play an important role (Abbasi et al., 2012; Nevens and Reheul, 2005; Borin et al., 1997). Split application of N fertilizer in different phenological stages are often recommended to improve NUE, increase yields (Sainz Rozas et al., 2004; Schröder, 1999; Dilz et al., 1982) and reduce the loss of N. However, there are other studies where it's showed that split application of N fertilizer to different crops did not affect their performance and productivity (Garrido-Lestache et al., 2005; Zebarth et al., 2004). Abbasi et al., 2012 reported NUE of maize grown under different N fertilizer sources varied with both N sources and split application. Other studies reported that split application of N fertilizer to different crops did not affect their performance and productivity (Liu and Wiatrak , 2011; Garrido-Lestache et al., 2005). The form or the source of added N plays an important role in regulating N transformations, changing N loss patterns and influencing NUE (Ladha et al., 2005). However the effect of N fertilizer forms or sources on the growth, yield and NUE of maize under field conditions had not been reported extensively. Van Groeniger et al. (2004) showed a lower NUE when using slurry than with mineral fertilizers on silage maize. Ryegrass showed a higher NUE when the N fertilization takes place as coverage (rielaborated data Trindade et al., 2008). Powell et al. (2010) and Aarts et al., 2000 denoted that manure/fertilizer-NUE varies from 16% to 77% and is very site-specific on dairy farms due to high dependence on climate, crops/pasture-soil and management variables. Further information about the environmental effects and management options for agricultural use of organic residues in Mediterranean areas are required.

We hypothesized that under Mediterranean conditions, the NUE of cattle effluents used to fertilize a double cropping system based on grasses with high N uptake all year round can be of the same magnitude of that of mineral fertilizers at similar rates of total distributed nitrogen. We also hypothesised that the characteristics of the effluents (e.g. slurry or farmyard manure) would influence NUE. This may have implications on the implementation of policies to mitigate nitrate leaching in nitrate vulnerable zones.

The aim of the research was to evaluate the effect of different types of N fertilization on the DM yield of herbage and N use efficiency in intensive double-cropping forage systems for dairy cattle in NVZ in Mediterranean environment. This in order to support and help farmers decisions on possible strategies management to adopt. In particular for recycle the different sources of farm available N and hence minimize off-farm mineral N fertilizer. The experiment was designed in a way to assess the influence of fertilization using cattle manure, cattle slurry, cattle slurry plus mineral-N and only mineral-N, on herbage dry matter , N uptake by the crop and NUE.

Materials and methods

Study site and experimental conditions

The case study is located in Arborea (Province of Oristano, Sardinia, Italy; 39° 46’ N; 8° 37’ E), that was improved and reclaimed in the 1930ies. The experiment was conducted between June 2009 and September 2011 in a private farm located inside the NVZ. The area is characterized by Mediterranean climate with the rainy period that occurs during autumn. Total average annual precipitation is 600 mm. The annual average temperature is 17° C, the coldest month is January with average around 10°C, while the warmer is August with about 24° C. The soil is sandy (>90% sand) and according to Soils Classification Systems of United Sates Department of Agriculture (USDA) classification (2006), it is Psammentic Palexeralfs. Soil physical-chemical properties measured at the beginning are reported in Table 1.

Experimental layout and crop management

The experiment was done in a 3 ha field (100 x 300 m^2). The crop rotation was based on the double–cropping forage system with maize silage (*Zea mais* L.) from June to September and Italian ryegrass (*Lolium multiflorum* Lam.) from October to May. This cropping system represents over 80% of the irrigated land in the case study area. Four fertilizer sources were compared at the same doses of N (316 and 130 kg N ha^{-1} for maize and ryegrass respectively), set on the basis of the N fertilization prescriptions for NVZ and on the crop N requirements and in accordance with local farmers:

i) MA= manure (mature cattle farmyard manure applied before sowing with a conventional spreader and followed by rotary tillage);

ii) SL= slurry (cattle slurry applied before sowing with a conventional spreader and followed by rotary tillage);

iii) MI= mineral (mineral fertilizer (ENTEC 26®) applied at the end of tillering for ryegrass);

Table 1 Main soil proprieties at the beginning of the experiment (2009)

	Orizzont			
Characterization	**Ap**	**C**	**2Btg1**	**2Btg2**
Depth, cm	45	77	99	124
Clay, g kg^{-1}	16	27	64	86
Sand, g kg^{-1}	970	960	932	899
Silt, g kg^{-1}	14	13	4	15
Bulk density, g cm^{-3}	1.59	1.39	1.55	1.80
Water holding capacity, %Vol 0 kPa	48.2	42.3	41.4	40.1
Field capacity (%Vol 33kPa)	7.5	3.8	5.9	8.1
Field capacity (%Vol 23kPa)	19.5	15.9	16.8	17.7
Wilting point %Vol 1500 kPa	3.4	1.1	2.2	4.2
Organic Matter, g kg^{-1}	26.8	2.0	1.3	1.3
Organic Carbon, g kg^{-1}	15.5	1.2	0.8	0.8
Tot N, g kg^{-1}	1.4	0.3	0.3	0.3

iv) SM= control (slurry+mineral, i.e. slurry as above but at a target rate of 100 and 70 kg ha^{-1} N for maize and ryegrass respectively, and mineral fertilizer (ENTEC 26®) at a rate of 216 and 60 kg ha^{-1} N applied at maize emergence or the end of ryegrass tillering respectively.

The experimental design was a 4x4 latin square design with a plot size of 12x60 m^2. A mixture of four varieties and hybrids of Italian ryegrass (*Lolium multiflorum* Lam. cv Meritra, Ivan, Littorio and Mowester) was sown during the last ten days of October and organic fertilizer, harrowing and milling were applicated every year. The mineral fertilizer was applied at half February. Auxiliary irrigation was provided to ryegrass when necessary to minimize crop water stress. Ryegrass hay was harvested in mid-May at early earing stage. Hybrid maize (Zea mays L.), cultivar Calcio (FAO 700), was sown during the first ten days of June and it was harvested between the 15th and the 20th of September at the dough stage (target of 33% DM for the whole plant). The sprinkle irrigation started in June until early August. Maize was treated with pre-emergence herbicides (Lumax®). Seedbed was prepared using a rotavator, ripper and harrow, while the harvesting was conducted during mid to late May at the earing stage. The silage maize and Italian-ryegrass were underwent the same crop management for each year and used the same hybrid or mixture respectively. All agro-techniques were applied using business-as-usual machinery and modalities. The rate of organic fertilizers were supplied considering a N content of the organic fertilizers derived from previous analyses (Table 2). However actual rates were monitored by sampling the fertilizers spread on the field and analyzing it for the total and mineral N composition. The time of application of organic fertilizers is constrained by the on-farm available storage volume and NVZ prescriptions that forbid applications from November to 15 February to prevent leaching. A slurry sample per plot was collected from the tank before spreading it in the field. A total of 37 slurry samples were collected, frozen and stored for analysis of ammonium, total N and dry matter. Cattle manure, which is stored for five months before spreading, was collected before application. A total of 39 samples was collected for physical and chemical analyses. The mixture has been incorporated during day by ploughing to minimize the volatilization as ammonia. The N supplied to the plots (Table 2) was calculated from the amount of fertilizer distributed and the nutrient concentration of the fertilizer.

Table 2 N Input from organic and mineral fertilizes (average ± std error) of Maize and Italian Ryegrass from 2009-2012 (MA= Cattle Manure; SL= Cattle Slurry; SM= Cattle Slurry+ Mineral; MI=Mineral).

	Italian Ryegrass	**Maize**
Treatments	**2009-12**	**2009-2011**
MA	152 ±24	378 ±38
SL	305 ±115	475 ±213
SM	217 ±53	369 ±60
MI	130 ±0	316 ±0

Measurements

Dry matter yield, N content and uptake

Crop yield, aboveground biomass at harvest and N removal were measured every year. The aboveground maize biomass production at harvest was assessed by sampling 1 kg fresh weight that was immediately cooled in plastic bags and hence dried in a forced-air oven at 65°C for 72 h to determine DM weight. The aboveground biomass production of Italian ryegrass was measured on sample areas of 1 m^2 per plot by cutting the grass with grass shears. After the sample collection, the entire plots were harvested using farm machinery and these products (fresh maize or ryegrass hay) were weighted immediately using electronic weighing cells positioned in a flat place under the tractor cart were the biomass was accumulated. The crop nutrient uptake was measured on the basis of the N content determined on samples through Kjeldahl method. Both measures were determined annually.

Indicators of N use efficiency

The N use efficiency of crops in the system were estimated as the amount of N removal by the harvested yield (N removal) divided by the total actual amount of fertilizer N supplied in each crop phase added to the estimated atmospheric N deposition:

$$NUE = Nremoval / (Nfertilizer + N\ deposition)$$

N deposition in the area was estimated as 2.4 kg N ha^{-1} $year^{-1}$ (Markaki et al., 2010).
NUE has a value of 1.0 when the amount of supplied N is equal to that removed at harvest.

Statistical analysis

Data were checked for normal distributions (Shapiro-Wilk W-test) and homogeneity of variance (Levene's Test) prior to submit them to the analysis of variance. Data on DM yield, herbage N content and N uptake for both crops for each treatment were treated with analysis of variance (ANOVA) using SAS/STAT® for Statistical Analysis V9 Package for Windows (1999). Tests of significance were made at a 95% confidence level. Regression analyses were performed between the amount of N applied to each crop (independent variable) and crop DM yield and N uptake (dependent variable).

Results

Ryegrass hay production

The DM yield of Italian ryegrass was significantly influenced by the fertilization system. The average annual DM yield of Italian ryegrass ranged from 2.35 (MA) to 7.94 t ha^{-1} (SM). The fertilization systems based just on organic fertilizers (MA and SL) showed an average of production 42% lower than that of SM and MI (Table 3). The analysis of variance showed a significant treatment X year interaction ($P<0.05$). In the first two years, the treatments including mineral fertilizers (SM and MI) have shown significant higher yield than organic treatment (MA and SL). In the third year, the SL treatment produced significantly more than in previous years at the same level of treatments including mineral fertilizers. High DM yields were obtained (c. 8.74 t ha-1 year-1) when 120-130 kg available N ha-1 were applied but rates above 200 available N ha-1 had relatively little effect on DM yield

Silage maize DM production did not show a significant treatment X year interaction (Table 3). The average DM yields ranged from 19.9 to 22.7 t ha^{-1} corresponding, respectively, to MA and SM treatments. The average maize DM yield showed significantly 15% lower production of MA fertilization system than all other treatments (Table 4).

Table 3 Mean Italian ryegrass hay and silage Maize yield (t ha^{-1} DM) as influenced by the N fertilization system (MA= Cattle Manure; SL= Cattle Slurry; SM= Cattle Slurry+ Mineral; MI=Mineral).

	Italian Ryregrass				Maize			
Treatments	**2009-10**	**2010-11**	**2011-12**	**Average**	**2009**	**2010**	**2011**	**Average**
MA	4.62 b	2,35 b	3.32 c	**3.53 c**	22.3	20.8	16.7	**19.9 B**
SL	4.13 b	2.40 b	5.34 b	**3.96 b**	22.9	22.3	21.9	**22.4 A**
SM	7.9 4a	5.07 a	5.54 b	**6.18 b**	23.1	23.5	21.4	**22.7 A**
MI	7.17 a	5.81 a	7.05 a	**6.76 a**	24.1	23.4	19.5	**22.3 A**

Means within a column followed by a different letter are significantly different at P < 0.05 according to Tukey's (HSD) test.

N content of the aboveground biomass N uptake

The total N content of Italian ryegrass hay at harvest was significantly higher (+1.9%) under the MI fertilization system than all other treatments. No significant treatment X year interaction was observed. In agreement with the values reported by Trindade et al., 2008, the treatment with mineral fertilizer increased the content of N in the forage, but we found average N concentration clearly lower in all treatments. The total N content in silage maize at harvest, average across the 3 years, ranged from 9.3 to 10 kg t^{-1} DM with significantly lower N content under MA (Table 4). Also in this case, the average concentration of N found in the treatments turned out to be lower than what was observed by Trindade et al., 2007.

Table 4 Effect of treatments on N concentration in silage maize and forage Italian ryegrass (kg t^{-1}DM) on years for each treatments (MA= Cattle Manure; SL= Cattle Slurry; SM= Cattle Slurry+ Mineral; MI=Mineral).

Treatment	**Italian ryegrass**	**Maize**
	Average 2009-12	**Average 2009-11**
MA	9.2 b	9.3 b
SL	8.9 b	10.0 a
SM	8.6 b	10.0 a
MI	10.8 a	9.6 ab

Means within a column followed by a different letter are significantly different at P < 0,05 according to Tukey's (HSD) test.

The average N removal of Italian ryegrass (Table 5) was about 50 kg N $ha^{-1}yr^{-1}$, below average as reported by Trindade et al., 1997, ranging from a minimum of 20 kg N ha^{-1} to a maximum of 102 kg N ha^{-1} in MA and MI treatments, respectively. A significant treatment x year interaction was observed only in the third year. A significantly higher N removal was observed with SL vs MA treatment in relation to the above mentioned higher DM production. The highest average values of N uptake were observed when the mineral fertilizer (SM or MI) was applied, but in the second year MI removal was significantly higher than SM. in according with the results reported by Perego et al., 2012 and Trindade et al., 2008.

In maize, the treatment x year interaction was not significant. The average N removal of silage maize ranged from 139 kg N ha^{-1} to 258 kg N ha^{-1} and showed significant lower removals (-15%) under MA than the other fertilization systems under comparison. The treatments MI, SM, SL removed an average of 221 kg N ha^{-1}, slightly lower than that found by Trindade et al., 1997.

N use efficiency

The average NUE in maize ranged from a minimum of 0.51 to a maximum of 0.99 in the MA and SL treatments, respectively. In the case of fertilization systems based just on organic materials, the amount of N actually applied was greatly dependent on the variability of the N content of the fertilizer, which despite the homogenization of the sampled materials, varied greatly also between samples collected at the same sampling date. In fact, the average N concentration in slurry was 2.6 % but it showed a variability of 70 %, while in manure was 0.71 % with a variability of 21%. Besides, the average dry matter content was 7.3 and 28.5 %, respectively for slurry and manure, while the average C content was 28.7 % DM for slurry and 29.4% DM for manure as reported in Table 6.

No substantial differences were observed between SL and SM treatments. The response of the crop was dose-dependent. The highest values of NUE were found in the SL treatment for dose about to 316 kg ha^{-1}, significantly higher than that observed in MA treatment in which the NUE values were around 45% (Fig. 1).
The Italian ryegrass showed a low efficiency of fertilizers with average NUE values ranged from 0.17 to 0.56 in the three years, corresponding to MA and MI treatments, respectively (Fig 2).

Table 5. N removal of silage maize and Italian ryegrass hay crop (kg N ha^{-1}) in relation to years and fertilization treatments (MA= Cattle Manure; SL= Cattle Slurry; SM= Cattle Slurry+ Mineral; MI=Mineral).

	Italian Ryregrass				Maize*			
Treatments	**2009-10**	**2010-11**	**2011-12**	**Average**	**2009**	**2010**	**2011**	**Average**
MA	35 ab	20 c	29 b	28 c	219	207	139	188 B
SL	30 b	26 c	47 a	34 c	225	250	190	219 A
SM	64 a	46 b	49 a	53 b	227	258	199	228 A
MI	56 a	102 a	62 a	71 a	337	254	161	217 A

Means within a column followed by a different letter are significantly different at P < 0,05 according to Tukey's (HSD) test.

Integration with mineral fertilizer at rates around 130 kg ha^{-1} in total coverage, resulted as significant in increasing NUE and reached average values of more than 59% higher than the only organic fertilizers. This was associated to the fact that the mineral fertilization was applied at the beginning of "tillering" stage, when the crop, fertilized in pre-sowing with only organic fertilizer, showed obvious signs of stress from N. There were no substantial differences between only SL and only MA.

Table 6. Organic fertilizers characterization: average 2009-2012 (coefficients of variation)

	Cattle Slurry	**Cattle Manure**
N tot (%)	0.26 (70%)	0.71 (21%)
C tot (% DM)	28.7 (63%)	29.42 (40%)
Dry matter (%)	7.3 (76 %)	28.5 (27%)
P (% DM)	3.68 (16%)	0.49 (78%)

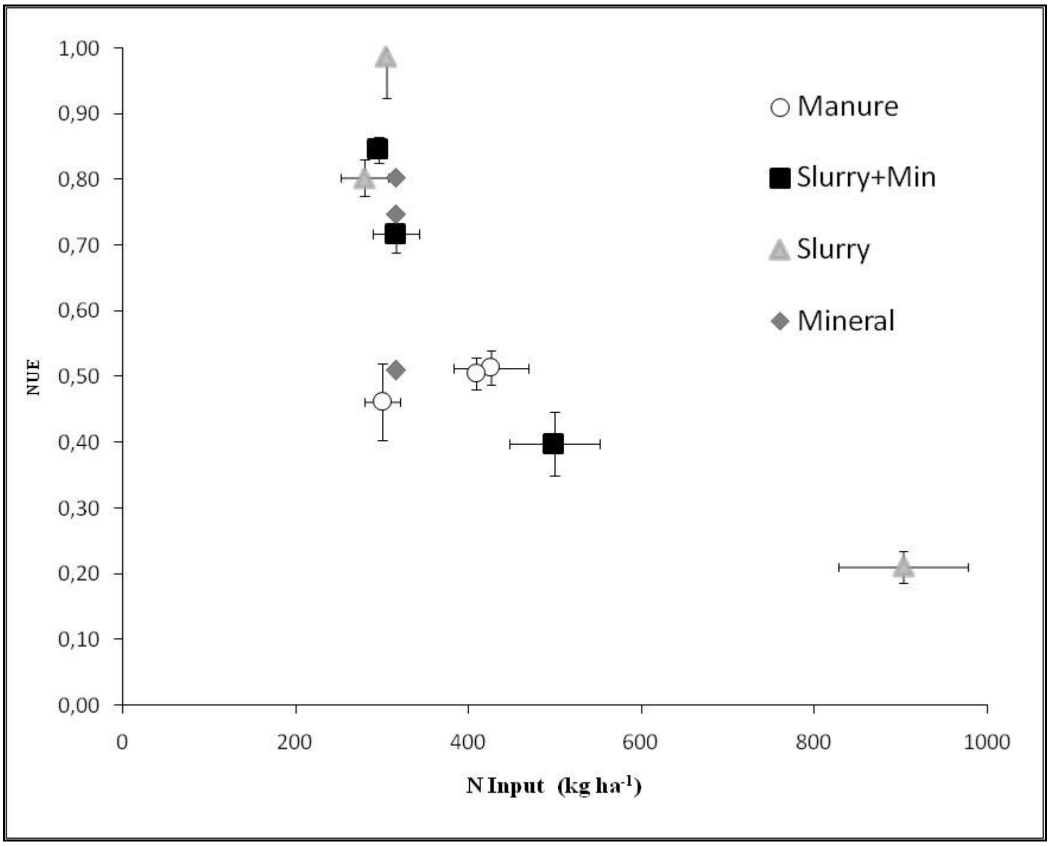

Figure 1. Relation between N input and NUE for organic and mineral fertilizers in Maize

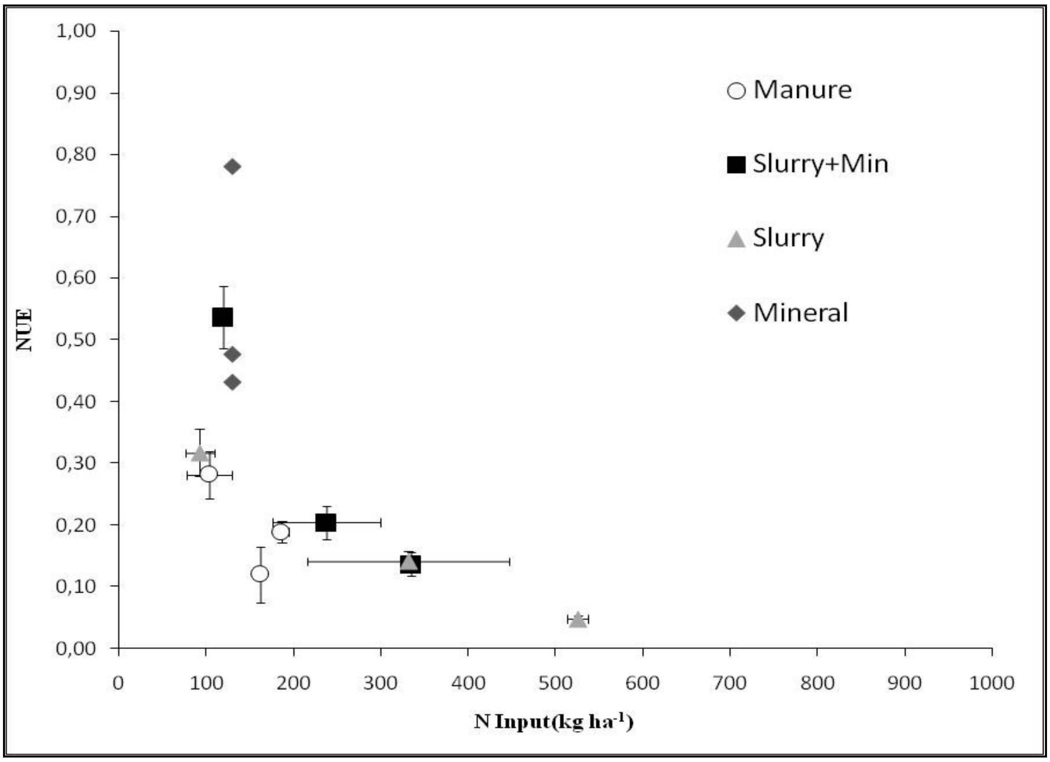

Figure 2 Relation between N input and NUE for organic and miner fertilizers in Italian ryegrass.

Discussion

Ryegrass hay production

The mineral fertilization of ryegrass in spring had a considerable importance on the production. At the end of the winter – beginning of spring, the ryegrass sward treated with just organic fertilizers clearly showed symptoms of N deficiency (light green foliage, yellow leaf tips, reduced growth) when compared with the one where mineral N was supplied. An exception was of the SL treatment in the third year, that was characterized by a relatively dry winter. We think that this is due to synchrony between uptake and supply of N, according to Trindade et al., 2008, in similar intensive system in North west of Portugal. High DM yields were reported also by Zavattaro et al., 2012 , Trindade et al., 2008 and Macoon et al. 2002 for intensive forage cropping systems under Mediterranean climate. In silage maize, Zavattaro et al., 2012, reported that the organic fertilizers performed similarly to Urea in different systems and Nguyen et la., (2013) and Kayser et al.(2011) indicated that the form of N input had no significant effect on the dry matter yields.

N content of the aboveground biomass N uptake

In agreement with the values reported by Trindade et al., 2008, the treatment with mineral fertilizer increased the content of N in the fodder , but we found average N concentration clearly lower in all treatments. The average concentration of N found in the treatments is turned out to be lower than what was observed by Trindade et al., 2008. The average N removal of Italian ryegrass was below average as reported by Trindade et al., 1997. Besides, Perego et al., 2012 and Trindade et al., 2008 had found highest average values of N uptake when the mineral fertilizer was applied. In silage maize, we observed N removal values slightly lower than that found by Trindade et al., 1997.

N use efficiency

The low NUE of organic fertilizers on ryegrass has been associated with leaching in autumn-winter, the distribution of pre-sowing fertilizer and possible losses due to volatilization, according to several studies (e.g. Trindade et al., 2008 reported ammonia

volatilization losses from organic fertilizers rather variable, Fageria and Baligar, 2005; Sommer et al., 1991 showed 10% gaseous-N forms lost; pointed out 30%ammonia lost; Schröder et al., 2007 estimated N losses due to ammonia volatilization for injected/incorporated manure at 5% and for mineral fertilizer at 1%.). As also observed by Carneiro et al., 2012, mainly when the autumn–winter crops sowing is delayed, mineral N should be applied only through top-dressing application(s), and not as usually occurs at crop sowing. It was observed that N applied at tillering was recovered more efficiently than that applied at emergence (Kirda et al., 2001; López-Bellido et al., 2005). Moreover, NUE decreased with increasing doses of organic fertilizer and varied a little among treatments at equal N doses as found also by van Groenigen et al., 2004.
Several studies (Bertora C. et al 2008; Kayser M. et al.., 2011) reported a similar N concentration in cattle slurry and in manure fertilizer in similar systems, but nobody has emphasized the large variability in N content in organic fertilizers that we observed clearly. This makes it very difficult to define a suitable fertilization plan for the crops in pre-sowing.

Conclusions

The NUE in the maize-ryegrass double cropping system under Mediterranean conditions is significantly influenced by the fertilization system and the type and time of distribution of organic and/or mineral fertilizers. The amount of total N distributed with slurry or manure is very uncertain in relation to the very wide range of concentration of N and water content in this effluent independently of the time of distribution. This is a major source of space and time variability of the N distributed with the slurry and make it difficult to design a precise distribution of N at field scale farmyard manure and particularly mineral fertilizers allow to design a more precise rating of the N fertilizer. During the Maize crop phase, the NUE of organic effluents was not significantly lower than mineral fertilizers, as almost no rain falls during summer and hence the water balance can be managed through irrigation, which therefore can allow almost complete control of the water percolation and hence nitrate leaching. The average NUE during the maize phase was 0,65 in SM, 0,67 in SL, 0,69 in MI and 0,47 in MA.

The fertilization system based just on farmyard manure was the least effective in terms of maize yield, but was whereas no differences were observed between mineral fertilizers, slurry or slurry + mineral. In the case of Italian ryegrass the response of organic effluent was the least effective. The pre-seeding distribution of these fertilizers, in October-November, combined with the relatively low ryegrass growth rates as constrained by short day length and low temperature, led to an unbalanced nutrition, with excess of soil mineral N for the early stage of ryegrass establishment and a deep N deficit in early spring, following the winter leaching of the mineral N fraction available soon after the effluent spreading, leading to a significant reduction of the spring ryegrass yield at hay harvest. The distribution of mineral fertilizers at the end of the winter is able to cover this N plant deficit resulting in significantly higher hay yields. The combination of slurry and mineral fertilizers was also effective in terms of crop yield, but the NUE was seriously constrained and inversely proportional to the effective amount of N distributed with slurry, in relation to the total N content of the effluent.

The optimization of the cropping system NUE, in the context of intensive dairy farming under Mediterranean irrigated conditions implies the use of organic effluents just for the maize fertilization, while the ryegrass should only be fertilized with top dressing of mineral fertilizers in late winter. Such practice is expected to minimize nitrate leaching and fertilization costs and hence can provide the scientific evidence supporting a site-specific implementation of the nitrate directive.

The environmental conditions considered in this study, the organic fertilization can achieve levels of NUE comparable or slightly lower than those of mineral fertilizer when applied to maize, while massive nitrate leaching were found under Italian ryegrass in winter because of the natural water surplus associated to high nitrate concentration independently of the fertilization management system. Therefore, we conclude that to mitigate nitrate pollution in these areas it is not sufficient to limit the amount of total N provided by animal manure, as many other factors can contribute to the nitrate leaching.

The dynamics of nitrate concentration It is necessary to reduce organic fertilizers and distribute mineral fertilizers at the beginning of the nitrogen stress of the crop.

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Chapter 2

Fertilization management and nitrate leaching in irrigated forage crops for Mediterranean dairy farming systems.

Abstract

The research aims at evaluating at field scale the relationships between the agronomic management of the animal effluents and the nitrate losses in intensive Mediterranean dairy farming systems. The experiment was conducted between June 2009 and September 2011 and was carried out in a private farm with a highly intensive dairy cattle production, within the Nitrate Vulnerable Zone (ZVN) located in the Central-Western Sardinia, Italy. This area is characterized by a shallow water table and sandy soils. A monthly monitoring of the soil water nitrates concentration was measured by 10 cm diameter disc lysimeters installed at at 60-80 cm depth in relation to four N fertilization systems (SM: slurry + mineral fertilizer; MI: mineral N fertilization only; SL: slurry only; MA: manure only) at a target N rate of 315+130 kg ha^{-1} for maize and Italian ryegrass respectively. Organic fertilizers were spread 2/3 and 1/3 at maize and ryegrass seeding respectively. Actual N rates of organic fertilizers depended on the wide variability of the effluent water and N content. A clear seasonal pattern of nitrate dynamics was observed in the three years. The nitrate concentration dynamics was relatively independent of the treatments, with top leaching occurring in autumn-winter because of the natural water surplus and the low N uptake from the ryegrass seedlings. Soil water nitrate concentrations were significantly influenced by the fertilization system, MI being on average intermediate (151mg L^{-1}) between SL or SM (205 and 169 respectively) and MA (78 mg L^{-1}). Exchanging mineral nitrogen for manure or slurry enabled higher precision in terms of actual rate of N fertilization but did not reduce the high soil water nitrate concentration observed in autumn-winter. Farmyard manure proved to be the most conservative fertilization system.

Key words: Maize silage, Italian Ryegrass, mineral nitrogen, slurry, manure, Nitrate Vulnerable Zones.

Introduction

Nitrogen (N) is a very mobile element in the soil as is easily transported in depth outside the root zone of plants (leaching) in the nitric (NO_3^-) form**.** NO_3^- leaching is an undesirable process which can result in adverse effects for both economic reasons and the degradation of surface and groundwater resources, that may ultimately result in eutrophication and non-drinkable water (OECD, 1982; Smith, 1998; Barton and Colmer, 2006; Wendland et al., 1993, Ten Berge, 2002, Delgado et al., 2008). To improve N use efficiency in agriculture, integrated N management strategies that take into consideration improved fertilization along with soil and crop management practices are necessary. Several efforts to develop improved management strategies for the application of N fertilizers have been made with particular attention to better utilization of mineral N fertilizers in Mediterranean areas. Mediterranean climate is characterized by cool, wet winters and hot, dry summers. In these conditions, NO_3^- losses by leaching are at risk particularly between October and January. Management and control of NO_3^- leaching is difficult because NO_3 losses are often intermittent and linked with seasonal land management, irrigation practices and fertilizer applications and/or irregular events, such as rain (Carpenter et al., 1998; Barton and Colmer, 2006).

Several studies carried out in Spain and Portugal showed that 50-90% of total N losses by leaching were measured during October-February (De Paz et al., 2009; Trindade et al., 1997; Goss et al., 1988). The N applied in autumn and during spring–summer, contributes to the high amounts of available N in the soil at the start of the rainy period (Trindade et al., 2009), conditions that promote significant N leaching (Trindade et al., 1997). In intensive dairy farming systems based on silage maize (Zea mays L.) and forage ryegrass (Lolium multiflorum Lam.) double cropping, organic fertilizers (slurry and manure) are applied regularly prior to sowing of each crop. Mineral fertilizer are often applied to compensate for the insufficient mineral N availability in particular at the end of the winter and to ensure high maize yields. The transport and leaching mechanism of N from manure application is more complex than that of inorganic fertilizers because manure application results in altered soil properties and involves concurrent mineralization, nitrification and denitrification of nitrogen (Geohring et al., 1998). Results from field experiments carried out in a Mediterranean area showed that the NO_3^- leaching potential from the application of slurries could be lower than those from N mineral fertilizer application (Daudén and Quílez, 2004; Trindade et al., 2009). It has also been recognized that NO_3^- leaching can be a common and sometimes serious problem when such effluents are used (Trindade et al., 1997; Daudén et al., 2004).

The Nitrate Directive (ND, 1991/676/EEC) was issued by the European Union in 1991 to prevent diffuse nitrate pollution of water bodies. The underlying objective of the ND is to reduce water pollution caused by nitrates from agricultural sources and to further prevent such pollution. The implementation mechanisms of the ND involve, among other things, the designation of Nitrate Vulnerable Zones (NVZs), which include areas that drain into vulnerable water bodies and which potentially contribute to nitrate pollution. Designation of NVZs is determined in relation to the achievement of a quality state in ground and surface water. A threshold was set for nitrate concentration in ground and surface water (50 mg L^{-1}) to designate NVZs.

The initial hypothesis of the work had concerned two aspects: i) in the Mediterranean intensive grassland systems, the seasonal dynamics of the concentration of nitrates is very wide and ii) the nitrates concentration in water percolation depends on the fertilization systems.

In this paper we quantified and evaluated the dynamics of nitrate leaching and the effects of N fertilization systems under a Maize-ryegrass cropping system in order to identify sustainable management options for enhancing N use by annual crops and mitigate nitrate pollution of water bodies.

Material and methods

Site and crop management

The experiment was conducted between June 2009 and September 2011 in a private farm located inside the NVZ in the dairy district of Arborea, Italy (39°47’ N 8°33’ E, 3 m asl). In this area the most diffuse cropping systems is based on a double cropping of silage maize and Italian ryegrass. The climate is Mediterranean and the mean annual temperature and precipitation are approximately 17°C and 600 mm, respectively. The soil were classified as Psammentic Palexeralfs (USDA, 2006). The soils properties were reported in Table 1 chapter 1 and Lai et al., 2012.

The experimental design was 4x4 latin square with a plot size of 12 x 60 m. We compared four fertilization systems at the same N target rate (316 and 130 kg N ha^{-1} for maize and ryegrass respectively), set on the basis of the N fertilization prescriptions for NVZ which are linked to the crop N requirements and in accordance with local farmers. These were:

i) MA= manure: the crop was fertilized only with mature cattle farmyard manure that was applied before sowing with a conventional spreader and followed by rotary tillage. Some 60% of the total amount was spread to maize at the end of May and 40% to ryegrass in October;

ii) SL= slurry: the crop was fertilized only with cattle slurry that was applied before sowing with a conventional spreader and immediately incorporated in the soil with rotary tillage;

iii) MI= mineral; the crop ewas fertilized only with a mineral fertilizer (ENTEC 26®) applied at the end of tillering for ryegrass and before sowing for maize);

iv) SM= control slurry+mineral: slurry at a target rate of 100 and 70 kg ha^{-1} N for maize and ryegrass respectively additioned with mineral fertilizer (ENTEC 26®) at a rate of 216 and 60 kg ha^{-1} N applied at maize emergence or the end of ryegrass tillering respectively.

All information on management practices in Maize-ryegrass cropping system are reported in chapter one. The rate of N actually supplied with organic fertilizers (Table 2, chapter 1) was calculated ex post from the volume of fertilizer distributed and the actual nutrient concentration of the fertilizer, that was sampled from the field spreader.

The cattle slurry was obtained from concrete slurry-storage pits at the farms and was well agitated prior to surface application using a vacuum-tank spreader. A total of 37 slurry samples were collected and stored frozen for analysis of ammonium, nitrate, total N content and dry matter.

The cattle manure is stored for five months before spreading and was sampled just before application. A total of 39 manure samples were collected for physical and chemical analyses.

The slurry or manure was incorporated into the soil within one day to minimize the N volatilization as ammonia. The time of application of organic fertilizers was constrained by the on-farm available storage volume and NVZ prescriptions that forbid applications in the November-February period to prevent leaching.

Water sampling and analysis

At the beginning of the experiment 36 "sucking" porous cups or lysimeters were installed in the 16 plots below the plowed layer at depths between 50 and 90 cm (average depth: 68 cm). They were placed in each plot in correspondence of the maximum depth of the “Ap” soil horizon in each profile. Special care was taken in repacking the soil from different layers in the original position. The lysimeters were connected to aboveground with plastic pipes at the border of each plot. Near the soil surface a rubber collar (20 cm diameter) was put around the plastic pipe to prevent water from seeping from the surface down through the pipe and to minimize the risk of preferential flow. The lysimeter discs were sampled at monthly intervals. Soil solution samples were obtained by applying a suction of -70 kPa for about 45 minutes using an electric pump. Before collecting the sample to be analyzed, pipes were clean out of the water left from the previous sampling in the connecting pipe and lysimeter. Collected samples were stored at 4°C until analysis. The samples were analyzed for the main chemical characteristics and the nitrate concentration was determined through ion chromatography. In this chapter we only report nitrate concentrations.

Water Percolation

The water balance in Maize-Italian ryegrass cropping system was estimated using the EPIC model (Williams, 1995) that had been previously calibrated and validated with field data. Weather data were collected from the station of Santa Lucia (OR, Italy) which is less than 20 km away of the experimental field.

Statistical analysis

The NO_3^- concentration data were submitted to an ANOVA test according to a 2 factor factorial design considering dates and treatments as fixed factors and dates as a sub-plot of a split plot design in randomized complete blocks (Gomez and Gomez, 1981). A Bartlett test was performed to test homoschedasticity of within sampling dates error variances. In case of eteroschedasticity, sampling dates bearing the highest or lowest variances were progressively excluded by the pooled analysis until the homoschedasticity test was not significant. The data collected in the sampling dates where variances were outliers were analyzed separately. The NO_3^-concentration data were transformed into log values prior to submitting to the parametric statistical analysis to meet the assumption of normality. The mean comparison were performed on transformed values but results were presented as means of the original data. A one-tailed t test was performed to check if at each sampling date treatments significantly exceed the threshold of 50 mg L^{-1} of NO_3^- concentration. Data analyses were performed using Microsoft Excel® and SAS (Sas institute, 1999).

Results

Nitrate concentration

The dynamics of NO_3^-concentration (mg L^{-1}) in the water percolation during the maize-Italian ryegrass rotation from 2009 to 2012 showed a strong seasonal pattern that was conserved across years. NO_3^- concentration dynamics was relatively independent of the treatments and seasonal averages exceeded 50 mg L^{-1} in most cases. The NO_3^-concentrations dynamic showed typically two maximum values: one in the summer, during the maize phase, and one in the winter, during the ryegrass phase (Figure, 1 a, b, c). Minimum values were typically observed at maize harvest in September and in late spring. The highest NO_3^- concentrations were observed in the autumn and winter period (from November to January) in all years and treatments. In all treatments, the NO_3^-concentrations increased progressively from the early Autumn to January and then decreased to reach a minimum in April and May, with the exception of spring 2012, when the observed NO_3^-concentrations remained almost steady until the end of April in all treatments.

The annual average concentration of nitrates from June 2009 to May 2012 fluctuated from 6 ± 3 to 593±104 mg L^{-1} in SL, from 0,8 ±0,3 to 586 ± 122 mg L^{-1} in MA, from 13 ± 3 to 460 ± 54 mg L^{-1} in SM and from 2,6 ± 0,4 to 390 ± 25 mg L^{-1} in MI (Figure 2). In the 3-years experimental period, the average (n = 4 reps x 33dates) NO_3^- concentration in SL, MA, SM and MI treatments was 202 ± 11,3 mg L^{-1}, 78 ±4,9 mg L^{-1}, 164 ±9,4 mg L^{-1} and 146 ± 10,4 mg L^{-1}, respectively.

The statistical analysis showed a high variability of NO_3^-concentration in soil water on all treatments. Heteroschedasticity was also observed between the error variances of each sampling date. For this reason, four dates corresponding to the two highest and the two lowest error variances were excluded from the pooled analysis between sampling dates, while the remaining 29 dates had homogeneous variances (Table 1). The four outlier dates were highlighted in Table 1 and corresponded to the sampling dates with highest (Feb and Mar 2012) and lowest (Nov Dic 2011) variances.

Considering the above mentioned 29 dates, the date x treatment interaction was not significant while highly significant effects between dates and between treatments were found on NO_3^-concentration in soil water. The overall mean soil water NO_3^- concentrations of SM and SL were significantly higher than MI, that was significantly higher than MA.

In the remaining four dates only one (Nov 2010) showed significant differences between treatments: the NO_3^- concentration of SL and MI was significantly higher than SM, that was significantly higher than MA. On this sampling date, mineral fertilizers for the winter crop were not yet distributed, as by experimental design.

Due to very high variability of the NO_3^- concentration between replicates of the same treatment, high mean values sometimes were not significantly higher than the 50 mg L^{-1} threshold. In the three years the mean NO_3^- concentration significantly eceeded the 50 mg L^{-1} in 12, 10, 5 and 4 out of the 33 sampling dates in SL, SM, MA and MI respectively. Of these 7/12, 7/10, 4/5 and 4/4 respectively in SL, SM, MA and MI occurred in the November-February interval and the remaining occurred in the spring 2012.
The MA treatment showed the lowest average nitrogen concentration of all other treatments. In 2009 and 2010 MA showed an average annual NO_3^-concentration of about 40 mg L^{-1}, but in 2012 the dynamics was similar to that of the other treatments.

Water percolation

Cropping systems showed an average yearly percolation of 308 mm , 362 mm, 170 mm and 123 mm in 2009, 2010, 2011 and 2012 respectively. In all years, percolation events mainly occurred during October to January (Figure 1, a, b, c) with a strong influence of precipitation (y) on percolation (x) being observed, since the linear regression between these two variables was: = 1,256x + 27,55 (r^2 =0,78). In three years experimental period, the average water percolation in autumn was about 190 mm and it has represented the 78% of total annual percolation. The month that showed the highest percolation was November, corresponding also to the rainiest month, and the average value was 77 mm. While in summer the percolation was near the zero, in fact, the irrigation water supply rarely exceeded crop transpiration demands. The rainfall in year 2010 was considerably higher than the long term average recorded for the region and this was reflected in the calculated drainage volumes.

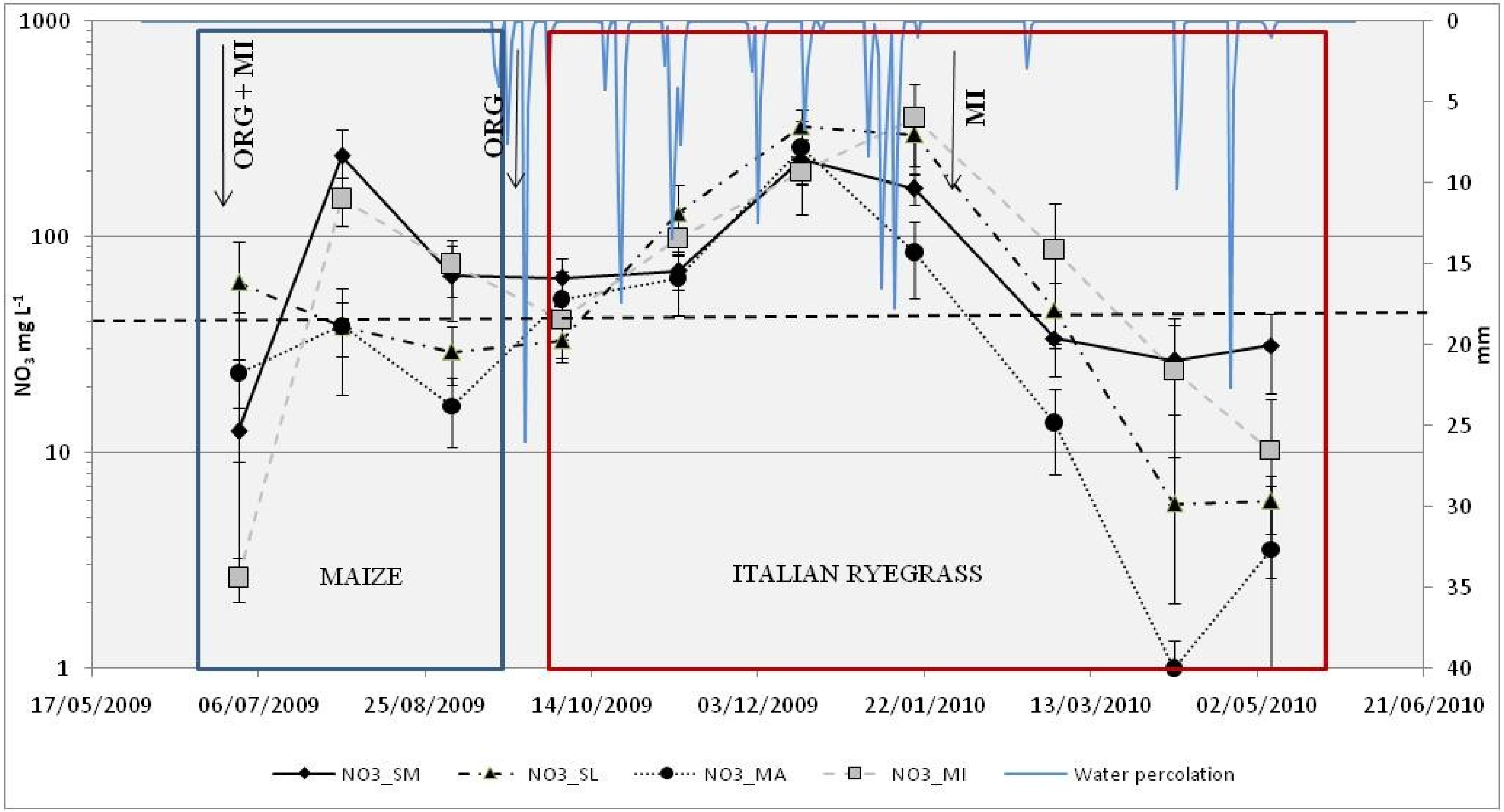

Fig 1, a. Dynamics of the NO_3^- concentration (mg L^{-1}) in the percolation water during the silage maize – Italian ryegrass rotation (2009–2010). Error bars indicate standard errors. Vertical arrows indicate the dates of fertilizers distribution. The horizontal dotted line indicates the legal threshold of 50 mg L^{-1} for drinkable water and the vertical dotted line indicates the end of cropping cycle.

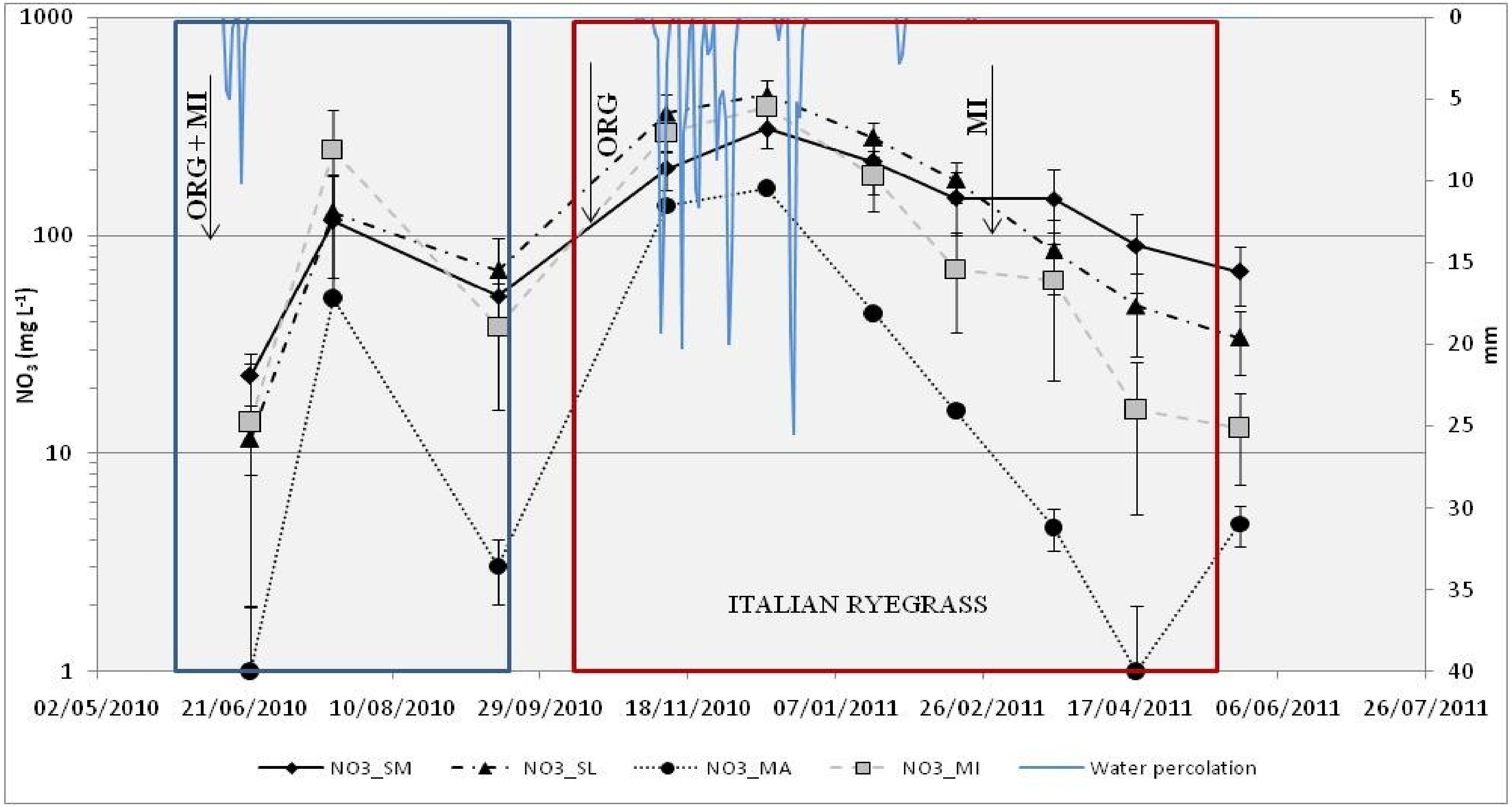

Fig 1, b. Dynamics of the NO_3^- concentration (mg L^{-1}) in the percolation water during the silage maize – Italian ryegrass rotation (2010–2011). Error bars indicate standard errors. Vertical arrows indicate the dates of fertilizers distribution. The horizontal dotted line indicates the legal threshold of 50 mg L^{-1} for drinkable water and the vertical dotted line indicates the end of cropping cycle.

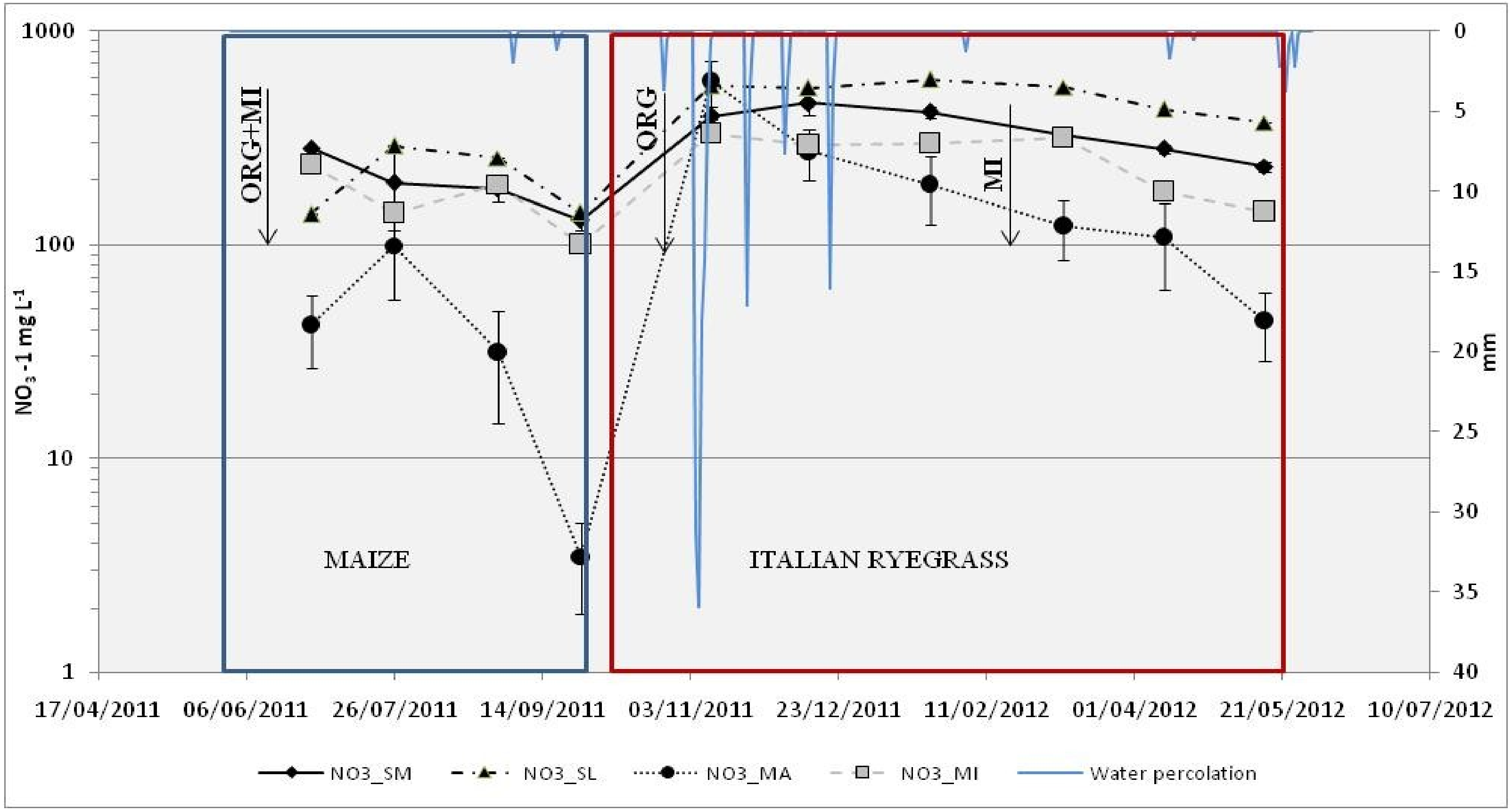

Fig 1, c. Dynamics of the NO_3^- concentration (mg L^{-1}) in the percolation water during the silage maize – Italian ryegrass rotation (2011–2012). Error bars indicate standard errors. Vertical arrows indicate the dates of fertilizers distribution. The horizontal dotted line indicates the legal threshold of 50 mg L^{-1} for drinkable water and the vertical dotted line indicates the end of cropping cycle.

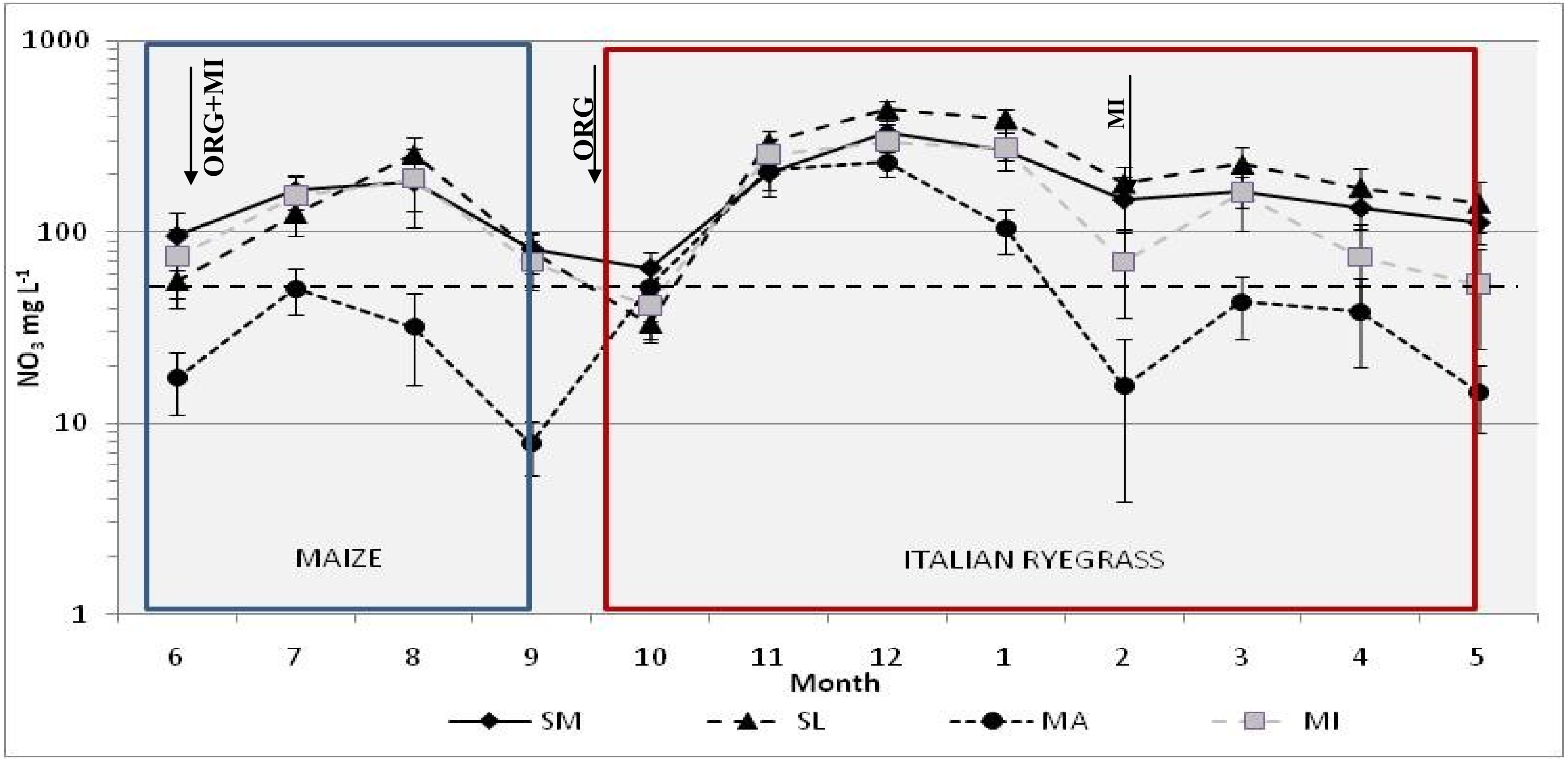

Figura 2. Dynamics of the average NO_3^- concentration (mg L^{-1}) in the percolation water during the silage maize – Italian ryegrass rotation in three years (2009–2012). Error bars indicate standard errors. Vertical arrows indicate the dates of fertilizers distribution. The horizontal dotted line indicates the legal threshold of 50 mg L^{-1} for drinkable water and the vertical dotted line indicates the end of cropping cycle.

Table 1 Average NO_3^- concentration (mg L-1) in sampling date for each treatments (SM= Slurry +Mineral, SL= Slurry, MA= Manure, MI=Mineral) and the statistical analysis.

Date	SM	SL	MA	MI	error MS*	d.f.	P*	CV%*
30_06_2009	13	61	23	3	1.91	6	0.60	63
09_07_2009	108	43	10	45	1.85	4	0.69	53
31_07_2009	237	38	39	150	2.07	9	0.20	37
02_09_2009	66	29	16	75	1.07	9	0.13	30
05_10_2009	65	33	51	41	1.47	9	0.59	36
04_11_2009	80	111	36	98	2.47	9	0.16	44
09_11_2009	59	147	93	n.a.	0.59	2	0.73	18
16_12_2009	229	325	261	199	2.24	9	0.50	31
19_01_2010	167	297	85	357	1.79	8	0.29	28
02_03_2010	34	46	14	87	1.47	8	0.73	38
07_04_2010	27	6	1	24	2.80	8	0.46	114
06_05_2010	31	6	4	10	1.72	9	0.14	89

Date	SM	SL	MA	MI	error MS*	d.f.	P*	CV%*
23_06_2010	22	6	1	4	1.62	9	0.16	63
07_06_2010	23	17	1	24	1.80	9	0.21	119
21_07_2010	117	128	52	246	1.72	9	0.64	39
15_09_2010	53	69	3	38	2.77	9	0.16	65
11_11_2010	203	365	137	296	0.15	9	0.01	7
15_12_2010	309	443	164	390	0.30	9	0.10	10
20_01_2011	219	281	44	188	1.83	9	0.24	30
17_02_2011	148	180	16	70	3.11	9	0.16	49
22_03_2011	147	86	5	62	5.09	9	0.17	86
19_04_2011	90	47	1	16	2.54	9	0.05	74
24_05_2011	68	34	5	13	0.37	8	0.00	21
28_06_2011	282	140	42	237	0.46	9	0.01	15
26_07_2011	195	290	99	141	2.15	9	0.40	31
30_08_2011	183	255	32	191	2.89	9	0.20	41

Date	SM	SL	MA	MI	error MS*	d.f.	P*	CV%*
27_09_2011	131	141	3	101	2.43	8	0.07	47
10_11_2011	400	554	586	332	0.47	9	0.79	12
13_12_2011	461	541	274	294	0.45	9	0.43	12
23_01_2012	417	593	191	298	0.68	9	0.31	15
08_03_2012	326	547	123	316	0.87	8	0.15	17
11_04_2012	281	431	109	177	1.79	9	0.37	27
15_05_2012	232	374	44	143	1.10	8	0.03	23
Media	164	202	78	146	1.91			

***values calculated on log transformed data**

Discussion

Nitrate concentration in water percolation

Nitrate dynamics concentration in the suction lysimeters water was very variable in relation to the meteorological factors and the N uptake dynamics by crops. Under the climatic conditions of the region, we observed that NO_3^- leaching was very high in autumn and winter as reported Trindade et al., 2008, corresponding to the rainfall period and low N uptake by Italian Ryegrass in this time. The leaching process, in Mediterranean condition, begins on average in middle of October, after an accumulated precipitation of about 100 mm (Carneiro et at 2012). The magnitude of the losses is mainly determined by the NO3−-N amount in the soil at the start of the leaching period. We observed that the fertilizer mode didn't affect the NO_3^- concentration in autumn. The mineral treatment, unfertilized in this time, could be considered a control treatment. It assumes that the NO_3^- concentration peak was due a high organic matter mineralization in the pre-sowing period of the Italian ryegrass, favored by silting of crop residues and by the high temperatures in this period that can promote soil microbial activity in agreement with those reported by Liang, et al (2010). Furthermore, Trindade et al., 2008 showed that residual amounts of NO_3^- -N in soil in October after the maize harvest ranged from 48 kg NO_3^- -N ha^{-1} in the control treatment to 278 kg NO_3^--N ha^{-1} in the heavily fertilized treatment. The N mineralized between October and March may have represented an important NO_3^--N source. This N can contribute to high leaching losses even when the NO_3^- amount that remained in the soil after the maize harvest is small (Trindade et al., 2008). Vert`es and Decau (1992) estimated leaching losses of about 100 kg N ha^{-1} during autumn and winter after maize crop that had left NO_3^- small quantities in the soil; they estimated that 90% of the leaching losses originated from N mineralization. Measures that could reduce nitrogen concentration in the soil profile at the beginning of autumn should be taken. The rational fertilization and cropping practices of the summer culture to avoid high residual-NO_3^- values, the early establishment of the autumn–winter crops or a reduction in the amount of N applied at the late sowing of those crops could be an example of measures that could be taken (Carneiro et al., 2012).

As reported by Carneiro et al., 2012 the mineral N should be applied only though top-dressing application(s) mainly when the sowing of autumn–winter crops is delayed, and not as usually occurs at crop sowing. Otherwise, the N losses by leaching would be significant. This may explain why in similar agro-environmental conditions, it was observed that N applied at tillering was recovered more efficiently than that applied at emergence (Kirda et al., 2001; López-Bellido et al., 2005). Nevertheless, we think that the no use of fertilizers in autumn can't be considered a valid solution to mitigate significantly the leaching of nitrates in the autumn-winter period. Also, compared with the other treatments, mineral fertilizer did not mitigate significantly the nitrates leaching in the autumn-winter as reported ND. Leaching losses are linearly related to N inputs, over-simplifying a complex N loss function which depends on the interactions between over-winter rainfall, soil type, cropping, and the rate/timing of fertilizer/manure applications (Silgram et al., 2001).

In addition, to make a fertilization plan with the organic fertilizer use is very difficult because the slurry is a source of NO_3^--N through mineralization during the autumn and winter (Cameron and Haynes, 1986). Due to the difficulty in predicting the slurry N availability, the amounts of fertilizer N applied frequently exceed the crop requirements leading to the accumulation of high levels of NO_3^--N in the soil after harvest.

In contrasting to Morari et al., 2012 (different cropping rotation, Maize- Wheat -Maize) we observed that the NO_3^- dynamics concentration was very low at the end of winter (period April-May) except for the dry years. This has been attributed to the low winter rainfall of that particular period. While, highest concentrations were recorded after fertilization and irrigation in according to Perego et al., 2012, in summer in maize,

Conclusion

In the Mediterranean intensive grassland systems, the seasonal dynamics of the concentration of nitrates is very wide. In Autumn, the nitrates concentration in water percolation didn’t depends on the fertilization systems. In summer the water balance can be managed through irrigation because in this time no rain falls and therefore can allow almost complete control of the water percolation and NO_3^- leaching. While the Autumn is a critical period where there were highest peaks of rain, water percolation and NO_3^- concentration in the soil. Exchanging mineral nitrogen for manure or slurry does not reduce nitrogen leaching and other nitrogen management strategies are necessary to address nitrogen losses. We propose that check era of early sowing of ryegrass to promote nitrates absorption (Brinsfield and Staver 1991) or insert another crop in this system that can uptake the nitrate that there is in the soil in autumn is necessary.

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Chapter 3

Assessment of the potential nitrate leaching from agricultural sources in a Mediterranean NVZ

Abstract

The research aims to assessment of the relations, at territorial scale, between nitrogen balance and nitrate concentration in groundwater in an Nitrogen Vulnerable Zone in intensive dairy farming systems under Mediterranean conditions. A pilot area, designated as "transect", was identified through underground flows and the spatial distribution of the nitrates concentration in groundwater. Nitrogen surplus was calculated for the agricultural years from January 2007 to May 2011 in 102 fields and it is computed as total nitrogen inputs minus total nitrogen outputs. The average N surplus in the transect was related to the water surplus determined with EPIC model for the principal cropping systems. The results showed that the high nitrate concentration in groundwater were clearly associated to a high nitrogen surplus at field scale in this specific contest, though only some 15-20% of the N surplus was found as nitrate in groundwater. The water balance indicated that most of the leaching occurs between November and February.

Key words: N surplus, N loss, double cropping, Silage maize, Italian ryegrass, groundwater, intensive dairy farming, Nitrate pollution.

Introduction

Nitrogen (N) losses from agriculture are negatively impacting a, air, and surface water quality (Wendland et al., 1993, Ten Berge, 2002, Delgado et al., 2008) especially in intensive cropping systems that are located in Nitrates Vulnerable Zones (NVZs) as defined by the EU nitrate directive. Nitrate pollution is often related to intensive agricultural systems, however the relationships between nitrogen inputs and nitrate concentration in groundwater are site-specific as they depend on a number of local factors influencing the processes of the nitrogen cycle (Morari et al., 2012). Predicting the movement of solutes through soil is difficult due to the heterogeneous nature of soils (Addiscott, 1996), of climate characteristics and management practices (Meisinger and Delgado 2002; Havlin 2004). To define the N balance (Nbal) for determine the N surplus in soil system considering all input components (mineral and organic fertilizers application, atmospheric deposition, mineralization, N from crop residues) and output (N uptake through harvested crops, soil leaching, immobilization, volatilization, denitrification, erosion and surface runoff), is rarely feasible, because the necessary data are not often available and the date collection is costly. Leach et al., 2003 reported that studies at the watershed scale can allow to calculate the N surplus and quantify the importance of the different input and output sources from the system, moreover these are able assess the potential impact of cropping systems on quality of surface and groundwater water (Osborne e Wiley, 1988; Rossi Pisa et al, 1996;. Gardi, 2001). The N balance represents a tool capable of identifying potentially critical situations and it can help to support more sustainable management of the N cycle. The Nbal method is widely used as a synthetic indicators of N use efficiency in agro-ecosystems to support the implementation of agro-environmental policies (Öborn et al., 2003; Oenema et al., 2003) and as a performance indicator (Bassanino et al., 2011). In general, it can be calculated at the farm scale (Ventura et al., 2008; Grignani and Acutis 19944; Simon e Le Corre 1992; Simon, 1995; Argenti et al., 1996) for the soil surface or at the territorial scale (Sacco et al., 2003) and its versatility is well documented. The OECD indicates the gross nitrogen balance methodology as the appropriate indicator to calculate comparable Nbal on a regional or national scale (OECD, 2007). Nevertheless, when Nbal are calculated as the difference between N application rate and N uptake by harvested crops (e.g.,Bach 1987; Sieling and Kage 2006), without accounting for mineralization and immobilization (Lord et al., 2002; Oenema et al., 2005), the Nbal proves to be a poor estimator of N amounts lost to the environment on a single year basis, and it is only weakly correlated with actual measured N losses (Schröder et al., 2004; Sieling and Kage 2006; de Ruijter et al., 2007; Rankinen etal.,2007). Buczko et al. (2010) tested four simple N loss indicators to assess their predictive properties and highlighted that Nbal indicator, for long term-averaged data, is better suited as indicated by also Sieling and Kage, 2006. Schroder et al., 2009 have shown that effective N surplus, based on the difference between the summed inputs of the plant available N and harvested N, proved to be the best indicator of leaching (R^2= 0,86) of cut grasslands. We hypothesized that the Nbal is a good indicator of the potential leaching of nitrate in the NVZ at territorial scale. The Nbal can effectively complement the territorial monitoring identifying management systems more sustainable and to formulate efficient hypotheses of process for the mitigation of nitrate pollution in this area. The study aimed to contribute to the quantitative assessment of the relations between Nbal and nitrate concentration in groundwater in an NVZ in a Mediterranean context of intensive agriculture, which can be considered representative of the Mediterranean irrigation districts.

Material and Methods

Survey site

The study at territorial scale was carried out over an area of about 425 ha located in central west of Sardinia (Arborea, Province of Oristano, Italy; 398 46' N; 88 37'E) (Figure 1) a Nitrate Vulnerable Zone (NVZ). The climate is typically Mediterranean and the specific characteristics are given in the previous chapters. The soil was characterized by sandy texture (90% CV=11%), sub-acid pH (6.7 CV=13%) and variable organic matter content (1.0% CV=47%) (data from Regional Agency for Environmental Protection of Sardinia, ARPAS). The production systems are based on intensive dairy cattle, with about 35,000 animals in an area of 5500 ha (ISTAT, 2010), making it the leading dairy producer in Sardinia and one of the most important dairy district in Italy. The forage cropping systems for the dairy livestock are based on the double cropping of silage maize (*Zea mays*) and Italian ryegrass (*Lolium multiflorum*) for hay or a winter cereal for silage, representing over 80% of the irrigated land, and with the remaining area being used to grow alfalfa (*Medicago sativa*) and horticultural crops.

A hydrogeological study allowed us to characterize the multi-layered aquifer hosted in the quaternary outcrops sands throughout the area. This aquifer was characterized by a low depth to groundwater and it was the most vulnerable to pollution. Nitrate concentrations in groundwater in mg l^{-1} were obtained from ARPAS's available on a quarterly basis from 2007 to 2009. Nitrate concentration data of 34 wells relative to the aquifer surface in NVZ area were spatialized using the Kriging method but only 11 wells have concerned the survey (Figure 2). The pilot area, indicated as transect, has been identified through to underground flows and spatial distribution of the nitrates concentration in sandy aquifer (Figure 3) from area to high nitrate concentration to very high concentration. Land use in the transect was considered representative of whole NVZ area. In this areas were conducted interviews to farmers on agronomic management to cultivation systems in for each of the fields (102 fields in total) in study area. The survey involved 25 farms and including 22 dairy cattle farms, 2 horticultural farms and 1 nursery. Data collection were related to time 2009-2011 and have contributed to the calibration of simulation EPIC model (Williams, 1995). Land cover for the main cropping systems in rotation, has been computed as a proportion of total size of the transect and was reported in Table 2.

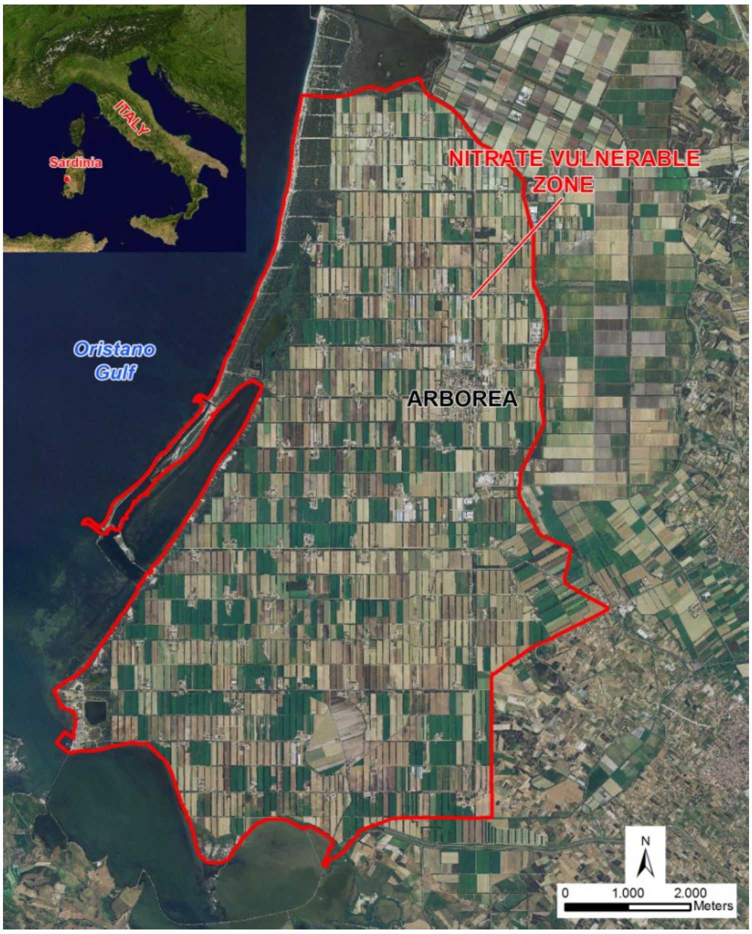

Figure 1. Study area

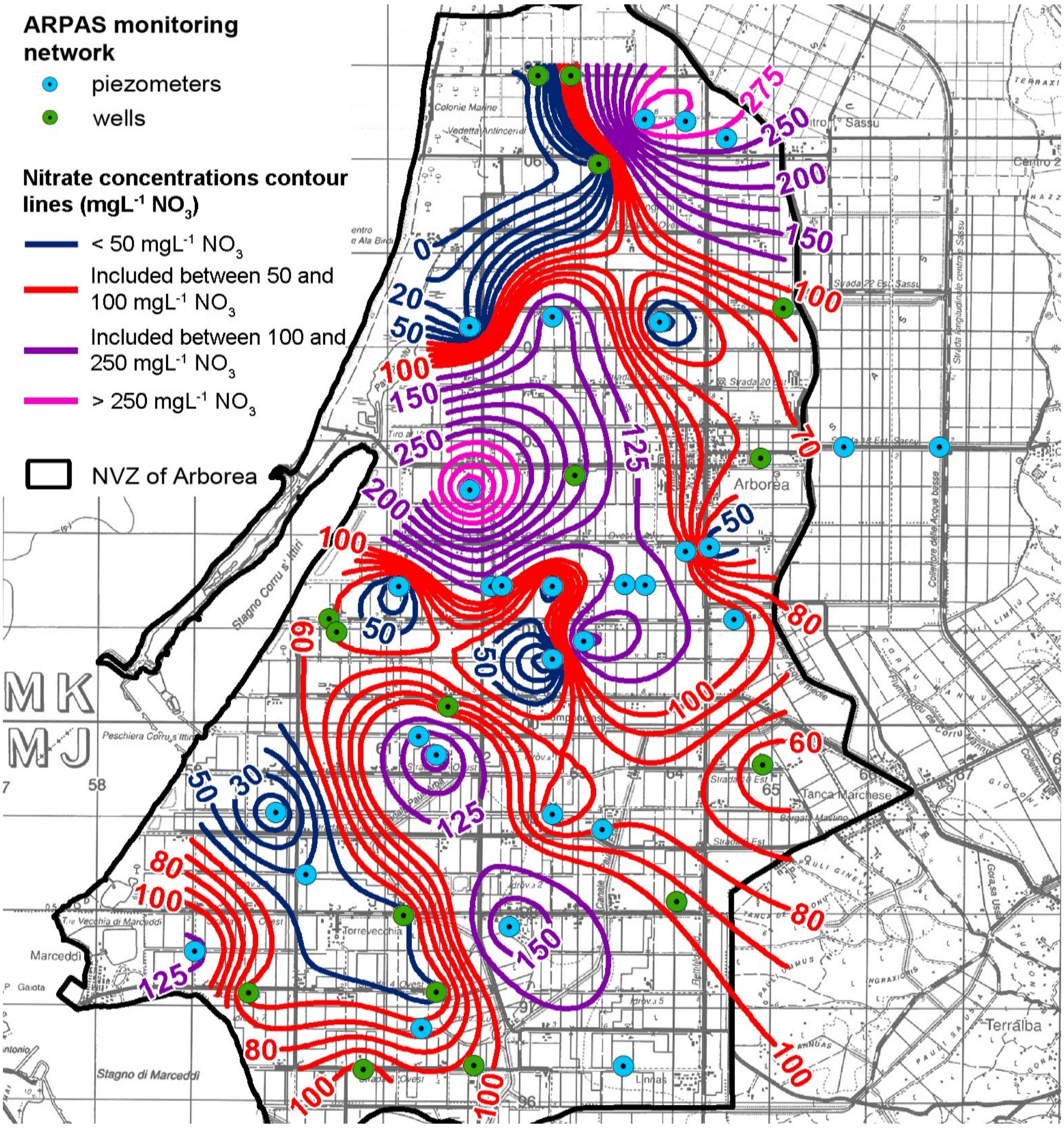

Figure 2. Average annual nitrate concentration in NVZ from 2007 to 2011.

Water balance

The average N surplus in the transect was related to the water surplus determined with EPIC model for the principal cropping systems (Maize-Italian ryegrass and Alfalfa) to order to calculate the amount of N potentially leachable, assuming zero the other losses.

N balance and N surplus

A soil surface budget methodology was adopted for calculating the Nbal; this ensured all major fluxes were accounted for, especially those potentially affected by mitigation. The Nbal was calculated for the agricultural years from January 2007 to May 2011.

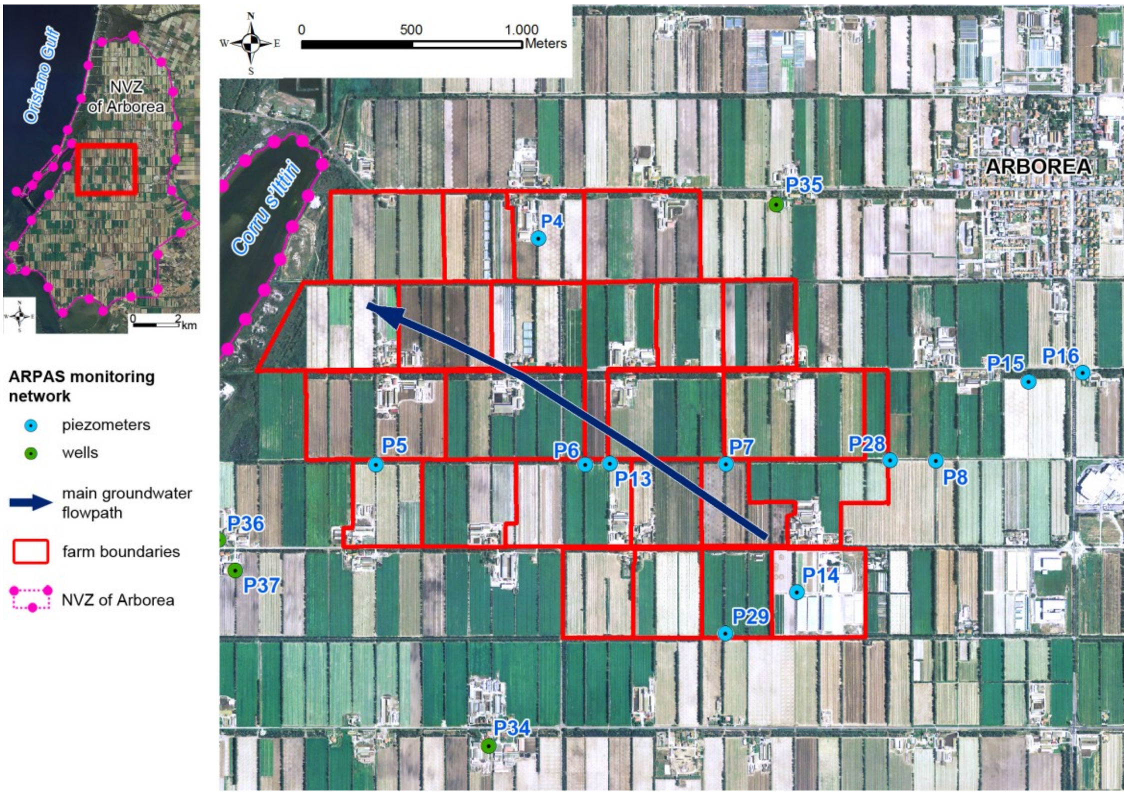

Figure 3. Transect area and underground flows.

The Nbal was determined in according to the methodology proposed by Grignani and Acutis (1994) adapted to the scale of the field. A soil surface budget methodology was adopted for calculating the nitrogen balance; this ensured all major fluxes were accounted for, especially those potentially affected by mitigation. The Nbal is computed as total nitrogen inputs minus total nitrogen outputs. Inputs to the *Nbal* are (i) biological

N fixation in the soil -*Nfix*, (ii) atmospheric deposition- *Ndep* (iii) livestock manure and mineral fertilizer-*Nfert*. Moreover, output elements was considered the amount of N removed (*N removal*) per tonne of crop yield Nremoval by harvested products:

$$Nbal = (N\,fert + N\,fix + N\,dep) - N\,removal$$

N fixation data in the literature reported values of 2.7% shoot dry matter in lucerna (e.g. Kristensen et al 1995).We estimate that in relation to the specific local conditions in the presence to sandy soil with a high organic matter content and views to the crops management practices (organic fertilization with cattle slurry in pre-sowing) 2% is more reliable. Due to the lack of official data, we assumed that the wet and dry N depositions were 2.4 kg ha^{-1} yr^{-1} (Markaki et al., 2010). Loss of nitrogen so determined was selected from IRENA (Indicator Reporting on the Integration of Environmental Concerns into Agriculture Policy), European Project coordinated by the European Environmental Agency (COM, 2000), as a nutrient budget indicator among 35 Agro-Enviromental indicators AEIs, calling Gross Nitrogen Balance.

N inputs from fertilizers were calculated from the amount of fertilizers applied and the forms and concentrations of nutrients present. The main fertilizers used in study transect were cattle slurry, cattle manure, Entec 26, urea, ammonium nitrate, Ca nitrate, potassium nitrate, potassium sulphate compound or mixed fertilizers. N inputs from organic manures was calculated by chemical physical analyzed data on 56 samples collected in three different years pre-fertilization in experimental field as reported in first and second chapter. N inputs from irrigation water were calculated and the N total amount was 4,5 mg L^{-1} and it was considered negligible. Data of N output were based on data of crop yields provided by interview to farmers and on N concentration data obtained by literature dates and in the experimental fields reported in table 1. The interview to farmers had including data relative to land use, soil management, fertilizer amounts and timings as reported in a specially designed data collection sheet (Attachment a).

Table 1 N (% DM) used for the calculation of the N removed by crops.

Crop removal	N (% DM)	Literature
Alfalfa forage	3.0	USDA hay quality guidelines
Italian Ryegrass forage	1.1	Misurated at field scale
Triticale forage	1.4	Licitra et al., 1996 rielaborated
Maize silage	1.1	Misurated at field scale
Italian Ryegrass silage	1.5	Licitra et al., 1996
Triticale silage	1.4	Giola et. al 2012
Grass-silage of Italian ryegrass	2.6	Licitra et al., 1996 rielaborated
Italian ryegrass dry forage	1.7	Licitra et al., 1996
Maize mash	1.4	http://alimenti.vet.unibo.it

Results

Water balance

The water balance is reported in figure 4. In the studied period, rainfall occurred in almost every month, but rain events were more frequent and intensive in autumn and spring. Extraordinarily heavy rainfall occurred in autumn 2010 (240 mm between October and November), while autumn 2007 were characterized by amounts of rainfall below the average of the period. Summer storms occurred almost every year. The higher total average precipitation was 810 mm and it was verified in 2010, while the drier year was 2007 where the total average precipitation recorded was 490 mm. The average annual N surplus was 236 mm. Major water surplus occurred in autumn–winter during each year. The water surplus was the lowest in 2007, about 60 mm, while in 2010 we observed the highest values, about 316 mm. The linear regression showed a strong influence between precipitation (y) and water percolation (x) observed in this time: y = 1,158x + 30,24 (r^2 = 0.76). Every year, irrigation started at the end of May but water fluxes increased in June, reaching a maximum daily value of around 470 m^3 ha^{-1} in 2010 in August 2007, and about 360 m^3 ha^{-1} in 2008-2011 in July and August. In summer, during the irrigation period there weren't significant fluxes of water leaving the system. The water balance for double crop maize-ryegrass showed that in five years about 65% of the percolation occurred in the period from October to January where the rainfall in general exceeding the ET by more than 270 mm averagely .

Nitrogen balance and N surplus

The area study of transect has affected average Agriculture Land Used (ALU) of 337 ha divided in the crop rotations reported in Table 2. In the 5 -years that have involved the agronomic survey from 2007 to 2011 land use in the transect has not changed much. The rotation maize-ryegrass has represented 80% of the surface ALU considered in the transect, while the rotation maize-triticale and maize- vegetable crop (mainly potatoes and carrots) have represented 5% and 2% respectively. The lucerne cultivated in the transect covers an area of hectares 30 (about 9%) while ALU area that remainder was cultivated by strawberries, zucchini, eggplants, tomatoes, watermelons, melons. The maize-ryegrass system showed the average annual N surplus higher than the other, about twice that found under maize-triticale system about 380 kg N ha^{-1} and 180 kg N ha^{-1}, respectively (Table 2). Only Italian ryegrass crop has showed a N surplus of about 190 kg N ha^{-1} against the 180 kg N ha^{-1} of maize.

The major N inputs by fertilizers were applied in maize and Italian ryegrass and the respectively doses were 375 kg N ha^{-1} and 290 kg N ha^{-1}, while the N inputs in Triticale and Alfalfa were 212 kg N ha^{-1} and 186 kg N ha^{-1}(Table 3). In all crops the N fertilizer inputs were higher than N removals from crops, except in Alfalfa. Nitrogen fertilizers were mainly applied in early fall months as organic fertilizers for winter crops and end May early June for maize. The mineral fertilizers were applied in February for Italian ryegrass and spilt in two rate in June for Maize (one dose in pre-sowing and one about after 20 days). Major N output was represented by crop harvest (Table 3). The Lucerne crop showed the major N removal (270 kg N ha^{-1}), followed by Maize crop (197 kg N ha^{-1}).

Table 2 Average N surplus (kg N ha^{-1}) per the principal crop rotation and Agricultural land use (in hectares and in percentage on total acreage) in the transect.

Cropping	**ALU (ha)**	**ALU (%)**	**Average N surplus (kg N ha^{-1}) 2007-2011**
Maize-Italian ryegrass	271	80	384
Maize-Triticale	15	5	181
Maize–horticulture crop	6,6	2	233
Alfalfa	30	9	2
Other crop	17	5	165
Total	340	100	328

ALU= Agricultural Land Used

Table 3 N balance of representative crops in the transept (average kg N ha^{-1}±St. err.)

Culture	Input	Output	N Surplus
Maize	375± 9.2	197±1.4	180±9.4
Italian Ryegrass	279±6.2	91±2.0	191±6.2
Triticale	212±19.3	114±9.4	101±21.4
Alfalfa	186±8.4	271±10.8	3±3.6

The N surplus in the transect showed a decreasing trend from 351 kg N ha^{-1}in 2007 to 291 kg N ha^{-1} in 2011. In the transect was observed a high nitrate potential leaching when there was higher N surplus and the lowest water surplus. The average nitrate concentration was estimated and was average 671 mg L^{-1}, five-six times higher than that found in groundwater.

The N surplus was significantly correlated with N application rates, in fact the linear regression analyses (figure 4) showed a strong influence (r^2= 0,80). Besides, the nitrate concentration estimated by cropping systems in the transect from 2007 to 2011, had showed strong relation between the N surplus and nitrate concentrations found in groundwater in the transect wells (r^2=0,80) (Fig. 5). The distribution of nitrates in groundwater does not match the distribution of N surplus data by field figure 6.

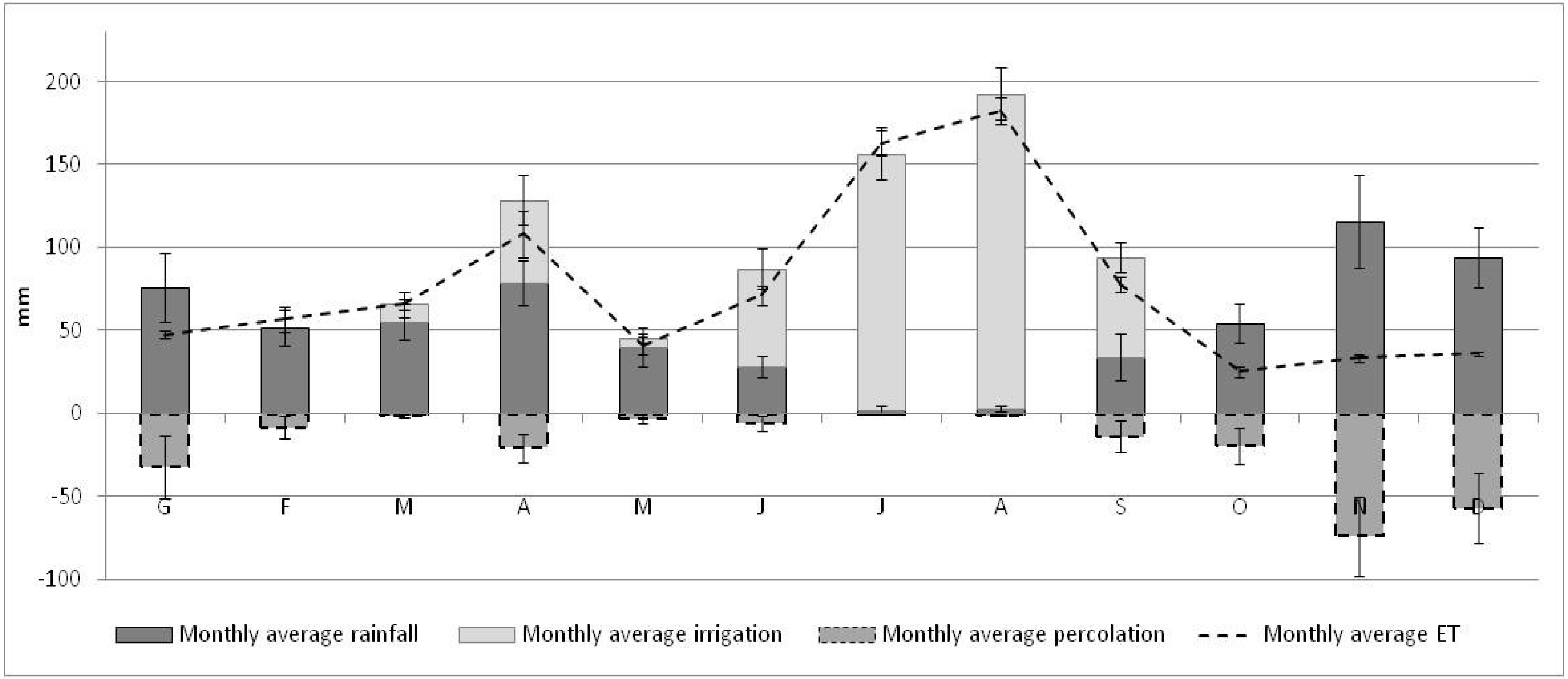

Figure 4. Dynamic of water balance in Maize-Ryegrass system from 2007 to 2011. Error bars indicate standard errors.

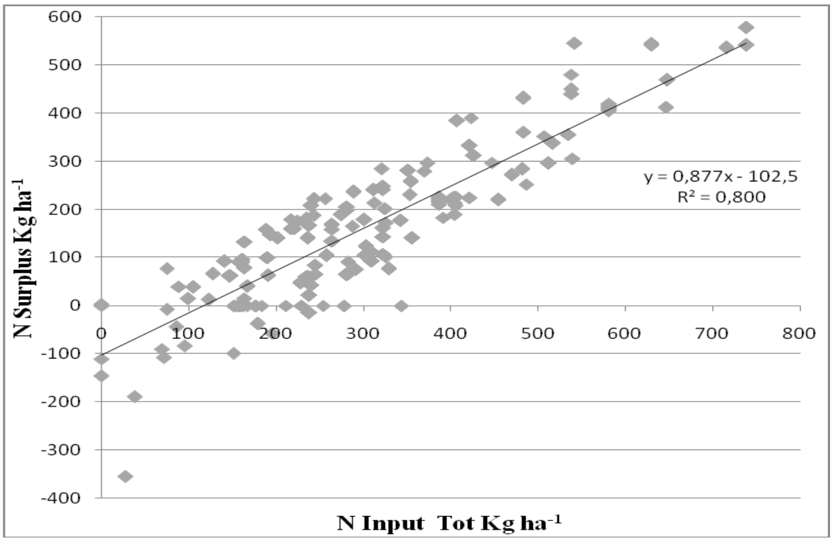

Figure 4 Correlation between N surplus (kg ha^{-1}) and N input (kg ha^{-1}) by fertilizer in cropping systems in transect.

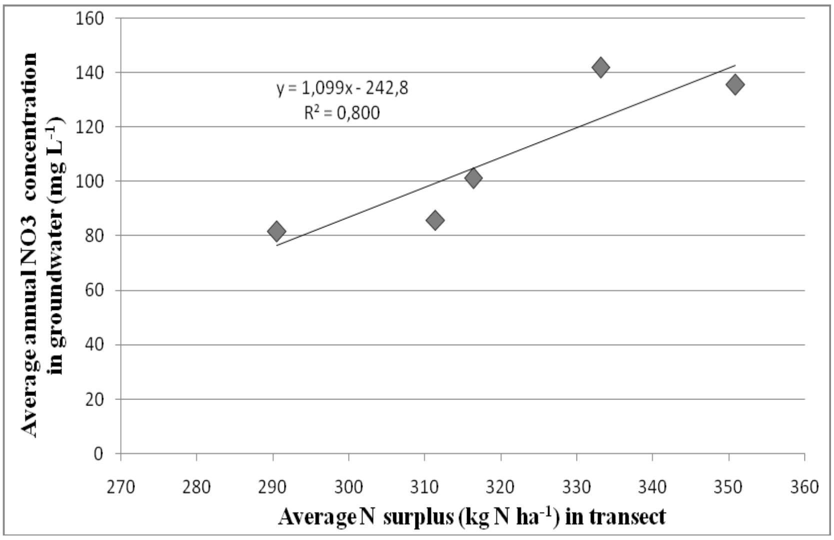

Figure 5 Correlation between N surplus (kg N ha^{-1}) estimated by cropping systems in the transect and the average annual NO_3concentration (mg L^{-1}) founded in groundwater in the wells transect.

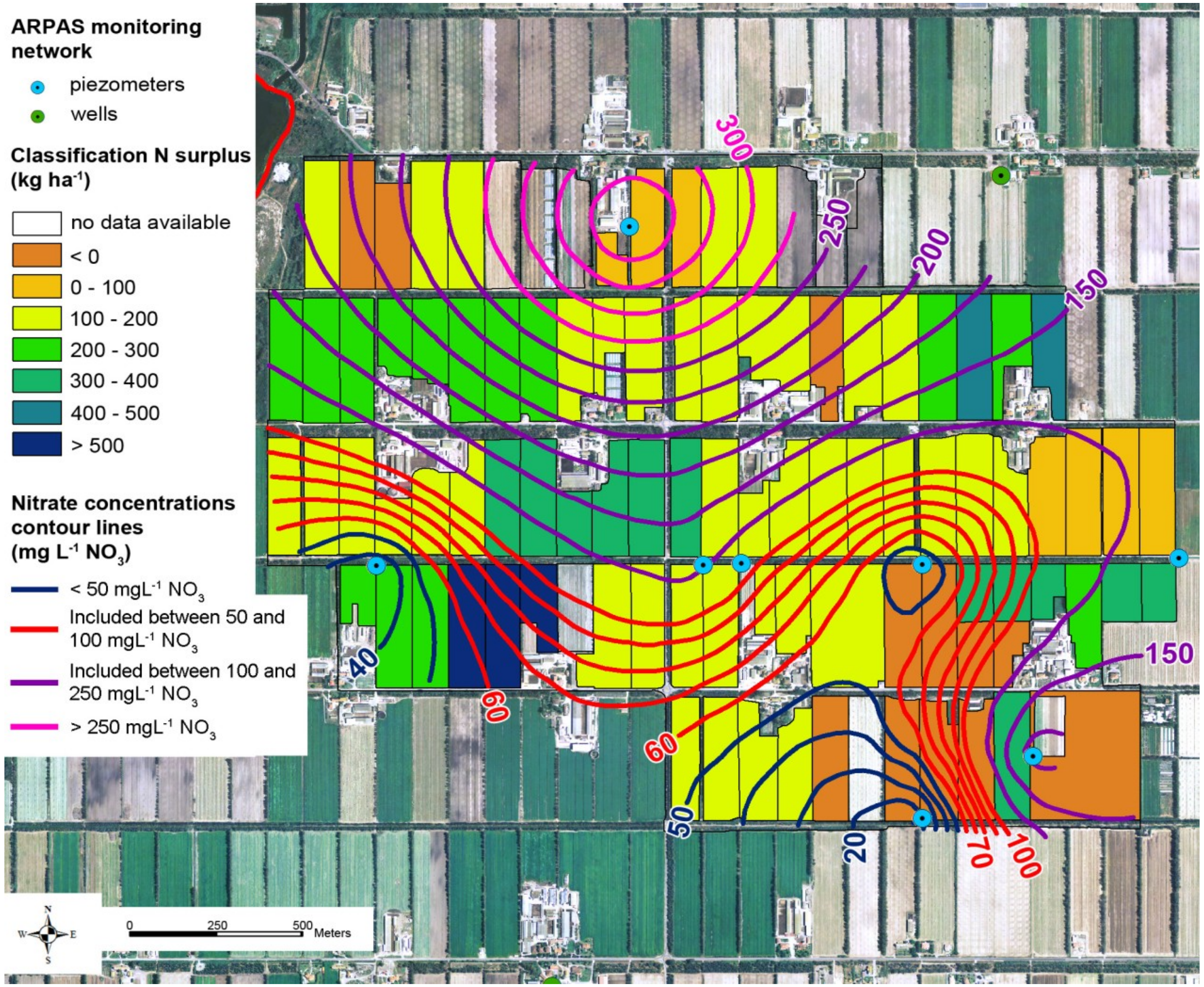

Figure 6. N surplus (kg N ha^{-1}) by fields in the transect area and spatialization of groundwater nitrate concentration.

Discussion

Despite the ND specifications in this area and the presence of fertilization plans on the territory we have observed clearly that there was high N surplus mainly in Maize-ryegrass system, despite high crop yields. Several studies observed average N surplus lower values than those observed in this area in only silage Maize or Maize -Italian ryegrass rotation (Zavattaro et al., 2012; Bassanino et al., 2011; Grignani et al., 2007).

Grignani and Zavattaro (2000) reported surplus was 128–335 kg N ha^{-1} $year^{-1}$ and Mantovi et al. (2006) calculated similar values in the Emilia-Romagna Region, Northeastern Po Valley, in cropping system based on silage maize and other cereals (grain sorghum and winter wheat).

High N surplus in Maize-Italian ryegrass system was due mainly to the high organic fertilizer doses distributed in pre-sowing of ryegrass, which is followed by the wettest period and N removal from crop is minor because it is in early stages of development. Synchrony of N supply with crop demand is essential in order to ensure adequate quantity of uptake and utilization and optimum yield (Fargeria and Baligar 2005). The triticale appears more conservative than the ryegrass. The fertilizers input in Maize-Triticale system were the same or minors respect with ryegrass system, but with higher N removal.

Numerous studies have shown that yields do not increase significantly when N fertilizer application rate exceeds a certain value, but N losses increases sharply (Raun and Johnson, 1995; Porter et al., 1996; Bhogal et al., 2000; Zhong, 2004).Several studies observed that annual N surpluses in different systems were significantly and positively correlated with N fertilizer application rate (Ju X.T et al., 2006, Grignani et al., 2007).

In this study the N surplus can be a useful indicator for understand the N load in each system but the apparent poor match between the N surplus at field scale and nitrate concentration in groundwater indicates that is not appropriate to predict the level and entity of nitrate pollution of groundwater under the specific conditions. In fact, N surplus is not all lost through leaching but is particularly important for denitrification and volatilization losses. The ammonia volatilization likely contributed to N gaseous losses, since several studies estimated around 10% of the N added with urea and ammonium nitrate (Puckett et al., 1999; Ventura et al., 2008) and 35% of NH_3–N added with manure (Hansen, 2006) as reported by Morari et al. 2012.

Moreover several studies showed that shallow water tables are frequently associated with poorly drained soils and anaerobic conditions. Under these conditions, nitrate denitrification can occur in the presence of organic carbon and denitrifying bacteria Morari et al., 2012. Moreover several studies showed that shallow water tables are frequently associated with poorly drained soils and anaerobic conditions. Under these conditions, nitrate denitrification can occur in the presence of organic carbon and denitrifying bacteria (Morari et al., 2012). Sacchi et al., 2013, reported that the distribution of nitrates in groundwater does not fully match the distribution of any of the identified N sources and that the denitrification is estimated to remove about 40–60% of the initial nitrates from groundwater in maize fields.

Conclusions

High nitrate concentration in groundwater were clearly associated to a high nitrogen surplus at field scale in the context of intensive dairy farming systems under Mediterranean conditions, dominated by silage maize and Italian ryegrass double cropping system. The N surplus was closely related to the high N input, which in turn is associated to the high farm gate nitrogen surplus of the dairy farming. The water balance of these farms indicated that most of the leaching occurs in the cold season (Nov-Feb), when the natural water surplus represent 72% of the total annual surplus even if only 29% of the total (rainfall+irrigation) annual water input. This is because of the typical Mediterranean winter season temperature and short daylength that constrains evapotranspiration and plant N uptake and the relatively high rainfall. The natural water surplus leading to N leaching cannot be controlled by farmers, as soil saturation occurs in winter almost independently of summer irrigation in summer, when rainfall is negligible. Therefore, an average high N surplus makes these farmlands very vulnerable to nitrate leaching. Only some 15-20% of the N surplus was found as nitrate in groundwater, assuming that it was diluted by just the percolation water. The remaining excess N would be immobilized as organic N, volatilized, denitrified or diluted by groundwater fluxes. Further studies are needed to make a quantitative assessment of the contribution of denitrification to the reduction of nitrate concentration in groundwater or N_2O emissions. It is likely that the increase in crop N use efficiency and the reduction of N inputs could effectively contribute to mitigate nitrate pollution without significant reduction of crop yield, but with increased costs for the disposal of effluents outside the NVZ.

Attachment

Transect n°__________ Message n°_______________location_______________________Owner_________________________ Type farms____ Members Family _____ Septic tank □ Sewerage□ Field n° ___________Coordinates__________________Sup.(ha)_________________												
2012	**Culture**	**Hybrid / Variety**	**Tipe Fertilizers**	**Dose**	**Time**	**Yield (t ha^{-1})**	**Sowing Date**	**Harvesting Date**	**Water consumption**	**Carrying capacity**	**n° interv.**	**n° hours**
C1												
C2												
C3												
2011												
C1												
C2												
C3												

Attachment a. Template used in interview to farmers

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Acknowledgements

This study was carried out within the Agroscenari Project- Adaptation scenarios of Italian agriculture to climate change- Project funded by the Italian Ministry of Agriculture and Forestry Policies with DM 8608/7303/08; IDRISK Project= Risk of pollution and prevention of the degradation of groundwater resources (Regione Autonoma della Sardegna, L.R. 7/2007); KNOW project =implementing the Knowledge of NitrOgen in groundwater (Regione Autonoma della Sardegna, L.R. 7/2007).

I thank the research team and all those farmers of Arborea who kindly accepted to be interviewed, my supervisor Prof. Pier Paolo Roggero, Dr. Giovanna Seddaiu and Dr. Luigi Ledda for their help in my training, Dr. Chiara Cappai for the chemical analyses and the support in lab, Eng. Alberto Carletti for his collaboration in the field experiment and the interviews; Dr. Luca Doro for the EPIC simulations; all the NRD staff.

I'm grateful to the Sardo's family for hosting our experimental activities in their farm and the Arborea Farmers' Cooperative for their fruitful collaboration during this study and all those farmers that kindly accepted to be interviewed.

A special thank to Dr. Armen Kemanian, Penn State University- Pennsylvania-USA, for his valuable comments and helpful discussions during my stay in the US.