Nutr Cycl Agroecosyst (2015) 102:359–370
DOI 10.1007/s10705-015-9707-4

ORIGINAL ARTICLE

Inorganic nitrogen losses from irrigated maize fields with narrow buffer strips

Osvaldo Salazar · Claudia Rojas · Fernando Avendaño · Piero Realini · Francisco Nájera · Yasna Tapia

Received: 16 March 2015 / Accepted: 5 June 2015 / Published online: 12 June 2015

Abstract Vegetated buffer strips (BS) can help prevent nitrogen (N) losses from fields by subsurface lateral flow, thus protecting water resources. The purpose of this study was to determine if narrow BS would effectively remove dissolved inorganic N from subsurface lateral flow. Nitrate–N (NO_3–N) and ammonia–N (NH_3–N) concentrations in subsurface lateral flow were measured at 1 m depth in a BS system consisting of five treatments: G: strip of grass (*Fescue arundinacea*); GS: strip of grass and line of native shrubs (*Fuchsia magellanica*); GST1: strip of grass, line of shrubs and line of native trees 1 (*Luma chequen*); GST2: strip of grass, line of shrubs and line of native trees 2 (*Drimys winteri*); and C: bare soil as control. Water samples for the NO_3–N and NH_3–N measurements were collected between June 2012 and August 2014 in observation wells located at the inlet (input) and outlet (output) of each treatment. The analyses showed that vegetated BS had NO_3–N removal efficiency ranging from 33 to 67 % (mean 52 %), with the G treatment showing the best performance in reducing NO_3–N concentrations in subsurface lateral flow. The BS treatments were not effective in reducing NH_3–N concentrations. The results suggested that N uptake by grass is the main process associated with the NO_3–N retention capacity of vegetated BS.

Keywords Fescue · Filter strip · N mineralisation · N uptake: nonpoint source pollution · Water quality

O. Salazar (✉) · F. Nájera · Y. Tapia
Departamento de Ingeniería y Suelos, Facultad de Ciencias Agronómicas, Universidad de Chile, Casilla, 1004 Santiago, Chile
e-mail: osalazar@uchile.cl

C. Rojas · F. Avendaño · P. Realini
Programa de Magíster en Manejo de Suelos y Aguas, Facultad de Ciencias Agronómicas, Universidad de Chile, Casilla, 1004 Santiago, Chile

Introduction

Extensive research has shown that the scale of the nitrogen (N) environmental issue has shifted from a local pollution problem to a continent-scale problem involving widespread pollution of oceans, often on a global scale (Doney 2010). In fact, humans have already doubled the rate of N entering the land-based N cycle and that rate is continuing to climb (Vitousek et al. 1997). Modern agriculture in particular has been recognised by both farmers and environmentalists as a significant source of N water pollution (Galloway et al. 2008; Robertson and Vitousek 2009).

A particular concern are agricultural areas surrounding water bodies, where N transport from soils to surface water has resulted in serious eutrophication problems in many parts of the world (Anderson et al. 2002; Salazar et al. 2013a). This ongoing eutrophication has led to widespread hypoxia and large permanently reducing bottom areas in marine costal ecosystems (Vahtera et al. 2007). Moreover, several adverse health effects can occur when humans or animals consume high nitrate–N (NO_3–N) drinking water, such as methaemoglobinaemia in infants, cancer and respiratory illness (Ward et al. 2005).

In Chile, some recent studies evaluated the impact of agricultural activities on the level of NO_3–N in water bodies (Fuentes et al. 2014, 2015; Salazar et al. 2014). They showed that measured NO_3–N values in water bodies adjacent to agricultural areas in central Chile, where maize is the most common crop, usually exceed the Chilean water quality standard for drinking water (10 mg NO_3–N L^{-1}). This is because in addition to applying high levels of N, most farmers use a furrow irrigation system with low application efficiency (<45 %) (Nájera et al. 2015). The risk of NO_3–N leaching from these irrigation systems is clearly high (Quemada et al. 2013). Similarly, in other Mediterranean agroecosystems in the world, irrigated maize fields with high N doses have been highlighted as posing a high risk of creating diffuse N pollution areas (Berenguer et al. 2009; Gabriel et al. 2012; Salmerón et al. 2011).

To counteract the undesirable consequences of excessive nutrient loads to aquatic systems, mitigation measures to reduce N losses from agricultural areas have to be implemented. It is important to note that the movement of N, particularly NO_3–N, is directly related to hydrological processes. Movement of dissolved inorganic N in water away from fields occurs through surface runoff and subsurface lateral flow towards surface water bodies and leaching to the groundwater. For such water movements, it is also important to consider the concentration of ammonia–N (NH_3–N), which is highly soluble in water and it is oxidised in the environment by nitrification to NO_3–N. However, it is also noted that in the soil NH_3–N and NH_4–N are in equilibrium, where certain clay minerals are capable of fixing NH_4–N providing some degree of protection against rapid nitrification and subsequent NO_3–N leaching. Thus strategies to reduce N losses should consider how to intercept and retain both these dissolved inorganic N forms (NO_3–N and NH_3–N) in water. A broad range of best management practices (BMPs) have been used for maintaining surface and groundwater water quality, for instance applying an appropriate N rate, timely fertiliser application, incorporation of fertiliser, cover crops to scavenge dissolved inorganic N forms and appropriate cropping/residue management (Prokopy et al. 2008). However, these BMPs are ineffective for retaining any N compounds that have reached the boundary of the fields by surface runoff and subsurface lateral flow, and thus cannot prevent their potential entry into nearby surface waters.

One possible alternative is the use of vegetated buffer strips (BS). These are strips of land with permanent vegetation, usually trees, shrubs and grass, which are located adjacent to water bodies such as nearby lakes, streams, ponds and wetlands (Mayer et al. 2006; Borin et al. 2010). Vegetated BS are characterised by high species density and diversity and are located in a transition zone or 'ecotone', specifically at the interface of terrestrial and aquatic ecosystems (Burt and Haycock 1993). This is considered a dynamic rather than a static zone, the attributes and interconnections of which depend on its transitional position between adjacent ecological systems. In vegetated BS, the soil undergoes continuous or periodic saturation, with mostly anaerobic conditions nearly free of dissolved oxygen due to saturation by the groundwater or its capillary fringe.

Vegetated BS are designed to intercept surface runoff and thus reduce the amount of sediment and dissolved N carried by surface runoff to surface water. The removal of N from surface inflows is induced by deposition of sediment-bound N and exchange of dissolved N with the soil/litter surface. In addition, in vegetated BS soil microbes and vegetation can facilitate the transformation and uptake of N moving in subsurface flow, thus protecting surface water resources. The removal of N in subsurface flows by vegetated BS can be partly explained by vegetation uptake, but the main mechanism for removal is usually denitrification (Cors and Tychon 2007; van Beek et al. 2007).

Although extensive research has been conducted on BS in recent years (Webber et al. 2010; Dunn et al. 2011; Larson and Safferman 2012; Wang et al. 2012), there is still a need to clarify some controversial points. First, it is well known that BS width may be positively correlated to N removal effectiveness. However, there is the need to investigate the role of BS in intensive farming systems, where a realistic and shareable proposal may be to convert a small proportion of a productive field to narrow vegetated BS (Balestrini et al. 2011). Thus some studies have evaluated the N buffering capacity of narrow vegetated BS of width ranging from 5 to 8 m (Borin et al. 2005; Balestrini et al. 2011), and report promising results for reducing N losses. However, in a 4-year experiment in the Netherlands, Noij et al. (2012) found that a 5-m wide BS was ineffective in mitigating N loads from agricultural areas to surface waters.

Second, Mediterranean agroecosystems are characterised by strongly seasonal rainfall distribution, with 75 % falling in the autumn–winter months, including intensive rainfall events. In these areas, the true capacity of narrow BS to retain N forms moving from agricultural soils to nearby surface waters through pulses in subsurface lateral flow is still unknown. The main aim of the present study was thus to evaluate the effectiveness of 5-m wide vegetated BS in removing dissolved inorganic N forms from subsurface lateral flow. This was done in a 29-month field experiment located in a Mediterranean agroecosystem in central Chile.

Materials and methods

Site description

The study site was located at the Caleuche experimental field (CLC), Pichidegua commune, O'Higgins Region (34°25′S, 71°21′W, altitude 136 m a.s.l.) in central Chile (Fig. 1). The vegetated BS experiment, which was set up in April 2012, included two native tree species that tolerate prolonged periods of water logging, namely *Luma chequen* (Mol.) A. Gray and *Drimys winteri* J. R. et G. Forster. The shrub component included was *Fuchsia magellanica* Lam., which can also resist waterlogging. The BS were sown with *Festuca arundinacea* Schreb. as the grass component. The tree and shrub components were at least 2 years old at planting, at which time they were 0.5–1 m tall. Five treatments were evaluated (Table 1), with three replicates per treatment, in a randomised block design. The plot size was 12 m (long) × 5 m (wide), with an area of 60 m^2 $plot^{-1}$.

The vegetated BS ran along the edge of a 4.2 ha maize field, perpendicular to the dominant slope. The field is bordered on one of its edges by an open drainage channel, here designated CLC (Fig. 1), which flows downstream to a major watercourse. During the study period, the water level in this channel fluctuated from zero (no water in the channel) to 0.40 m, corresponding to about 1.8 m below the soil surface.

In the maize field, the soil is usually prepared using a disc plough in September and maize is sown in early October and harvested in early April (cropping season). Mean grain yield is 12.5 Mg ha^{-1} and maize stalks are removed from the experimental area after harvest. During the cropping season, the farmer applies 600 kg of compound fertiliser ($N–P_2O_5–K_2O$: 25–10–10) per ha at planting using subsurface

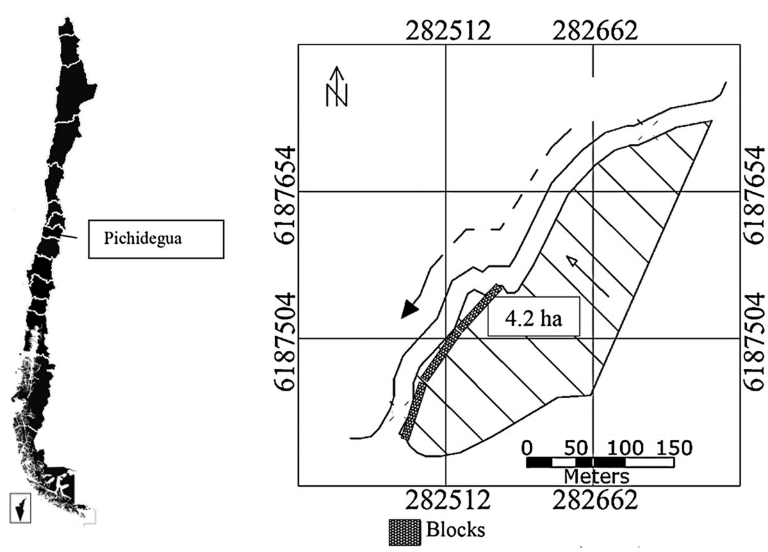

Fig. 1 (*Left*) Location of the Caleuche experimental site (CLC) in central Chile and (*right*) site map showing buffer strip blocks and UTM coordinates (X and Y axes)

Table 1 Description of the five treatments compared in the narrow buffer strip experiment

Treatment	Species	Description
Control (C)	–	Strip of bare soil 5 m wide
Grass (G)	*F. arundinacea*	Strip of grass 5 m wide
Grass + shrub (GS)	*F. arundinacea* + *F. magellanica*	Strip of grass 4 m wide, and row of shrubs (6 per row) 1 m wide
Grass + shrub + tree 1 (GST1)	*F. arundinacea* + *F. magellanica* + *L. chequen*	Strip of grass 3 m wide, row of shrubs (6 per row) 1 m wide and row of trees (6 per row) 1 m wide
Grass + shrub + tree 2 (GST2)	*F. arundinacea* + *F. magellanica* + *D. winteri*	Strip of grass 3 m wide, row of shrubs (6 per row) 1 m wide and row of trees (6 per row) 1 m wide

band placement and 650 kg urea per ha after planting at V7 stage using side-dressing. These fertilizers supplies 450 kg N ha^{-1} (150 kg N ha^{-1} from the compound fertiliser and 300 kg N ha^{-1} from the urea). According to nitrogen balance calculations, an estimated surplus of 250 kg N ha^{-1} was available for N losses if we would consider the N over-fertilisation and N mineralisation.

In addition, the maize is irrigated using a furrow system. In each of the two crop seasons of the experiment, approximately 15,000 m^3 water ha^{-1} were applied, divided between eight irrigation events during the cropping season. The soil was fallow between April and September.

The climate in the study area is classified as temperate with dry, warm summers, corresponding to Csb according to the Köppen–Geiger system (Peel et al. 2007). Mean annual temperature at the site is 14.1 °C and mean annual precipitation around 490 mm, mostly falling between May and October (INIA 1989). The rainfall distribution is strongly seasonal, with 75 % falling in the autumn–winter period (April–September). Climate data (i.e. precipitation, temperature, etc.) were obtained from a weather station located 2 km north–west of the experimental site, which also provided the weather data needed to calculate reference evapotranspiration (*ETo*) according to the FAO Penman–Monteith combination equation (Allen et al. 1998). Potential evapotranspiration (*Ep*) was calculated as:

$$Ep = Kc \times ET0 \quad (1)$$

where *Kc* is the reference evapotranspiration factor (crop coefficient) for grass calculated according to Allen et al. (1998). For bare soil (evaporation), grass and maize *Kc* values proposed by Allen et al. (1998) were used, whereas for shrub and trees were used values proposed in the literature for similar species (Salazar et al. 2013b).

Soil characterisation

The soils is characterised by flat topography (slope ranging from 1.6 to 2.0 %), thin clay soil deposits with imperfect drainage and a duripan (Cqm horizon) ranging from 0.50 to 1.0 m depth, which is classified according to Soil Taxonomy as part of the fine loamy, mixed, thermic family Typic Duraqualf (Casanova et al. 2013). The Cqm horizon effectively restricts downward seepage and nutrient transport in the area from the field to the outer edges of the BS systems and favours subsurface lateral flow from the maize field to the open drainage channel. The latter was monitored by a surface and subsurface (duripan level) topographical survey, as shown in Fig. 2a.

Soil samples were collected from the soil horizons for chemical and physical characterisation, including properties such as: pH in water (1:2.5), electrical conductivity (EC) in saturated extract, soil organic matter (SOM), total nitrogen (NT) according to Sadzawka et al. (2006) and cation exchange capacity (CEC) by NaOAc 1 N at pH 8.2. Soil physical properties such as bulk density (Db), available water content (AWC) and soil texture were determined according to Sandoval et al. (2012). The results of these analyses are summarised in Table 2.

Soil water dynamic was studied to evaluate its impact on dissolved inorganic N losses. Decagon soil water sensors (ECH2O EC-5) were set in each plot at 1 m depth and connected to a datalogger to record the variation in water content during the study period. In addition, cumulative infiltration and soil infiltration

Fig. 2 Treatment layout **a** in the field, where *solid line* shows the surface level and *dashed lines* the duripan level and **b** in vegetated buffer strip plots

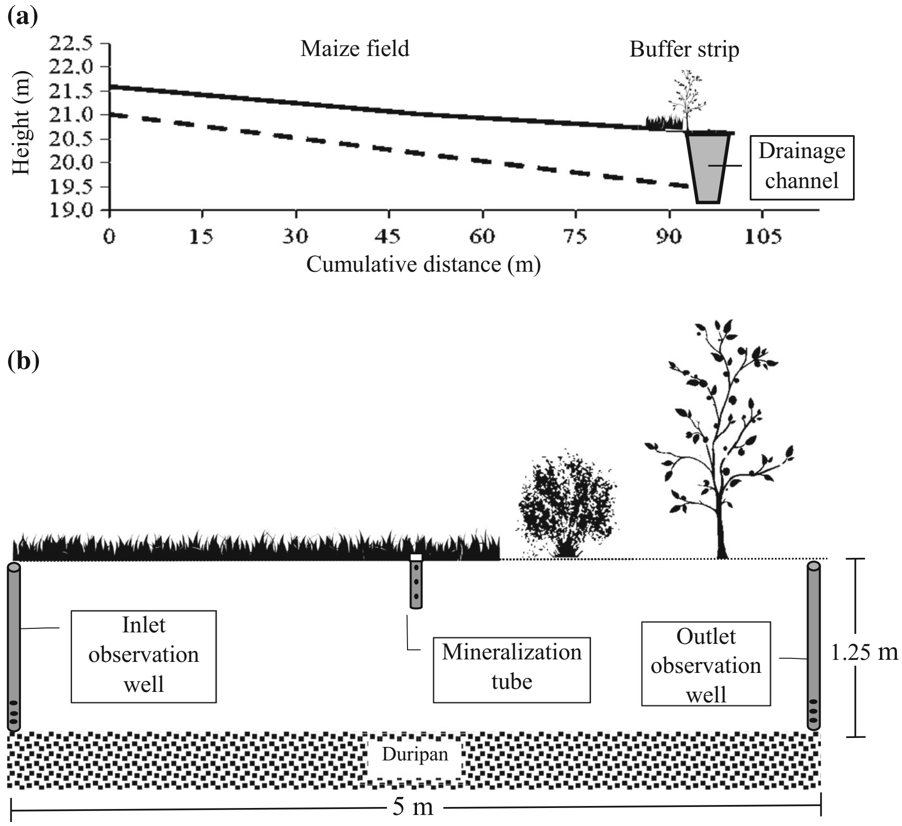

Table 2 Soil properties at the Caleuche site used for the narrow buffer strip experiment

Horizon	Depth (m)	Soil properties							
		Db (mg m^{-3})	Textural class	AWC (%)	SOM (%)	N_T (%)	pH (H_2O)	EC (dS m^{-1})	CEC ($cmol_{(+)}$ kg^{-1})
Ap	0–0.09	1.36	CL	20.47	3.03	0.07	7.21	2.00	14.80
A2	0.09–0.22	1.17	CL	19.87	3.42	0.07	6.85	2.40	10.93
AB	0.22–0.35	1.13	CL	17.63	6.52	0.14	6.42	1.74	17.06
Bt1	0.35–0.51	1.01	CL	16.70	6.27	0.15	6.38	1.73	17.90
Bt2	0.51–0.75	1.20	CL	25.61	4.87	0.09	6.82	1.76	17.61
C	0.75–0.96	1.05	SL	24.66	4.95	0.09	5.94	1.59	14.89
2Cqm	0.96–1.15+	–	–	–	–	–	–	–	–

Db bulk density, *AWC* available water capacity, *SOM* soil organic matter, N_T total nitrogen, *EC* electrical conductivity, *CEC* cation exchange capacity, *CL* clay loam, *SL* silty loam

rate were measured using a single-ring infiltrometer cylinder with 0.50 m diameter (Bouwer 1986). The estimated saturated hydraulic conductivity was 3.1 μm s^{-1}.

Measurements

In each BS, an observation well was installed at the inlet (input) and at the outlet (output) (Fig. 2b). Each well comprised a PVC tube (0.035 m diameter), 1 m long, placed vertically with nine holes (0.01 m diameter) at equal spacings between 1 and 1.25 m soil depth. An extraction tube connected to a syringe was used to collect water samples from each observation well between June 2012 and August 2014. In total, there were 33 samplings per plot, giving a total of 99 input samples and 99 output samples per treatment.

The water samples were chilled on ice in coolers and delivered to the Laboratory of Soil and Water Chemistry at the Faculty of Agricultural Sciences, University of Chile, where they were stored and analysed within 24 h. The samples were filtered using syringe nitrate-free filters (0.45 µm), which gave a clear filtrate in which NO_3–N was measured by the NO_3 chromotropic acid method using a Hach kit (NitraVer® X Reagent Set Cat. No. 26053-45, USA) and NH_3–N was measured by the ammonia-salicylate method using a Hach kit (AmVerTM Nitrogen Ammonia Reagent, USA). Both forms of nitrogen were measured using a UV–Vis spectrophotometer (Hach DR5000, USA).

The N removal effectiveness (%) was calculated as the percentage difference in NO_3–N concentration between the input observation well and the output observation well in the BS, according to Balestrini et al. (2011).

To determine N grass uptake, plant tissue samples were collected twice a year: before the maize cropping period (autumn–winter) and before maize harvest (autumn–winter), using a point quadrat (0.25 m^2) with three replicates. In total, four samplings were carried out during the study period. These plant samples were used to determine total dry mass at 70 °C and total N content (Sadzawka et al. 2004). In addition, plant tissue samples (leaves, trunk and roots) were taken from trees and shrubs for determination of total N content (Sadzawka et al. 2004) at the end of the experiment. To determine N uptake by trees and shrubs, total N content was multiplied by dry mass, estimated using allometric equations developed in a pot experiment (10 kg soil) for each tree and shrub component (n = 12 per plant species).

The covered-cylinder method constructed from PVC was used to assess in situ N mineralisation in undisturbed soil cores every year: (a) in autumn–winter an initial soil sample was collected at 0–0.25 m and two PVC covered-cylinders were installed; (b) and in spring–summer an initial soil sample was collected at 0–0.25 m and another two PVC covered-cylinders were installed. Each PVC covered-cylinder was 0.05 m in diameter and 0.285 m high and was set in the centre of each plot (Fig. 2b). Perforations were added to the sidewall of the tubes to promote soil moisture as well as temperature to equilibrate with the surrounding soil environment, where four holes (0.01 m diameter) were made at 0.1 and 0.2 m from the top. The PVC covered-cylinders were placed upright and pressed lightly to ensure good contact between soil and the underlying soil. Soil was backfilled around the PVC covered-cylinder, so that the level of the soil within PVC covered-cylinders and outside the PVC covered-cylinders was approximately the same (Hanselman et al. 2004). The top of the cylinder was covered by a cap to prevent rainfall or irrigation inputs from promoting N leaching during the in situ incubation period (Dou et al. 1997). The soil cores in the PVC cylinders were removed from the field every 3 months, and all soil samples were analysed for mineral N (N–NO_3 + NH_4) by the KCl extraction and steam distillation method according to Sadzawka et al. (2006). Cumulative net N mineralisation was then determined as the difference between initial and final in situ incubation values (Kolberg et al. 1997). A positive result meant that the process of mineralisation had dominated, while a negative result denoted that immobilisation had exceeded mineralisation.

Water balance

A water balance was calculated to estimate the amount of water leaving the BS (BS_o) using a water balance equation for the BS area over 0–1 m depth, which can be represented as:

$$BS_o = BS_i + PP - \frac{\Delta W}{\Delta t} - Ep - U - DP \tag{2}$$

where BS_i is the water coming from the maize field, PP is the precipitation, W is the amount of water added or removed over 0–1 m depth based on the water content measurements in soil core at 0 month and 29 months, t is the time (29 months), Ep is the crop evapotranspiration rate, U is upflux of shallow groundwater into the rootzone (capillarity), and DP is the deep percolation. The last two variables in Eq. (1) (U and and DP) can be neglected because the presence of the Cqm horizon.

The ΔW was determined on the soil samples collected at 0 month and 29 months after drying at 105 °C for 48 h. Multiplying volumetric water content by the depth interval was used to calculate the water content to a depth basis. The BS_i was estimated from field studies carry out in maize fields in the area (Salazar et al. 2014; Nájera et al. 2015).

Statistical analysis

Statistical comparisons were made between treatments. All data were tested for normality prior to statistical testing. Because of non-normal distribution of dissolved inorganic N concentrations, comparisons were made using the non-parametric Kruskal–Wallis test. When significant differences were found ($p \leq 0.05$), pair-wise comparison of treatments was performed using the non-parametric Tukey test. Analysis of variance (ANOVA) at $p < 0.05$ using a randomised complete block design was used to test the significance of differences between treatments in N uptake and mineralisation/immobilisation. All statistical analyses were performed using Minitab Version 17.0 Software.

Results and discussion

Climate conditions and soil water content

During the study period (29 months), precipitation amounted to 909 mm, which was 34 % lower than mean precipitation according to INIA (1989) in the zone (Fig. 3). In the hydrological years April 2012–March 2013 and April 2013–March 2014, precipitation was 379 and 196 mm, respectively, with 74–94 % of the total amount falling during autumn–winter.

A simple water balance using Eq. (1) was calculate as is shown in Table 3. In this balance there were two inputs to the BS: the precipitation and the water coming from the maize field (surface runoff plus subsurface lateral flow). During spring–summer season, the poorly efficient surface irrigation system in the maize crop generated surface runoff and subsurface lateral flow that supplied the BS system with around 1000 mm per cropping season (Salazar et al. 2014), which was the major water input to the BS system during the study period. In addition, there were two outputs: the evapotranspiration (*Ep*) and the water leaving the BS (surface runoff plus subsurface lateral flow). The calculated *Ep* in the grass BS was 170 mm during autumn–winter and 320 mm during spring–summer. Unlike bare soil, in vegetated BS the *Ep* can reduce the subsurface flow rate, thus helping to prevent removal of NO_3–N from the groundwater. The water content sensors in the treatments indicated that the soil profile was saturated for most of the time, with the water content at 1 m during the study period ranging from 0.47 to 0.52 $m^3\ m^{-3}$ (data not shown). These results confirmed that there was a continue subsurface lateral flow from the maize field to the BS during the study period.

Dissolved inorganic N forms in subsurface lateral flow

In general, the NO_3–N concentrations in lateral subsurface from maize fields were reduced after passage through all vegetated BS treatments (Table 4). The NO_3–N concentrations in the G treatment showed a significant concentration reduction ($p < 0.05$) in comparison with the control (C). Vegetated BS (G, GS, GST1 and GST2) had a NO_3–N removal efficiency ranging from 33 to 67 %, with a mean value of 52 %. Unlike vegetated BS treatments, in the bare soil (C) the NO_3–N concentrations increased by 24 %, showing the lowest removal efficiency and significant differences ($p < 0.05$) with the BS treatments. No significant differences ($p > 0.05$) in NH_3–N concentration were detected between treatments and NH_3–N removal efficiency of treatments with trees (GST1 and GST2) showed a higher NH_3–N retention capacity ($p < 0.05$) than the other treatments.

During the crop growing season, mean NO_3–N input concentration to the treatments ranged from 3.6 to 18 mg NO_3–N L^{-1}, with a maximum value of 66.6 mg NO_3–N L^{-1}. This was due to the large N surplus from high N fertilisation rates and irrigation events, when most water (>45 %) is lost due to surface runoff and deep percolation. During the fallow season, mean NO_3–N input concentrations in the treatments were lower than during the growing season, with values ranging from 2.2 to 6.5 mg NO_3–N L^{-1} (maximum 33.0 mg NO_3–N L^{-1}). These results suggest that most of the surplus N could have been leached by excessive irrigation during the crop growing season (spring–summer), while a lower

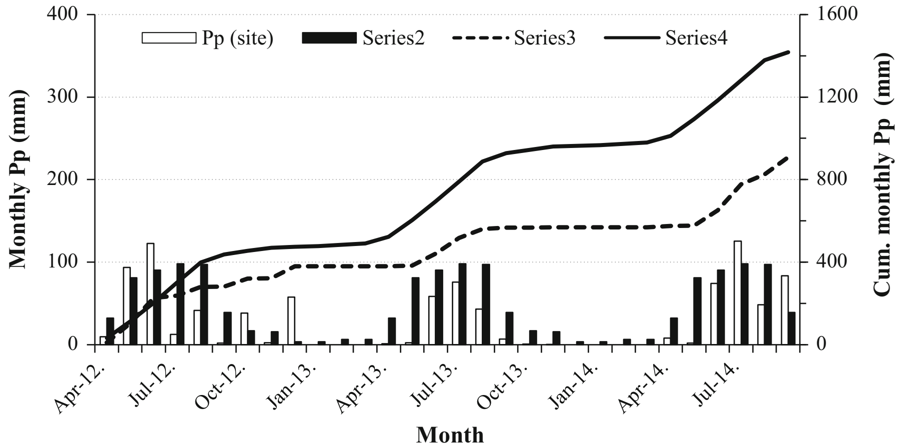

Fig. 3 Monthly and cumulative (Cum.) precipitation (Pp) at the field site during the study period (April 2012–August 2014), and historical mean monthly and cumulative precipitation in the zone (INIA 1989)

Table 3 Mean annual water balance in the different treatments during the study period

Season/treatment[a]	Water balance[b] (mm $year^{-1}$)			
	BS_i	PP	Ep	BS_o
Autumn–winter				
C	115	295	93	231
G	115	295	170	154
GS	115	295	175	149
GST1	115	295	196	129
GST2	115	295	191	134
Spring–summer				
C	1.045	12	129	990
G	1.045	12	321	797
GS	1.045	12	334	785
GST1	1.045	12	386	733
GST2	1.045	12	373	746

[a] For treatment details see Table 1

[b] Calculated according Eq. (2)

amount of residual N may still be present in the soil in autumn–winter (Salazar et al. 2014).

The output NO_3–N concentrations in subsurface lateral flow in the treatments during the study period are shown in Fig. 4. During the crop growing seasons, mean NO_3–N output concentrations in vegetated BS systems ranged from 1.5 to 4.1 mg NO_3–N L^{-1}, with a maximum value of 30.1 mg NO_3–N L^{-1}. The NO_3–N output concentrations in the bare soil control (C) showed a similar trend. During the fallow season, mean NO_3–N output concentrations in vegetated BS systems ranged between 2.0 and 5.9 mg NO_3–N L^{-1} (maximum 29.8.0 mg NO_3–N L^{-1}). In contrast, during the fallow season bare soil (C) showed higher mean NO_3–N output concentrations (8.6 mg NO_3–N L^{-1}) and maximum values (58.6 mg NO_3–N L^{-1}). Compared to the vegetated BS, the higher NO_3–N concentrations in the bare soil during the fallow season were related to the higher amount of water leaving the T0 (Table 3). Clearly the BS reduce available water volume and in consequence the subsurface water flow because vegetation consumes water through transpiration (Brauman et al. 2007). It was found that the vegetated BS treatments were able to retain N forms moving from agricultural soil to nearby surface waters during intensive precipitation events in the fallow season.

The NO_3–N concentrations were above standard drinking water limits (10 mg NO_3–N L^{-1}) in 17 and 9 % of the total input and output samples, respectively. Although these represent a small proportion of the total samples, some high NO_3–N concentrations (>50 mg NO_3–N L^{-1}) which could pose a particular risk in human consumption were recorded.

During the crop growing season, mean NH_3–N input concentrations to the BS systems ranged from 1.8 to 4.5 mg NH_3–N L^{-1}, with a maximum value of 15.5 NH_3–N L^{-1}. Similarly, during the fallow season mean NO_3–N input concentrations in BS systems ranged between 1.2 and 2.5 mg NH_3–N L^{-1}, with a maximum value of 15.0 mg NH_3–N L^{-1}. The output NH_3–N concentrations in subsurface lateral flow in the treatments during the study period are shown in Fig. 5. During the crop growing and fallow seasons, mean NH_3–N output concentrations in the BS systems showed high variability, ranging between 0.6 and 8.3 mg NH_3–N L^{-1} (maximum 22.4 NH_3–N L^{-1}).

Table 4 Mean input and output values of nitrate–nitrogen (NO_3–N) and ammonia–nitrogen (NH_3–N) concentrations in the different treatments

Treatment[a]	NO_3–N[b]			NH_3–N[b]		
	Input (mg NO_3–N L^{-1})	Output (mg NO_3–N L^{-1})	Removal (%)	Input (mg NO_3–N L^{-1})	Output (mg NO_3–N L^{-1})	Removal (%)
C	5.33a	6.63a	−24d	1.98a	3.86a	−95c
G	5.56a	1.85b	67a	2.72a	2.90a	−7b
GS	5.77a	2.93ab	49b	2.02a	2.58a	−28b
GST1	8.05a	5.41ab	33c	3.36a	0.65a	81a
GST2	6.45a	2.61ab	60ab	1.62a	0.56a	65a

[a] For treatment details see Table 1

[b] Means (n = 99) within columns with different letters are significantly different (Kruskal–Wallis test, $p < 0.05$)

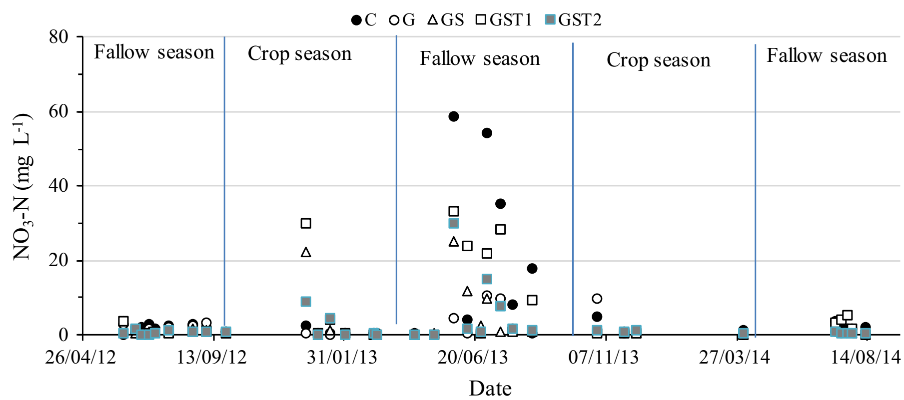

Fig. 4 Nitrate–N (NO_3–N) concentrations in subsurface lateral flow at outlet observation wells in the treatments during the study period

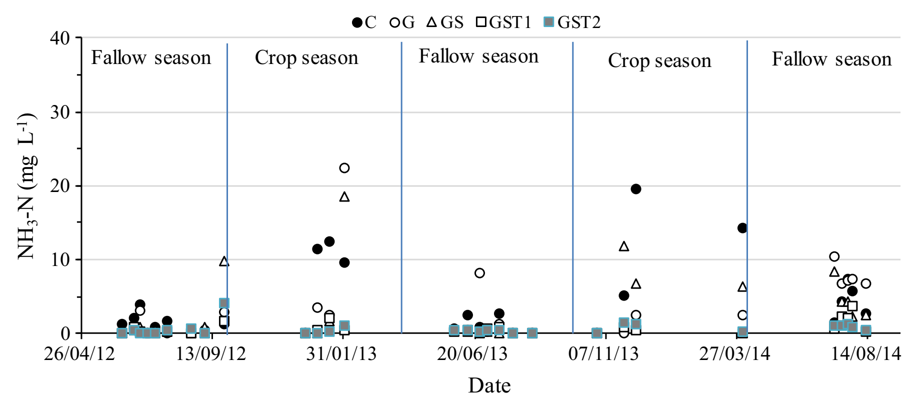

Fig. 5 Ammonia–N (NH_3–N) concentrations in subsurface lateral flow at outlet observation wells in the treatments during the study period

Effectiveness of narrow buffer strips in removing inorganic N forms

The results of this study corroborate previous findings that narrow vegetated BS can retain NO_3–N associated with subsurface lateral flow within 3–5 years (Yamada et al. 2007; van Beek et al. 2007). However, the BS treatments did not have an impact in reducing NH_3–N concentrations, which may be related to the continuing equilibrium between NH_3–N and NH_4–N in groundwater and conversion of the latter to NO_3–N.

There are three main N process related to NO_3–N retention capacity in vegetated BS treatments: uptake, immobilisation and denitrification. Below, we discuss these different NO_3–N processes in relation to the functioning of the vegetated BS.

According to N uptake measurements, the shrubs and trees used were too small to have a detectable impact on N uptake (<1 kg N ha^{-1}). Therefore, N uptake was mainly attributable to the grass component. Similar results were found by Borin and Bigon (2002) in a study on narrow buffer strips (5 m width) with one line of trees. The mean N uptake by the grass component in our BS treatments in shown in Table 5. It is important to note that around 70 % of the N was taken up during spring–summer because the grass had better growing conditions, such as high temperature, solar radiation, etc. The G treatment, in which the BS were completely covered by grass, clearly had higher N plant uptake capacity than treatments GS, GST1 and GST2, which had 80, 60 and 60 % grass cover, respectively. This may explain the significant differences between G and C in the output NO_3–N concentrations in lateral subsurface flow, as 1.1 kg NO_3–N $plot^{-1}$ $year^{-1}$ was assimilated by the grass component. It is important to note that the G treatment showed a mean decrease of 3.7 mg NO_3–N L^{-1} between input and output concentrations (Table 4), which corresponds to about 0.1 kg NO_3–N $plot^{-1}$ $year^{-1}$ [soil depth = 1 m; $\varphi = 0.52\ m^3\ m^{-3}$ (effective porosity)]. Therefore, grass N uptake may explain the N retention capacity of the BS treatments.

The results on in situ net N mineralisation in the BS treatments showed that N immobilisation was dominant during autumn–winter, while N mineralisation was dominant during spring–summer (Table 6). Overall, N mineralisation was the dominant process during the year, particularly in the study soil with its high SOM content (3–6 %). It is important to note that if there is not a cover crop present that assimilates the available N generated by N mineralisation on the soil, there would be permanent N diffuse pollution to the drainage channel at the site. This may partly explain the increase in NO_3–N output concentrations in the bare soil (C) treatment. Thus the cover crop N stock, unless it is removed by harvesting, can be released back to BS via decomposition of plant litter. Therefore, periodic harvesting of cover crop could ensure plant uptake remains a continued net nutrient removal mechanism (Hefting et al. 2005).

It is also possible that denitrification was associated with the NO_3–N retention capacity of the vegetated BS treatments, as reported in other studies (Cors and Tychon 2007; van Beek et al. 2007). For instance, Borin and Bigon (2002) noted that the N retention observed in BS during winter, when plant uptake is negligible, could be the result of denitrification. In addition, Balestrini et al. (2011) noted that in BS it is important to consider the indirect effect of vegetation on denitrification through the release of organic matter to the soil. In our study, the water sensors indicated that most of the time the soil at 1 m depth was saturated, resulting in anaerobic conditions. Moreover, the subsoil had a high MOS content (MOS = 5 %), both soil conditions suitable for denitrifying bacteria. However, measurements of denitrification values would be necessary to confirm this N loss.

Table 5 Mean nitrogen (N) uptake by grass component per year in buffer strip (BS) treatments

Treatment	N uptake[a]	
	kg N ha^{-1}	kg N m of BS^{-1}
C	–	–
G	123a	1.2a
GS	60b	0.6b
GST1	57b	0.6b
GST2	68b	0.7b

[a] Values within columns with different letters are significantly different (ANOVA test, $p < 0.05$)

Table 6 Mean nitrogen (N) mineralisation during autumn–winter and spring–summer in the buffer strip treatments

Treatment	N mineralisation[a] (kg N ha^{-1})	
	Autumn–winter	Spring–summer
C	−16	115
G	2	119
GS	−74	124
GST1	2	143
GST2	−40	138

[a] Differences between treatments were not statistically significant (ANOVA test, $p > 0.05$)

Conclusions

Narrow buffer strips (BS) covered by permanent grass (G treatment) reduced the concentration of NO_3–N in subsurface lateral flow in the Mediterranean agroecosystem studied here during the 29-month study period. However, the BS did not have an impact in reducing NH_3–N concentrations. The permanent grass BS system acted as a large NO_3–N sink in a short time scale, even during intensive precipitation events in autumn–winter, when the BS through transpiration reduces the subsurface water flow and in consequence the N losses. Nitrogen uptake by grass was an important process in the N retention capacity of the BS, whereas young trees and shrubs showed limited N uptake in the same period, i.e. they functioned as minor sinks and only played a limited role in mitigating NO_3–N pollution. Measurements of in situ net N mineralisation in the BS treatments showed that N immobilisation was dominant during autumn–winter, while N mineralisation was dominant during spring–summer.

Acknowledgments The authors thank the Department of Soil and Engineering at the University of Chile for supporting this study. We also thank Norma Sepulveda and Jorge Farias, who helped in sampling and laboratory analysis. This research was partially funded by FONDECYT de Iniciación 2011 Grant No. 11110464. This study was funded by FONDECYT de Iniciación 2011 Project No. 11110464.

Conflict of interest The authors declare that they have no conflict of interest.

References

Allen RG, Pereira LS, Raes D, Smith M (1998) Crop evapotranspiration—guidelines for computing crop water requirements—FAO irrigation and drainage paper 56. FAO—Food and Agriculture Organization of the United Nations, Rome

Anderson DM, Glibert PM, Burkholder JM (2002) Harmful algal blooms and eutrophication: nutrient sources, composition, and consequences. Estuaries 25:704–726

Balestrini R, Arese C, Delconte CA, Lotti A, Salerno F (2011) Nitrogen removal in subsurface water by narrow buffer strips in the intensive farming landscape of the Po River watershed, Italy. Ecol Eng 37:148–157

Berenguer P, Santiveri F, Boixadera J, Lloveras J (2009) Nitrogen fertilisation of irrigated maize under Mediterranean conditions. Eur J Agron 30:163–171

Borin M, Bigon E (2002) Abatement of NO_3–N concentration in agricultural waters by narrow buffer strips. Environ Pollut 117:165–168

Borin M, Vianello M, Morari F, Zanin G (2005) Effectiveness of buffer strips in removing pollutants in runoff from a cultivated field in North-East Italy. Agric Ecosyst Environ 105:101–114

Borin M, Passoni M, Thiene M, Tempest T (2010) Multiple functions of buffer strips in farming areas. Eur J Agron 32:103–111

Bouwer H (1986) Intake rate: cylinder infiltrometer. In: Klute A (ed) Methods of soil analysis, vol 1, 2[a] edn. ASA and SSSA, Madison, pp 825–844

Brauman KA, Daily GC, Duarte TK, Mooney HA (2007) The nature and value of ecosystem services: an overview highlighting hydrologic services. Annu Rev Environ Resour 32:67–98

Burt TP, Haycock NE (1993) Controlling losses of nitrate by changing land use. In: Burt TP, Heathwaite AL, Trudgill ST (eds) Nitrate-processes, patterns and management. Wiley, UK, pp 341–368

Casanova M, Seguel O, Salazar O, Luzio W (2013) Soils of Chile. Soils of the world soils series. Springer, Germany

Cors M, Tychon B (2007) Grassed buffer strips as nitrate diffuse pollution remediation tools: management impact on the denitrification enzyme activity. Water Sci Technol 55:25–30

Doney SC (2010) The growing human footprint on coastal and open-ocean biogeochemistry. Science 328:1512–1516

Dou H, Alva AK, Khakural BR (1997) Nitrogen mineralization from citrus tree residues under different production conditions. Soil Sci Soc Am J 61:1226–1232

Dunn AM, Julien G, Ernst WR, Cook A, Doe KG, Jackman PM (2011) Evaluation of buffer zone effectiveness in mitigating the risks associated with agricultural runoff in Prince Edward Island. Sci Total Environ 409:868–882

Fuentes I, Casanova M, Seguel O, Nájera F, Salazar O (2014) Morpho-physical pedotransfer functions for groundwater pollution by nitrate leaching in Central Chile. Chil J Agric Res 74:340–348

Fuentes I, Casanova M, Seguel O, Padarian J, Nájera F, Salazar O (2015) Preferential flow paths in two alluvial soils with long-term pig slurry additions in the Mediterranean zone in Chile. Soil Res. doi:10.1071/SR14264

Gabriel JL, Muñoz-Carpena R, Quemada M (2012) The role of cover crops in irrigated systems: water balance, nitrate leaching and soil mineral nitrogen accumulation. Agric Ecosyst Environ 155:50–61

Galloway JN, Townsend AR, Erisman JW, Bekunda M, Cai Z, Freney JR, Martinelli LA, Seitzinger SP, Sutton MA (2008) Transformation of the nitrogen cycle: recent trends, questions, and potential solutions. Science 320:889–892

Hanselman TA, Graetz DA, Obreza TA (2004) A comparison of in situ methods for measuring net nitrogen mineralization rates of organic soil amendments. J Environ Qual 33:1098–1105

Hefting MM, Clement J, Bienkowski P, Dowrick D, Guenat C, Butturini A, Topa S, Pinay G, Verhoeven JTA (2005) The role of vegetation and litter in the nitrogen dynamics of riparian buffer zones in Europe. Ecol Eng 24:465–482

INIA (1989) Mapa Agroclimático de Chile (Agroclimatic Map of Chile). Instituto de Investigaciones Agropecuarias (INIA), Santiago, Chile (in Spanish)

Kolberg RL, Rouppet B, Westfall DG, Peterson GA (1997) Evaluation of an in situ net soil nitrogen mineralization in dryland agroecosystems. Soil Sci Soc Am J 61:504–508

Larson RA, Safferman SI (2012) Field application of farmstead runoff to vegetated filter strips: surface and subsurface water quality assessment. J Environ Qual 41:592–603

Mayer PM, Reynolds SK, McCutchen MD, Canfield TJ (2006) Riparian buffer width, vegetative cover, and nitrogen removal effectiveness: A review of current science and regulations. EPA/600/R-05/118. Environmental Protection Agency, Cincinnati, USA

Nájera F, Tapia Y, Baginsky C, Figueroa V, Cabeza R, Salazar O (2015) Evaluation of soil fertility and fertilisation practices for irrigated maize (*Zea mays* L.) under Mediterranean conditions in central Chile. J Soil Sci Plant Nutr 15:84–97

Noij IGAM, Heinen M, Heesmans HIM, Thissen JTNM, Groenendijk P (2012) Effectiveness of unfertilized buffer strips for reducing nitrogen loads from agricultural lowland to surface waters. J Environ Qual 41:322–333

Peel MC, Finlayson BL, McMahon TA (2007) Updated world map of the Köppen–Geiger climate classification. Hydrol Earth Syst Sci 11:1633–1644

Prokopy LS, Floress K, Klotthor-Weinkauf D, Baumgart-Getz A (2008) Determinants of agricultural best management practice adoption: evidence from the literature. J Soil Water Conserv 63:300–311

Quemada M, Baranski M, de Lange MNJ, Vallejo A, Cooper JC (2013) Meta-analysis of strategies to control nitrate leaching in irrigated agricultural systems and their effects on crop yield. Agric Ecosys Environ 174:1–10

Robertson GP, Vitousek PM (2009) Nitrogen in agriculture: balancing the cost of an essential resource. Annu Rev Environ Resour 34:97–125

Sadzawka A, Grez R, Carrasco MA, Mora M (2004) Métodos recomendados para tejidos vegetales (Recommended Methods for Vegetal Tissues). Comisión Nacional de Normalización y Acreditación, Sociedad Chilena de la Ciencia del Suelo, Chile (in Spanish)

Sadzawka A, Grez R, Carrasco MA, Mora ML (2006) Métodos Recomendados para Suelos Chilenos (Recommended Methods for Chilean Soils). Comisión Nacional de Normalización y Acreditación, Sociedad Chilena de la Ciencia del suelo, Santiago, Chile (in Spanish)

Salazar O, Wesström I, Joel A, Youssef MA (2013a) Application of an integrated framework for estimating nitrate loads from a coastal watershed in southeast Sweden. Agric Water Manag 129:56–68

Salazar O, Hansen S, Abrahamsen P, Hansen K, Gundersen P (2013b) Changes in soil water balance following afforestation of former arable soils in Denmark as evaluated using the DAISY model. J Hydrol 484:128–139

Salazar O, Vargas J, Nájera F, Seguel O, Casanova M (2014) Monitoring of nitrate leaching during flush flooding events in a coarse-textured floodplain soil. Agr Water Manag 146:218–227

Salmerón M, Isla R, Cavero J (2011) Effect of winter cover crop species and planting methods on maize yield and N availability under irrigated Mediterranean conditions. Field Crop Res 123:89–99

Sandoval M, Dörner J, Seguel O, Cuevas J, Rivera D (2012) Métodos de Análisis Físicos de Suelos (Methods for soil physical analyses). Departamento de Suelos y Recursos Naturales, Universidad de Concepción. Publicación no. 5, Chile (in Spanish)

Vahtera E, Conley DJ, Gustafsson BG, Kuosa H, Pitkänen H, Savchunk OP, Tamminen T, Viitasalo M, Voss M, Wasmund N, Wulff F (2007) Internal ecosystem feedbacks enhance nitrogen-fixing cyanobacteria blooms and complicate management in the Baltic Sea. Ambio 36:186–194

van Beek CL, Heinen M, Clevering OA (2007) Reduced nitrate concentrations in shallow ground water under a non-fertilised grass buffer strip. Nutr Cycl Agroecosyst 79:81–91

Vitousek PM, Aber J, Howarth RW, Likens GE, Matson PA, Schindler DW, Schlesinger WH, Tilman GD (1997) Human alteration of the global nitrogen cycle: causes and consequences. Issues Ecol 1:1–15

Wang L, Duggin JA, Nie D (2012) Nitrate–nitrogen reduction by established tree and pasture buffer strips associated with a cattle feedlot effluent disposal area near Armidale, NSW Australia. J Environ Manag 99:1–9

Ward MH, deKok TM, Levallois P, Brender J, Gulis G, Nolan BT, VanDerslice J (2005) Workgroup Report: drinking-water nitrate an health-recent findings and research needs. Environ Health Perspect 113:1607–1614

Webber DF, Mickelson SK, Wulf LW, Richard TL, Ahn HK (2010) Hydrologic modeling of runoff from a livestock manure windrow composting site with a fly ash pad surface and vegetative filter strip buffers. J Soil Water Conserv 65:252–260

Yamada T, Logsdon SD, Tomer MD, Burkart MR (2007) Groundwater nitrate following installation of a vegetated riparian buffer. Sci Total Environ 385:297–309