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Creating wetlands for the improvement of water quality and landscape restoration in semi-arid zones degraded by intensive agricultural use

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ABSTRACT

Increasing interest in restoring wetlands within a multipurpose approach is observed in degraded lands submitted to intensive human uses. This study evaluates the effectiveness of constructed and natural wetlands in removing nutrients from agricultural wastewater and their potential contribution to landscape heterogeneity in semiarid Monegros area, NE Spain. To achieve the first aim, wetland plots of differing sizes (50, 200, 800 m^2) were constructed upon fields abandoned four years earlier. Water has been sampled at the inflow and outflow of the plots for two years. Results show a 24–43% rate of total nitrogen removal and no clear trend for phosphorus in constructed wetlands. Slight effectiveness improvements took place in the second working year and in large-size plots. For the second goal, a number of catchments with wetlands that originated as a consequence of irrigation were selected. These wetlands do not contribute significantly to improving the landscape diversity of agricultural catchments. Based on this experimental work, it is estimated that it should be necessary to restore wetlands in 3.25 and 5.60% of total watershed areas to remove most nitrogen from wastewater. Their restoration should be implemented also with the objective of increasing the landscape diversity of zones extensively transformed and homogenized by agricultural practices.

1. Introduction

Wetland restoration should integrate objectives across spatial scales in order to cover the range of natural interactions (Mitsch and Gosselink, 2000; Zedler, 2000). Wetland construction is mostly focused on water quality improvement, although there is an increasing scientific interest in multipurpose approaches (Wheeler, 1995; Comín et al., 2001).

The capacity of constructed wetlands to remove nutrients from agricultural runoff has evolved mostly based on their nutrient removal efficiencies and hydraulic characteristics (Brix and Schierup, 1989; Hammer, 1992; Kadlec and Knight, 1996; Romero et al., 1999). New interest is arising now in restoring and creating wetlands to buffer non-point source pollution at watershed scale (Raisen and Mitchell, 1995; Mitsch and Gosselink, 2000) because nutrients are responsible for eutrophication of natural aquatic ecosystems in river basins and coastal seas (Goldman and Horne, 1983). Nutrient retention differs depending on factors such as nutrient concentration, seasonality, hydraulic loading, water-residence time, soil type, plant species, and water chemistry. Results for nitrogen removal used to be as high as 99% of the inflow concentration, whereas results for phosphorus are very variable (Zurayk et al., 1997; Mitsch and Gosselink, 2000; Braskerud, 2002a,b; Callaway et al., 2003; Kadlec, 2003; Koskiaho et al., 2003; Bruland and Richardson, 2004; Fink and Mitsch, 2004). Increasing residence time favors nutrients removal (Kadlec and Knight, 1996). However, it is well known that reaching

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effective nutrient removal may take a few growing seasons because of lack of well developed below- and above-ground plant–microbial interactions (Mitsch and Jørgensen, 2004). This is a particularly critical point in dry environments where lack of water may limit the hydrological requirements of biogeochemical processes removing nutrients in wetlands (Mandi et al., 1998; Lissner et al., 1999b; Comín et al., 2005).

Few experiments have been carried out with constructed wetlands in semiarid zones compared to humid ones (Cerezo et al., 2001; Howell et al., 2005). High temperature and radiation during the growing season are characteristic of semiarid climates, affecting microbial activity and plant productivity, directly or indirectly (e.g., through increased salinity). The relationship between salinity and wetland functioning is a major gap in ecological research. The effects of salinity increase on *Phragmites australis* were studied by Lissner et al. (1999) in experimental conditions, allowing relatively good plant growth up to 10 psu (practical salinity units).

Improvement of landscape diversity has also been addressed by constructing wetlands (Knight, 1992; Comín et al., 2001). Thus, wetlands should be restored and constructed with a suitable size ensuring their integration at landscape scale. Several contributions have been reported providing guidelines for planning at catchment scale. Hammer (1992) recommended choosing an acceptable 2% of the catchment area for wetlands establishment to virtually treat all runoff. Equally, Larson et al. (2000) showed a design that intercepts agricultural drainage with a watershed to wetland ratio of 22:1, meaning that approximately 4.7% of the catchment was covered by wetlands. Finally, it has been suggested that a sufficient improvement of the water quality would be achieved returning between 3 and 5% of the Mississippi catchment area to wetlands (Mitsch et al., 2001). Additionally, modeling incorporates different criteria for restoration suitability which provides a variety of results for wetland restoration potential at watershed scale depending on the selected and weighted variables (White and Fennessy, 2005). Moreover, it has been illustrated that wetland functioning is more effective in upper reaches of a catchment than in lower ones in spite of the fact that nutrient concentration of runoff treated is lower in the first (Prato et al., 1995; Woltemade, 2000).

Large areas of the central Ebro river valley (NE Spain) were transformed into irrigated agricultural fields during the 1950s–1990s causing soil and water salinization and landscape homogenization (Tedeschi et al., 2001; Causapé et al., 2004). Nowadays, there is a social demand for increasing landscape diversity, enhancing habitat quality for wildlife, and improving river water quality. Accordingly, we established two goals for the present study: estimating the efficiency of newly constructed wetlands on old irrigated fields to remove nitrogen and phosphorus from agricultural wastewater and determining their potential contribution to increase landscape heterogeneity.

2. Study area

Monegros is a 270,000 ha region located in the central part of the Ebro River basin, NE Spain (Fig. 1). Climate is semiarid and Mediterranean–continental. Average annual temperature is 14.5 °C; average annual precipitation is 400 mm (Pedrocchi, 1998) with a high interannual variability (Comín and Williams, 1993). Soils are mainly dominated by a Tertiary structure composed of clays with different salinization levels (conductivity ranges between 1 and 10 mS cm^{-1}) with inserted sandstone stratus. Locally, this structure is covered by gravels 2–4 m wide. Most of this land was transformed into irrigated agricultural fields during the 1950s–1990s. Soil salinization and abandonment of agriculture is now widespread in many parts of this region. The original pseudo-steppe vegetation, dominated by Rosemary (*Rosmarinus officinalis*), Thyme (*Thymus* sp.) and halophytes (*Salicornia* sp.) as well as perennial grasses (*Lygeum spartium*, *Brachypodium retusum*) changed into a landscape dominated by new halophytic species (*Tamarix africana*, *Atriplex* sp., *Suaeda vera*) in abandoned dry elevated zones and helophytes (*P. australis*, *Typha latifolia*, *Scirpus holoschoenus*, *S. maritimus*, *Carex divisa*) in the abandoned wetter and lower zones of the valleys. Most of these wetlands are colonized by *P. australis*, which is spreading and increasingly dominating the plant community because of its tolerance to changing water level and salinity (Lissner et al., 1999a). *Phragmites*-dominated wetlands exist in the lower parts of non-irrigated valleys in the study area; this indicates that this type of wetlands are spontaneously formed at the hydrological discharge zones of the valleys.

In the study area, 19 catchments were selected for the landscape survey. An abandoned field (1.92 ha) that was irrigated in the past was used to establish experimental wetland plots (Fig. 1). They were banked and flooded with wastewater from the surrounding agricultural irrigated fields to study their effectiveness for water quality improvement.

3. Material and methods

3.1. *Constructed wetlands*

Nine experimental plots of different size (small: 5 m × 10 m; medium: 10 m × 20 m; large 20 m × 40 m) were established in an abandoned agricultural field and they were operative during the growing seasons 2004–2005 (Fig. 1). Plots were essentially covered by *P. australis* (with scattered stems of *T. latifolia*, *Scirpus lacustris* and *C. divisa*) spontaneously grown since field abandonment four years earlier. They persisted thanks to continuous water contributions from some leaks of a nearby ditch. These leaks were repaired before plot conditioning. Wetlands showed a density of *P. australis* of 165.15 ± 115.57 stems/m^2, height of 152.32 ± 56.34 cm and aboveground biomass of 1.58 ± 1.19 kg/m^2. All plots were kept flooded as long as possible with the same agricultural wastewater that was collected in a ditch with a dike at 100 m distance and conveyed separately to each plot by a distribution pipe (Fig. 1). Wastewater came from soil infiltration of irrigated land in the surrounding fields. Samples of the agricultural wastewater were collected before entering the experimental plots to analyze main water characteristics (Table 1). These characteristics were determined in the same way as those colleted at inflow and outflow of the plot. Inflowing wastewater was regulated in each plot by one to three valves in order to

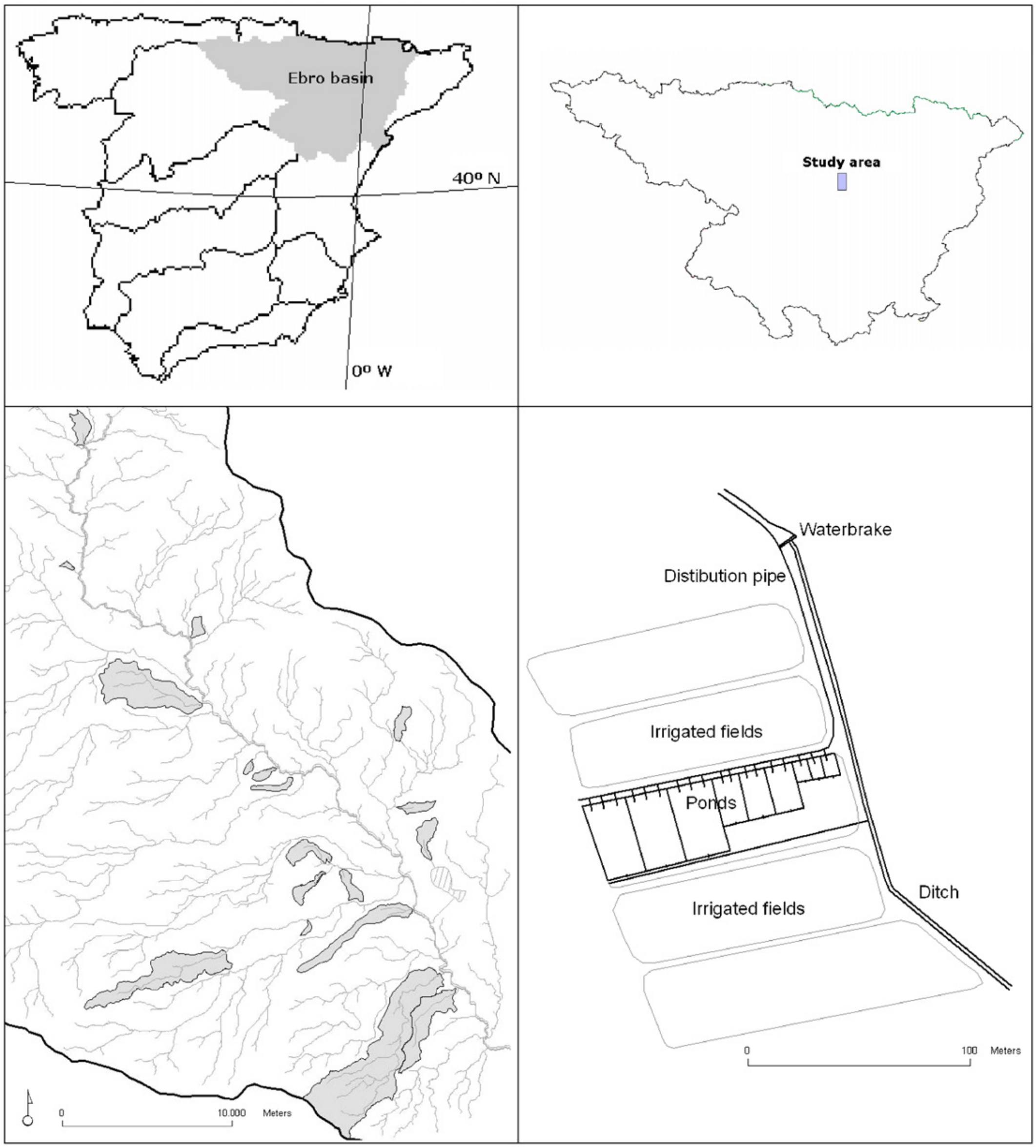

Fig. 1 – Map of the study area with the main catchments of the Spanish Iberian Peninsula (top left), study area located in the Ebro basin (top right), selected catchments (in grey) included in the Flumen basin and its drainage network (bottom left) and plan of the experimental wetlands (bottom right).

avoid flow concentration (Reed et al., 1995). Water depth was fixed at 10 cm allowing a water-residence time between one and four days. Nevertheless, the climatic conditions and the lack of wastewater made often difficult to maintain a constant water level and water turnover. Water flow was estimated after the time needed to fill a container of known volume. Since the hydrologic regime depended on watering periods, it was difficult to completely provide a constant flow to the experimental plots.

Samples of inflowing and outflowing water were collected in all the plots three times in 2004 and four times in 2005 during the growing season. In situ field measurements of temperature, pH, dissolved oxygen (DO), and electrical conductivity (EC) were performed with calibrated electronic apparati. Water samples were filtered (0.8 μm mesh size pre-combusted filters) the same day of sampling, and total dissolved solids (TDS), total suspended solids (TSS), turbidity, alkalinity, Cl^-, Ca^{2+}, Mg^{2+}, Na^+, and K^+ were measured using standard methods in the laboratory (APHA, 1998) Micronutrients were analyzed from frozen (at $-30\,^{\circ}$C) sample aliquots one month after sampling using standard methods (APHA, 1998). Soil salinity was estimated as EC of a saturated paste made of 1:5 proportion of soil and distilled water. ANOVA and Kruskal–Wallis tests were used to test for statistical differences between inflow and outflow water characteristics, and for differences of nutrient retention between plots of different size.

3.2. *Existing wetlands*

All wetlands larger than one hectare located in catchments of irrigated farms of the Flumen River and included in Monegros

Table 1 – Characteristics of the wastewater running off the surrounding irrigated fields at the wetlands inflow

	Maximum	Minimum
pH	8.21	7.51
EC (μS)	1165.00	498.00
DO (mg/l)	11.89	5.58
TSS (μg/l)	3.20	1.07
TDS (mg/l)	772.00	544.00
Chloride (mg/l)	176.00	80.80
Sulphates (mg/l)	123.31	47.20
Ca^{2+} (mg/l)	140.90	73.20
Mg^{2+} (mg/l)	27.00	17.90
Na^{+} (mg/l)	226.25	52.03
K^{+} (mg/l)	2.04	0.00
Turbidity (mg Pt/l)	54.00	3.00
Alkalinity (mg $CaCO_3$/l)	362.40	259.10
NO_3–N (mg/l)	20.65	5.80
NO_2–N (μg/l)	61.71	0.00
Ammonium (μg/l)	190.21	16.99
Organic N (mg/l)	16.12	0.00
TN (mg/l)	22.82	16.11
SRP-P (μg/l)	98.87	3.17
Organic P (μg/l)	177.89	34.91
TP (μg/l)	276.76	38.93

Maximum and minimum values are given only because of the high variability of the data along the year.

area were selected. In addition to them, six more catchments with wetlands smaller than one hectare and five more with no representative wetland surfaces were also selected for study, i.e. 19 (in all). The criteria followed to select these additional catchments were the topographical, geological, and geographical similarity to those with large wetlands (Figs. 1 and 2). All 19 catchments were digitalized in ArcGIS 8.3 (ESRI, Inc.) with aerial photographs 0.5 m resolution taken in 2003. Land cover types in the photographs were checked in situ by direct observation and their areas estimated using ArcGIS 8.3. The considered land cover types were arboreal vegetation, dry scrublands–grasslands, erosion deserts, urbanized areas, dry farmlands, irrigated farmlands, abandoned farms, arboreal cultivations, wetlands, channels, livestock farms, and irrigation ponds Then, landscape spectral diversity was estimated

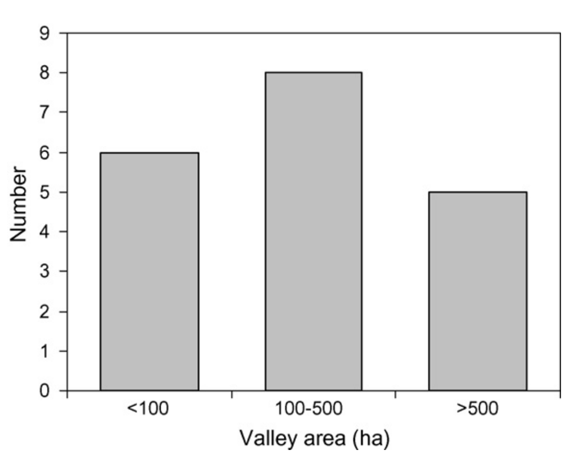

Fig. 2 – Number of selected valleys studied for landscape characteristics sorted in three size groups.

using homogeneous segments of 6 ha (300 m × 200 m) distributed along the major geographical axis of the catchment. We used the Shannon diversity index ($H = -\sum p_i \log_{10} p_i$, where p_i is the area of landscape type i/total segment area) to estimate landscape heterogeneity in each catchment. Spectral landscape diversity was calculated by accumulating areas along each transect. Obtained diversity spectra of catchments were grouped by sizes in the same way as done in Fig. 2. This calculation gives us information about the spatial heterogeneity of the landscape diversity (Margalef, 1974; Turner et al., 1989; Comín et al., 2001).

4. Results

4.1. *Water quality*

The water entering the experimental wetlands was fresh with relatively high dissolved salt content and alkalinity. It was typical agricultural wastewater, with nitrate and phosphate concentrations over those advised by European Directives (European Communities, 2000).

A clear difference between nitrogen concentration at wetland inflow and outflow was observed (Fig. 3). Total nitrogen (TN) decreased from 16–19 mg/l in the inflowing water to 12–16 mg/l in the outflowing water. Most of TN (78–90%) was dissolved inorganic nitrogen in form of nitrate (NO_3–N) that decreased from 70–83 to 46–63 mg/l. This facilitated nitrogen removal by biogeochemical processes that reached 24–43% of the total inflowing nitrogen in the second year (with a peak of 98% in June 2005). No significant differences of nitrogen retention between plot sizes were observed after ANOVA analysis ($P = 0.05$). However, a trend relating increasing nitrogen retention, both NO_3–N and TN, to the size of the plot is observed for the full operating period (Fig. 4). The second year of wetland functioning showed a higher retention for TN and NO_3–N than the first with similar inflow concentrations (Figs. 3 and 5). In summer with a TN inflow concentration of 20 mg/l, retention increased from 19% the first year to 52% the second year. In autumn, with a similar TN inflow concentration, removal changed from 12% the first year to 41% the second year. Fig. 5 shows also a severe drop in winter of NO_3–N in regards to TN that remained at similar concentration values during all the year. This means an increase of the proportion of organic N, probably due to the reduction of fertilizer contributions and the increase of decomposition of crop remains.

Phosphorus removal efficiency was very low compared to nitrogen in these wetlands. Export or no net change was observed during half of the observations in all plots, with no significant differences between plot sizes (Fig. 3). After two years of wetland functioning, nutrient removal was not influenced by water-residence time, which was entirely dependent on watering regime of upper farms. Additionally, no relation could be established between water-residence time and plot size.

No significant changes in pH, EC, DO, TDS, TSS, turbidity, alkalinity, Cl^-, Ca^{2+}, Mg^{2+}, and K^+ were observed between inflow and outflow in the plots. There were no significant differences between inflow and outflow concentration of sodium either (12.92 ± 20.52%).

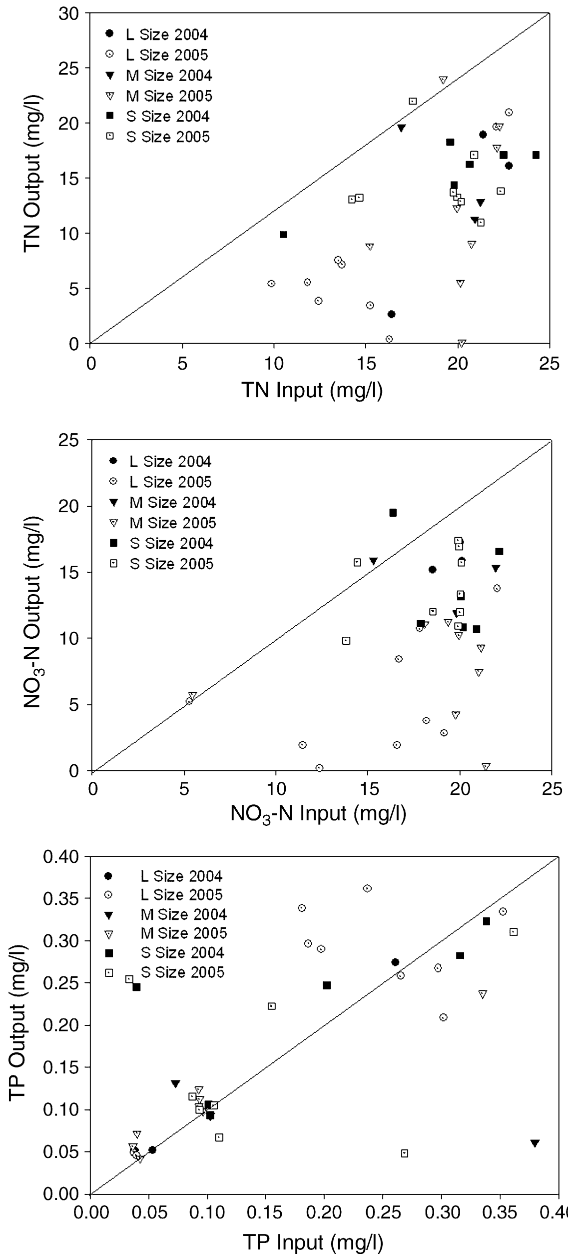

Fig. 3 – Input/output relation for samplings taken the same day for total nitrogen (TN; top) and dissolved inorganic form of nitrate (NO_3–N; medium), for years 2004 and 2005 and by wetland area (S, 50 m^2; M, 200 m^2; L, 800 m^2). Input/output relation for total phosphorus (TP; bottom).

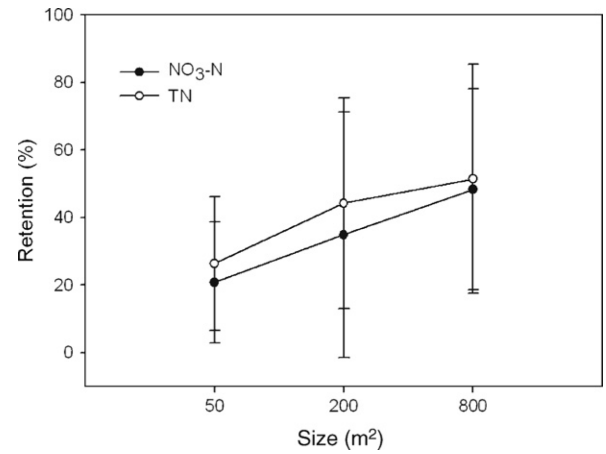

Fig. 4 – Retention percentage of TN and NO_3–N in relation to wetland area.

4.2. *Wetlands features and landscape diversity*

All the wetlands existing in our study area used for the landscape approach were located in catchments mostly occupied by irrigated agricultural fields (67–70% of their total areas). These wetlands had different forms (elongated, branched, lobed, amoeboid, and irregular) and were located in different positions in the catchments (valley bottoms, streams, and hillsides). Most of these wetlands were smaller than 500 ha and just one of them was larger than 1000 ha (Fig. 2). The area of these wetlands is 1.5–4% of their respective catchment areas. The overall landscape diversity was low (maximum 0.72). Higher values were observed in larger valleys, although little differences were found in the heterogeneity between sizes of catchments (Fig. 6). Large catchments showed cases with low diversity values at the head that increased along the transect (e.g., V08 in Fig. 6), and other cases that were high at the head and remained constant along the transect (e.g., V19 in Fig. 6). Medium catchments also had cases that showed low diversities at the head and after increased more or less quickly (e.g., V12 and V16 in Fig. 6). The rest of them had low values since the beginning, remaining constant and even decreasing advancing towards the mouth (e.g., V03 and V06 in Fig. 6). In small catchments diversity had mostly diagonal evolution, providing certain heterogeneity.

We observed four different types of diversity spectra: (a) a rectangular type, which was characterized by reaching the maximum diversity at the proximal zones of the transect (e.g., V19 in Fig. 6), (b) inverse rectangular type, characterized by very low diversity for most of the transect length but increasing up to the maximum diversity value after integration of the last parts of the transect (e.g., V12 and V08 in Fig. 6), (c) a horizontal type, characterized by an almost constant diversity for all the diversity spectrum (e.g., V06 in Fig. 6), and (d) a diagonal type, indicated a heterogeneous landscape increasing its diversity along the major axis of the valley (e.g., V17 in Fig. 6). The first diversity spectrum type was representative of catchments which still preserved untransformed zones at the head of the catchment (first areas of the transect). The second spectrum type was typical of catchments extensively transformed into irrigated fields, but still preserving wetland zones at lowest part of the catchment which coincided with the last parts of the transect. The third diversity spectrum type corresponded to catchments with extensive irrigation use but also transformed physically to flat the terrain and facilitate this land use. The fourth type of spectrum was characteristic of small catchments which still preserve some landscape heterogeneity.

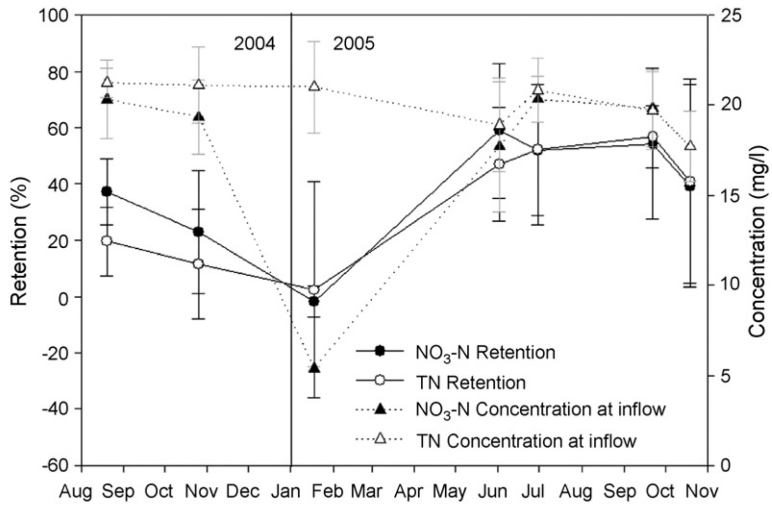

Fig. 5 – Seasonal changes of TN and NO_3–N removal percentage and TN and NO_3–N concentration at inflow during the study period.

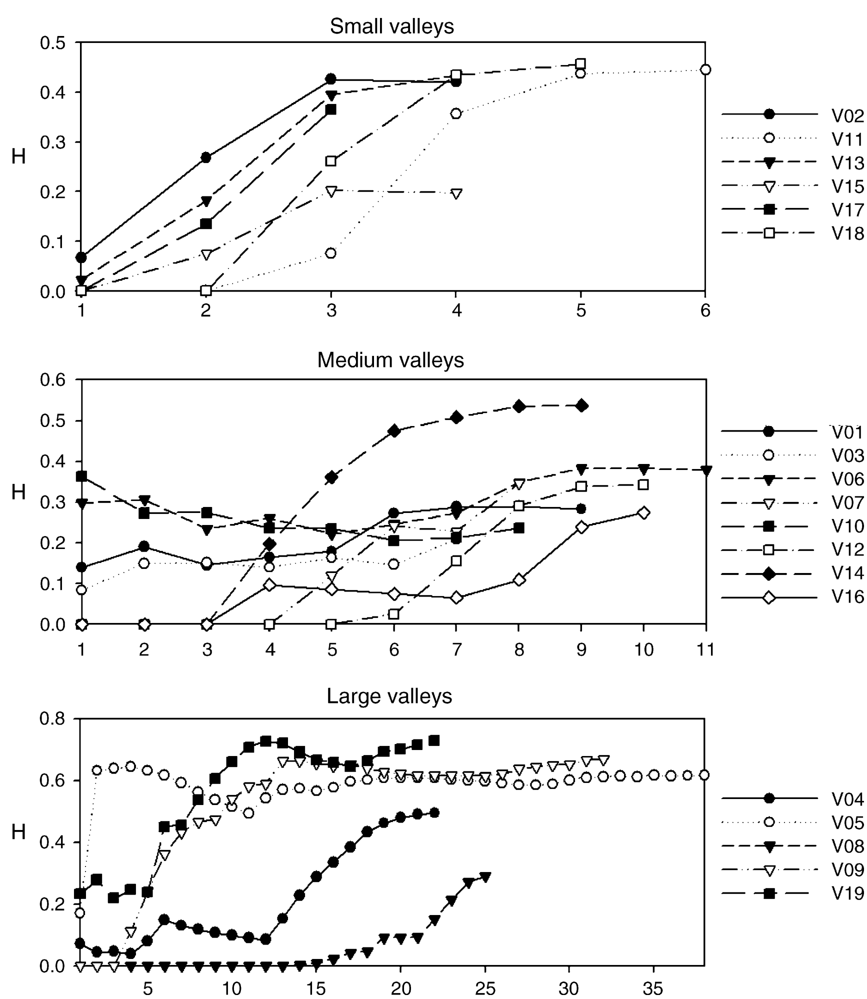

Fig. 6 – Spectral diversity of transects of the selected catchments. H refers to landscape heterogeneity of a catchment (Shannon index). Horizontal axis indicates the number of each segment and represents cumulative values of the studied transect area from the head to the mouth of the catchment.

5. Discussion

This study is an example of how constructed wetlands in a dry environment mitigate the problems related to non-point pollution from irrigation of agricultural fields. Our results encourage this approach to reduce the nutrient enrichment of water and increase landscape diversity.

Our experimental wetlands showed a remarkable N removal efficiency (24–43% of the inflowing N concentration), in spite of the fact that they were at the initial stages of their development (two years). For phosphorus, the experimental wetlands ranged from exportation to removal of 80% of the phosphorus input. These removal efficiencies are within the ranges of other case studies. Mitsch and Gosselink (2000) reviewed the results for a number of wetlands and reported a range of 40–95% of N retention and from exportation to 99% retention of phosphorus. Comín et al. (2001) reported a range of 50–98% of N retention in *Phragmites* dominated wetlands after four years of operation in similar climatic conditions. In a boreal zone, Koskiaho et al. (2003) measured 0–36% N retention with lower residence times in the wetlands (6–24 h) than in our study, and no significant results for phosphorus.

The hydrologic loading of these wetlands fluctuated and was entirely dependent on climatic conditions which regulated the water available for irrigation, and consequently, the amount of agricultural wastewater. Therefore, water-residence time was very variable in our experimental wetlands (usually between one and four days). In addition, the exceptional severe drought of 2005 maintained the wetlands dry (with no surface water) for several periods during the growing season and very likely reducing removal efficiencies. These flow characteristics may have determined the lack of relationship between nutrient removal and water-residence time or pond size. Similar situations have been studied submitting wetlands to drying and rewetting periods. An increase of extractable phosphorus release has been reported after rewetting the wetland soils (Chepkwony et al., 2001; Venterink et al., 2002).

The effect of plots size showed a trend, although not significantly. Larger wetlands (800 m^2) removed more N than medium (200 m^2) and small ones (50 m^2). Some differences between the first and second years of wetlands operation were observed. An increase of TN retention from 10–20% the first year to 40–50% the second year has been detected for similar inflow concentrations of TN. This means that these wetlands are progressing in their nutrient removal function towards and advanced state with high nutrient removal efficiency as it is usual in similar wetlands (Mitsch and Gosselink, 2000; Comín et al., 2001).

Landscape diversity slightly increased with catchment area although the maximum landscape diversity was very low (0.72) compared to other landscapes (Eiden et al., 2000). No relationship was observed between catchment area and type of landscape diversity spectrum. Large catchments only show rectangular and inverse rectangular diversity spectra. Medium-size catchments show horizontal and inverse rectangular spectra. Landscape diversity spectra of small catchments are diagonal. This lack of relationship may be attributed to the extensive land use cover change in favor of irrigated agriculture which has been a common fact all around the study area and has eliminated scale dependent landscape patterns. The result of this land use cover change has been an intensive landscape homogenization.

The expected diversity spectrum in a heterogeneous landscape is a diagonal one which would correspond to catchments with a gradient of landscape types along the major geographic axis in our study areas. It was only observed in small catchments, which does not allow incorporating large irrigated areas in the spectrum while maintaining a mosaic of land use cover types along the transect. This gives spatial heterogeneity to the overall catchment. Compared to this relatively heterogeneous catchment type, those with other types of diversity spectra have a relatively homogeneous landscape. This is a consequence of extensive land use cover changes performed at the bottom of the valley (rectangular spectrum), at the head of the valley (inverse rectangular spectrum), or all along the valley (horizontal spectrum).

In general, wetlands are located at the bottom of valleys as a consequence of surface run-off and/or groundwater discharges. Thus, restoration or construction of wetlands will strengthen an increase of diversity at lower parts of the catchments provided they were landscape-integrated at suitable sizes.

The experimental constructed wetlands covered 1.4% of its watershed where and they were able to remove up to 50% of the total nitrogen (which is the most critical pollutant in the zone) of the agricultural wastewater during the second year of functioning with an alternately (dry/flood) retention time of one to four days. It is expected that this efficiency will improve after a few years of wetland functioning. In any case, increasing the wetland area up to twice that of our experiments (approximately 3% of the total catchment area) will ensure a high removal efficiency of nitrogen by wetlands. This area (3%) is of the same order as those proposed by others to play a highly efficient role in treating non-point source pollution in agricultural watersheds (Hammer, 1992; Larson et al., 2000; Mitsch et al., 2001).

In addition to remove nutrients, restored wetlands can contribute the added value of increasing landscape diversity if they are distributed in different zones and not only at the bottom of the catchment, thus decreasing the homogeneity of areas extensively transformed by agricultural activities.

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