An ounce of prevention is worth a pound of cure: managing macrophytes for nitrate mitigation in irrigated agricultural watersheds

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Abstract

Although ubiquitous elements of agricultural landscapes, the interest on ditches and canals as effective filters to buffer nitrate pollution has been raised only recently. The aim of the present study was to investigate the importance of in-ditch denitrification supported by emergent aquatic vegetation in the context of N budget in agricultural lands of a worldwide hotspot of nitrate contamination and eutrophication, i.e. the lowlands of the Po River basin (Northern Italy). The effectiveness of N abatement in the ditch network (>18,500 km) was evaluated by extrapolating up to the watershed reach-scale denitrification rates measured in a wide range of environmental conditions. Scenarios of variable extents of vegetation maintenance were simulated (25%, 50% and 90%), and compared to the current situation when the natural development occurs in 5% of the ditch network length, subjected to mechanical mowing in summer.

Along the typical range of nitrate availability in the Po River lowlands waterways (0.5−8 mg N l-1), the current N removal performed by the ditch network was estimated in 3,300−4,900 t N yr-1, accounting for at most 11% of the N excess from agriculture. The predicted nitrate mitigation potential would increase up to 4,000−33,600 t N yr-1 in case of vegetation maintenance in 90% of the total ditch length. Moreover, a further significant enhancement (57% on average) of this key ecosystem function would be achieved by postponing vegetation mowing at the end of the growing season.

The simulated outcomes suggest that vegetated ditches may offer new agricultural landscape management opportunities for effectively decreasing nitrate loads in surface waters, with potential improved water quality at the watershed level and in the coastal zones. In conclusion, ditches and canals may act as metabolic regulators and providers of ecosystem services if conservative management practices of in-stream vegetation are properly implemented and coupled to hydraulic needs.

Keywords: nitrate pollution, ditch network, denitrification, vegetation management, Po River basin

1. Introduction

Denitrification, the reduction of nitrate ($N\text{-}NO_3^-$) to nitrogen (N) gases under anaerobic conditions, is globally considered the main biogeochemical process responsible for permanent removal of anthropogenic N along the terrestrial-freshwater-estuarine continuum. Understanding the relative contributions of terrestrial and aquatic compartments to denitrification, and its temporal and spatial variability, remains a significant challenge in biogeochemistry (Burgin et al., 2013; Duncan et al., 2013; Anderson et al., 2014). Furthermore, landscape science aimed at developing best management practices for improving water quality in agricultural watersheds could benefit from increased knowledge on denitrification (Dollinger et al., 2015; Kalcic et al., 2018). Several studies have highlighted that an internal generation of large N loads in irrigated landscapes impacted by intensive agricultural activities may not necessarily result in high export to downstream aquatic ecosystems and coastal zones (Bartoli et al., 2012; Castaldelli et al., 2013; Romero et al., 2016). Both plot-scale and basin-scale observations demonstrated that the landscape potential capacity to buffer N increases in relation to the density of small-size aquatic ecosystems (e.g. wetlands, reservoirs, drainage ditches and drains) (Lassaletta et al., 2012; Powers et al., 2015; Hansen et al., 2018), and the availability of dissolved inorganic N forms in surface waters is negatively correlated to the recirculation degree of irrigation water (Hitomi et al., 2006; Törnqvist et al., 2015). Indeed, water management practices aimed at guaranteeing a supply for agricultural uses (i.e. flow regulation, water diversions), deeply affect the N delivery patterns from the agro-ecosystems to the terminal water bodies. In particular, during the vegetative period, water volumes diverted from the main waterways are redistributed, together with associated solutes, on the landscape via extensive networks of canals and ditches where increased flow-path lengths and hydraulic residence time offer high opportunity for N processing and losses (Barakat et al., 2016; Mortensen et al., 2016).

While the role of undisturbed headwater streams, wetlands and buffer zones in watershed N dynamics has been extensively investigated in the past (Peterson et al., 2001; Mayer et al., 2007; Passy et al., 2012; Hansen et al., 2018), the interest on agricultural ditches as effective N filters has been raised only recently (McPhillips et al., 2016; Veraart et al., 2017; Speir et al., 2017; Schilling et al., 2018). Although ubiquitous elements of human-impacted watersheds, and usually accounting for the greatest part of the total length of waterways, agricultural ditches still remain largely understudied compared to other aquatic ecosystems (Pina-Ochoa and Alvarez-Cobelas, 2006) and are scarcely included in restoration programs compared to wetlands and vegetated buffer strips (Dalgaard et al., 2014; Dollinger et al., 2015; Faust et al., 2018). The usually low habitat complexity of these modified waterways is a direct consequence of human-driven alterations and management practices (e.g. homogeneous morphology of the riverbed, channelization, burial, artificial flow regime, reduction of riparian vegetation, dredging). This condition has been traditionally associated with reduced efficiencies in organic matter processing and N removal compared to natural systems, leading to the belief that these systems act simply as conduits for N (Pinay et al., 2002; Beaulieu et al., 2015; Moore et al., 2017). However, multiple features of agricultural ditches may support a high mitigation potential towards $N\text{-}NO_3^-$: i) their being the first point of contact for diffuse and point N loads entering the hydrological network; ii) the occurrence of the three primary controls directly influencing occurrence and magnitude of denitrification, i.e. anoxic environment, availability of $N\text{-}NO_3^-$ and organic carbon; iii) the tight terrestrial-aquatic coupling resulting from their extensive and capillary distribution across the landscape; iv) the high opportunity for N microbial processing due to long hydraulic residence time and large ratio between biological active surfaces and in-stream water volumes carrying excess nutrients; v) the shallow water depth that potentially supports the role of aquatic vegetation as "ecosystem engineer" in regulating biogeochemical processes and providing the development of denitrification hotspots, such as biofilms on submerged portions and oxic-anoxic niches in organic matter-rich sediments; vi) the frequent recirculation of water through the landscape, which maximizes the interaction among water and bioreactive surfaces, especially during spring and summer period when high water temperatures (up to >25 °C in temperate zones) enhance microbial processes (McClain et al., 2003; Hines, 2006; Marion et al., 2014; Soana et al., 2017). All the abovementioned features make agricultural ditches and canals more similar to wetlands than to higher-order streams (Faust et al., 2018; Vymazal and Březinová, 2018).

Only a few attempts were made to assess the magnitude of catchment-scale denitrification in low-order waterways and drainage ditches, but modelling tools and geospatial approaches were often based on the up-scale of measurements of potential denitrification rates determined under optimal conditions of N-NO_3^- and organic carbon availability, not always reflecting the actual *in situ* activity (Oehler et al., 2009; Christopher et al., 2017; O'Brien et al., 2017). Moreover, the potential for water quality improvement exerted by vegetation in the ditch network has never been assessed at the watershed scale. In-stream vegetation is an important interface between croplands and surface water bodies, thus its presence and abundance are considered key elements in determining the potentiality of the canal network to provide ecosystem services in general (Bolpagni et al., 2013; Boerema et al., 2014; Dollinger et al., 2015), and water purification in particular, due to a complex synergistic action with bacterial communities (Pierobon et al., 2013; Taylor et al., 2015; Vymazal and Březinová, 2018). Nevertheless, aquatic vegetation is often considered only as a hindrance for water circulation by water management authorities, and thus regularly removed to preserve the hydraulic performance (Levavasseur et al., 2014).

The Po River catchment, the largest hydrographic system in Italy (652 km, more than 71,000 km^2, about a quarter of the national territory), is one of the most densely populated and agriculturally productive areas in Europe, but also a paradigmatic case study for N pollution, eutrophication and related implications for environmental policies (Viaroli et al., 2018; Martinelli et al., 2018). The plain zone (46,000 km^2), the largest Italian alluvial basin, is crossed by an extensive network of mostly artificial canals and ditches with irrigation, drainage, and flood control purposes. A comprehensive N budget has proven that the deltaic portion of the catchment, intensively cultivated and irrigated, acts as an effective N sink, buffering not only the N surplus leached from the croplands but also part of the N load generated by upstream agro-ecosystems and imported with drainage and irrigation water (Castaldelli et al., 2013). Multiple lines of evidences suggest that denitrification in vegetated ditches accounts for the majority of N losses during water transit through the hydrological network (Pierobon et al., 2013; Castaldelli et al., 2015; Soana et al., 2018).

Our hypothesis is that in highly hydraulic-regulated and simplified, agricultural watersheds, landscape management may deeply affect the balance between N sources and sinks and thus, at a widely variable range, also the quality of outflows and N delivery to coastal areas. We investigated whether N excess in intensive agriculture impacted watersheds may be efficiently controlled by conservative management of in-stream macrophytes to promote denitrification. Our case study was the Po River system because it represents a worldwide hot spot of $N\text{-}NO_3^-$ contamination and eutrophication. We hypothesize that vegetated ditches may offer new management opportunities for effectively decreasing NO_3^- loads in surface waters due to the intertwined action of macrophytes and microbial communities promoting N processing and sustaining their natural depuration capacity.

The effectiveness of N abatement in the ditch network was evaluated by combining reach-scale denitrification rates from previous studies in the area, measured across a wide range of environmental conditions, and GIS-based upscaling. The potentiality of the ditch network to buffer watershed-scale $N\text{-}NO_3^-$ pollution was quantified for four different levels of vegetation maintenance (5%, 25%, 50% and 90% of the total length) and two mowing timings, i.e. the *current management* where the cutting is performed in the middle of the summer and the *conservative management* where the cutting is postponed to the end of the growing season.

2. Material and Methods

2.1 Study area description

The study area included the Po River plain below 50 m above sea level (a.s.l.), an area of ~ 9100 km^2 (Fig. 1A, B), covering ~15% of the whole Po River basin but also the most intensively cultivated zone. We used a 20-m digital elevation model, provided by the Italian Ministry of the Environment and Protection of Land and Sea (http://www.pcn.minambiente.it) to generate a slope map from which the area laying within the 50 m sea-level curve was selected. The Po River crosses this area with its

300 km long final reach before the delta in the Adriatic Sea. This area has a humid subtropical climate (Type Cfa, according to Köppen classification), with a yearly average rainfall of 700-800 mm, mainly concentrated in autumn and spring (Joint Research Centre meteorological datasets, http://eusoils.jrc.ec.europa.eu/library/Data/EFSA/).

The landscape surrounding the river course is a typical over-exploited plain devoted to farming (mainly cows and pigs) and agricultural practices, resulting in >87% of the area cultivated by corn, wheat and temporary grassland as main crops (6th Agricultural Census, National Institute for Statistics, 2010, http://dati-censimentoagricoltura.istat.it), while approximately 6% of the surface is classified as urbanized land (~1.8 million inhabitants, ~11% of the total population in the Po River catchment, average density 200 inhabitants km^{-2}; 15th Population and housing census, National Institute for Statistics, 2011, http://dati.istat.it/), and <3% as forest and semi natural area (Corine Land Cover inventory 2012, level 1; Fig. 1C). Following the enactment of the European Water Framework Directive (2000/60 CE), about half of the whole study area was declared "vulnerable to nitrates from agricultural sources". More than 270 municipalities (surface from 6 to 405 km^2) are included totally or partly within the study area, belonging to nine Italian provinces, three in the Lombardy Region (Lodi, Cremona and Mantova), five in the Emilia-Romagna Region (Piacenza, Parma, Reggio Emilia, Modena and Ferrara), and one in the Veneto Region (Rovigo).

The area is homogenous in terms of source of the irrigation water (i.e. Po River) and chemical quality of surface waters. During the growing season (May−September), water is diverted mainly from the Po River into some main canals to irrigate croplands by an extensive open-earth ditch network of >18,500 km (Fig. 1D; average density ~3 km per km^2 of utilised agricultural land), managed by fourteen local land reclamation authorities. Flat topography, low soil permeability and slopes of a few cm per km generate low water velocities (up to 10 cm s^{-1}) and bottom sediments are usually a combination of muddy sand or muddy silt.

Heavy management practices of the ditch network (i.e. dredging, mechanical mowing, chemical weeding) has led to the complete disappearance of the submerged vegetation (Piccoli and Gerdol, 1983). The voluntary introduction, widespread diffusion and establishment of the exotic grass carp (*Ctenopharyngodon idella*, Valenciennes 1844) contributed as well to the biological control of submerged plants and to the turbid, phytoplankton-dominated status of most watercourses (Milardi et al., 2015). Bank mowing is performed twice a year, in the middle of the summer (i.e. August) to facilitate water flow during the period of highest water demand for irrigation, and before the period of intense rainfall (i.e. October) to reduce flood risk. Natural propagation and evolution of helophytes are strongly affected by routine management practices that have been in place since 1990s and as a result, vegetation is maintained only in isolated stretches of the ditch network with low flood risk, representing about 5% of the total length. Macrophyte stands are generally not monospecific but composed by two main species, *Phragmites australis* (Cav.) Trin. ex Steud. and *Typha latifolia* L., and to a lesser extent also by *Glyceria maxima* Hartm. At peak biomass, standing stocks range from 230 to 550 g of dry biomass per m^2 (Pierobon et al., 2013). Before the introduction of mechanical vegetation mowing (in the 1980s), vegetation cover was supposed to be 90% of the total length, with the exclusion of the deeper canals used also for navigation and where depth, sediment resuspension and turbidity prevented both submerged and emergent macrophytes to develop.

2.2 Denitrification rates in vegetated and unvegetated ditches

Measurements of N-NO_3^- removal rates (kg km^{-1} d^{-1}) obtained for vegetated and unvegetated ditches by reach-scale in-out N-NO_3^- budgets along the irrigation period (May–September), corresponding to the macrophyte growing season, were synthesized from previous sampling campaigns (Pierobon et al., 2013; Castaldelli et al., 2015). These experimental activities were conducted on several drainage ditches belonging to the hydrological network of the Po di Volano basin, a deltaic reclaimed alluvional area (~2,600 km^2; Fig. A1, Appendix A), representing about one third of the total area under investigation in the present study. The studied sites were representative of the dominant waterways type of the whole Po River lowlands, in terms of chemico-physical features (i.e. dissolved inorganic N and organic carbon availability), substrate, hydraulic regime, routine management practices and, if present, emergent aquatic vegetation. N-NO_3^- removal rates were estimated from changes in N-NO_3^- loads along the selected reached by adopting a Lagrangian sampling scheme during stable weather and flow conditions. Details about experimental design, analytical methods and calculations of N-NO_3^- removal rates are reported in Pierobon et al. (2013) and Castaldelli et al. (2015).

As vegetation presence and N-NO_3^- availability were considered the main factors affecting ditch N dissipation capacity, predictive relationships between incoming water N-NO_3^- concentrations and daily reach-scale N-NO_3^- removal rates were built employing the previous acquired datasets, separately for vegetated (data from Pierobon et al., 2013 and Castaldelli et al., 2015) and unvegetated ditches (data from Pierobon et al., 2013). Specifically, water N-NO_3^- concentrations were plotted against experimental values of N-NO_3^- daily removal rates and the obtained regression was used to predict N-NO_3^- removal in vegetated ditch sediments (Dr_V) as a function of water N-NO_3^- availability spanning the typical range found in N-polluted artificial waterways of the Po River lowlands (0.5−8 mg L^{-1}). Since this dataset did not include direct measurements of N-N_2 production rates, we provided independent evidences supporting the hypothesis of denitrification being the main process responsible for N-NO_3^- dissipation in vegetated sediments. First, our previous study (Pierobon et al., 2013) demonstrated that plant N uptake and sequestration in biomass represent a small fraction of the total N-NO_3^- consumption (<5%). This statement was further supported by presenting dataset of N-N_2 production rates and corresponding N-NO_3^- removal rates obtained by the simultaneous application of the N_2 open channel method and reach-scale N-NO_3^- budget in vegetated ditch sediments (Castaldelli et al., 2015; Castaldelli et al., 2018, Soana et al., 2018). The N_2 open channel approach, proposed by Laursen and Seitzinger (2002), provides the direct estimate of whole-system denitrification in running waters by measuring the variation of its end-product, i.e. N_2, within a water parcel moving from upstream to downstream. A model-based approach is used to solve for denitrification rates, correcting the variations of N_2 for atmospheric exchanges during transport, according to concentration gradients and channel morphology (e.g. width and depth). This approach allows to estimate *in situ* net N_2 fluxes (i.e. total denitrification, including denitrification coupled to nitrification, minus N_2 fixation) under natural conditions, at a scale comparable to the reach-scale N-NO_3^- budgets. Details about experimental design, analytical methods and calculations of N-N_2 production rates are reported in Castaldelli et al. (2015). Descriptive statistics of the used datasets are reported in Appendix A (Figs. A2, A3 and A4).

Daily denitrification rates for unvegetated sediments (i.e. N-N_2 production rates, Dr_{UV}, mg m^{-2} d^{-1}) were estimated according to a simple diffusion-reaction model proposed by Christensen et al. (1990) for N-NO_3^--rich streams, and previously tested in unvegetated sediments of several shallow slow-flow aquatic environments of the Po River Plain (Bartoli et al., 2008; Pinardi et al., 2009; Racchetti et al., 2011):

$$\mathrm{Dr_{UV}} = \mathrm{SOD} \cdot 0.8 \cdot \left[\sqrt{1 + 0.82.\frac{\mathrm{N - NO_3^-}}{\mathrm{DO}} \cdot \frac{1}{0.8}} - 1\right]$$

Where:

SOD = Sediment Oxygen Demand (mg m^{-2} d^{-1})

N-NO_3^- = concentrations of water column nitrate (mg L^{-1})

DO = concentrations of water column dissolved oxygen (mg L^{-1})

0.8 = ratio between the activities per unit of volume in the denitrification and oxygen respiration zones, found to be relatively constant in stream sediments and biofilms (Nielsen et al., 1990)

0.82 = ratio between the diffusion coefficients of N-NO_3^- and DO (Christensen et al., 1990; Nielsen et al., 1990).

The predicted rates represent denitrification of N-NO_3^- diffusing from the water column into anoxic sediments, which constitutes the dominant source of NO_3^- required for denitrification when concentrations are generally higher than 0.5 mg L^{-1}, as demonstrated for sediments across a wide range of freshwater and marine ecosystems (Pina-Ochoa and Alvarez-Cobelas, 2006; Seitzinger et al., 2006). Furthermore, nitrification in organic-rich muddy sediments may be severely limited by oxygen availability, thus being only a minor source of N-NO_3^- for coupled denitrification (Racchetti et al., 2011; Soana et al., 2015). N-N_2 production rates obtained on a surface basis were converted into values expressed per unit of ditch length (kg km^{-1} d^{-1}) by considering an average ditch width of 3 m (Pierobon et al., 2013; Castaldelli et al., 2015).

Water quality data of the ditch network (i.e. T-water temperature, DO, N-NO_3^-, BOD_5- Biochemical Oxygen Demand) were provided by the Regional Agency for the Environmental Protection (ARPA) of Lombardy and Emilia-Romagna Regions (monthly data, from 2009 to 2014, http://www.arpalombardia.it/Pages/Acqua.aspx; https://www.arpae.it/index.asp?idlivello=112). Stations on artificial waterways (N=70) located within the 50 m a.s.l. area were selected from the ARPA surface water monitoring network, which include natural systems, such as streams, rivers and lakes. The equation by Christensen et al. (1990) was applied to all ARPA surveys on ditch network for which datasets of water T (°C), DO (mg L^{-1}) and N-NO_3^- (mg L^{-1}) were concomitantly available. Descriptive statistics of the water quality datasets are reported in Appendix A (Table A1). A range of daily denitrification rates was calculated for each month from January to March and from August to December.

Experimental measurements of SOD for ditch environments of the Po River plain are limited to summer months (Castaldelli et al., 2015; Soana et al., 2017) and seasonal evolution is lacking. Thus, following Soana et al. (2011), reasonable estimates of SOD ditch sediments were calculated as a function of water T by applying the equation obtained from a large dataset of SOD values measured in a vast array of shallow eutrophic environments having muddy bottoms similar to slow-flow agricultural waterways (Pinardi et al., 2009; Racchetti et al., 2011; Pinardi et al., 2011; Ribaudo et al., 2011; Soana et al., 2015) (Appendix A, Fig. A5).

2.3. Scaling N-NO_3^- removal rates to the ditch network: scenarios of vegetation maintenance

In order to extrapolate N-NO_3^- removal measurements to the entire study area, a detailed map of the canal and ditch network (Fig. 1D) was created in QGIS 2.18 by merging vector data obtained by the geoportals of the Italian regions included in the study area (Lombardy Region, http://www.geoportale.regione.lombardia.it/; Emilia-Romagna Region http://geoportale.regione.emilia-romagna.it/it; and Veneto Region, http://idt.regione.veneto.it) and of the Po River Basin Authority within the framework of the Water Management Plan of the Po River Basin (http://pianoacque.adbpo.it/piano-di-gestione-2015/).

Four scenarios of in-stream vegetation maintenance were simulated, 5% (current level), 25%, 50% and 90% (historic level) of total ditch length, while the remaining length was assumed as unvegetated. For each scenario, two options of vegetation management were assessed, i.e. the *current management* where the mowing is performed in the middle of the summer (effect of vegetation on N dynamics for 120 days, from April to July), and the *conservative management* where the mowing is postponed to the end of the growing season (effect of vegetation on N dynamics for 210 days, from April to October). The potential N-NO_3^- removal capacity of the ditch network was thus tested in eight different conditions (four extensions of in-stream vegetation x two options of vegetation management) and compared to the N budget in the surrounding agricultural lands.

The following equation was used to predict the N-NO_3^- removal capacity (t yr^{-1}) of the ditch network considering the *current management* of the vegetation:

$$\mathrm{NR_{cur}} = ((100 - \mathrm{V\%}) \cdot \mathrm{L} \cdot \sum(\mathrm{Dr_{UV}} \cdot \mathrm{n}) + \mathrm{V\%} \cdot \mathrm{L} \cdot \mathrm{Dr_{V}} \cdot \mathrm{N_{cur}}) * \frac{1}{1000}$$

where:

V% = scenario of vegetation maintenance (5%, 25%, 50%, 90% of total ditch length)

L = total ditch length (km)

Dr_{UV} = daily rates in unvegetated ditch sediments calculated by the model proposed by Christensen et al. (1990) (kg km^{-1} d^{-1}) for each month from January to March and from August to December

n = number of days of each month (from January to March and from August to December)

Dr_V = daily rates of N-NO_3^- removal (kg km^{-1} d^{-1}) in vegetated ditch sediments calculated as a function of water N-NO_3^- availability in the range 0.5–8 mg L^{-1} by employing the predictive relationship explained in paragraph 2.2

N_{cur} = 120, numbers of days with vegetation maintenance in the *current management* (d yr^{-1})

The following equation was used to predict the N-NO_3^- removal capacity (t yr^{-1}) of the ditch network considering the *conservative management* of the vegetation:

$$\mathrm{NR_{cons}} = \left((100 - \mathrm{V\%}) \cdot \mathrm{L} \cdot \sum(\mathrm{Dr_{UV}} \cdot \mathrm{n}) + \mathrm{V\%} \cdot \mathrm{L} \cdot \mathrm{Dr_V} \cdot \mathrm{N_{cons}}\right) \cdot \frac{1}{1000}$$

where:

V% = scenario of vegetation maintenance (5%, 25%, 50%, 90% of total ditch length)

L = total ditch length (km)

Dr_{UV} = daily rates in unvegetated ditch sediments calculated by the model proposed by Christensen et al. (1990) (kg km^{-1} d^{-1}) for each month from January to March and from November to December

n = number of days of each month (from January to March and from November to December)

Dr_V = daily rates of N-NO_3^- removal (kg km^{-1} d^{-1}) in vegetated ditch sediments calculated as a function of water N-NO_3^- availability in the range 0.5–8 mg N L^{-1} by employing the predictive relationship explained in paragraph 2.2

N_{cons} = 210, numbers of days with vegetation maintenance in the *conservative management* (d yr^{-1})

To determine the likely variation of the potential N-NO_3^- removal capacity of the entire ditch network, the interquartile range (first and fourth quartiles as inferior and superior extremes) for N removal in unvegetated condition was combined with the interquartile range for N removal in presence of vegetation. Thus, for each scenario of in-stream vegetation and condition of water N-NO_3^- availability, best-case and worst-case situations were estimated.

2.4. Calculation of N budget in agricultural land

A N balance in agricultural land of the investigated area was compiled for the year 2010 by integrating official census data in a nutrient budgeting approach previously applied to some sub-basins of the Po River system (Soana et al. 2011; Bartoli et al., 2012; Castaldelli et al. 2013; Pinardi et al., submitted) and other Italian temporary rivers (De Girolamo et al. 2017). N budget (t yr^{-1}) was estimated as the net difference between N inputs and N outputs across the Utilised Agricultural Area (UAA) and calculated as follow:

$$\mathrm{N\ budget} = \mathrm{N_{Man}} + \mathrm{N_{Fert}} + \mathrm{N_{Fix}} + \mathrm{N_{Dep}} - \mathrm{N_{Harv}} - \mathrm{N_{Vol}} - \mathrm{N_{Den}}$$

where:

N_{Man} = N in livestock manure applied to UAA (t yr^{-1})

N_{Fert} = synthetic N fertilizer applied to UAA (t yr^{-1})

N_{Fix} = agricultural N_2 fixation associated with N fixing crops (t yr^{-1})

N_{Dep} = atmospheric N depositions on UAA (t yr^{-1})

N_{Harv} = N exported from UAA with crop harvest (t yr^{-1})

N_{Vol} = NH_3 volatilization (t yr^{-1})

N_{Den} = denitrification in UAA (t yr^{-1})

UAA was summarised as arable land (cereals, industrial crops, and fresh vegetables), grassland (temporary and permanent pasture for fodder production), and permanent woody crops. N budgets were first calculated at the municipal scale, i.e. the smallest administrative unit at which official agricultural statistics are available. Municipality-level N budgets were then weighted for the percentage of each municipality surface included within the 50 m a.s.l. boundaries, and finally summed up to obtain the total budget of the study area. Input, output, and surplus of each municipality as well as the large scale N balance were expressed in unit of mass per time (t yr^{-1}), and on a per-area basis, after normalization for the corresponding UAA (kg ha^{-1} yr^{-1}).

N surplus represents the excess N unused by crops that remains in the soil, i.e. an indicator of the N use efficiency in the agricultural system. Being net of losses to the atmosphere, it is also a proxy of the potential source of diffuse pollution for surface and ground waters, via runoff and leaching, to be compared to the N retention capacity ascribed to the ditch network under different scenarios of vegetation restoration. A detailed description of data sources, budget equations and uncertainty assessment of N budget is reported in Appendix B.

2.5. Statistical analyses

Differences between vegetated and unvegetated sites in N-NO_3^- removal rates, chemico-physical (water T, N-NO_3^-, DO) and hydraulic (discharge) features obtained from experimental activities were assessed by the non-parametric Mann-Whitney test, due to lack of variance homogeneity for most of the datasets. A multiple regression model of reach-scale N-NO_3^- removal rates vs. water T, N-NO_3^- and DO and discharge was established, separately for vegetated and unvegetated sites. Critical level was $p=0.05$ and statistical analyses were conducted with SigmaPlot 11.0 (Systat Software, Inc., CA, USA).

3. Results and discussion

3.1. Water quality of the ditch network and reach-scale N-NO_3^- removal rates in vegetated and unvegetated sediments

Table 1 reports the main chemico-physical and hydraulic features of vegetated and unvegetated ditches of the Po River lowlands for which experimentally measured values of reach-scale N-NO_3^- removal rate are available from previous studies (Pierobon et al., 2013, Castaldelli et al., 2015). Ditch inflow water was highly variable in term of N-NO_3^- and DO concentrations, but quite homogeneous in temperature, reflecting the typical quality of the Po River lowland waterways during the irrigation period. The reported parameters were not significantly different between vegetated and unvegetated sites ($p>0.05$). In the regression analyses, only N-NO_3^- was found to significantly explain N-NO_3^-

removal rate variability, indicating that its availability was a key factor controlling N retention processes in the ditch network when water temperature generally exceeds 20°C. This evidence highlights the absence of saturation in N removal along the measured $N\text{-}NO_3^-$ concentration range and supports the up-scale simulations conducted for different conditions of water $N\text{-}NO_3^-$ availability. Reach-scale $N\text{-}NO_3^-$ removal rates were predicted by incoming $N\text{-}NO_3^-$ concentrations for both vegetated (linear regression, $y=(1.939\pm0.221)x-(0.371\pm0.601)$; $R^2=0.647$, slope: $p<0.0001$, intercept: $p>0.05$, N=44) (Fig. 2A) and unvegetated ($y=(0.364\pm0.053)x+(0.289\pm0.095)$; $R^2=0.441$, slope: $p<0.0001$, intercept: $p>0.05$, N=61) ditch sediments (Fig. 2B). Daily $N\text{-}NO_3^-$ removal was significantly higher in vegetated ditches than in unvegetated ones (*p*<0.001), with rates ranging from 0.01 to 20.52 kg km^{-1} d^{-1} (median value 1.94 kg km^{-1} d^{-1}), and from 0.02 to 3.10 kg km^{-1} d^{-1} (median value 0.50 kg N km^{-1} d^{-1}), respectively.

Summing up experimental activities performed in the studied area, the capacity of agricultural canals and ditches to control N pollution is maximised if the following features are simultaneously met: low water depth (<30-40 cm), low water flow (3-6 cm s^{-1}), concentrations of dissolved inorganic N higher than 0.5 mg L^{-1}, and availability of dissolved organic matter (e.g. BOD_5 > 5 mg L^{-1}). In these conditions, if emergent vegetation is present, a reduction of up to 40% of the incoming N load throughout the irrigation period can be reached in a 1 km-long stretch. Otherwise, in the same chemo-physical conditions but in absence of vegetation, N loads behave almost conservatively. Reach-scale $N\text{-}NO_3^-$ consumption and $N\text{-}N_2$ production rates in vegetated ditch sediments, measured by applying simultaneously in-out $N\text{-}NO_3^-$ budget and N_2 open channel method were positive correlated ($R^2=0.9224$, slope: $p<0.0001$, intercept: $p>0.05$, N=39), proving that denitrification of water column $N\text{-}NO_3^-$ was the main reaction responsible for its dissipation, a key process in the context of eutrophication-related issues (Fig. 3).

Official monitoring surveys of the whole ditch network showed water features typical of eutrophic freshwater environments of temperate zones (Fig. 4), with concentrations of $N\text{-}NO_3^-$ and BOD_5 constantly higher than 1.5 mg L^{-1} and 3.0 mg L^{-1}, respectively, along the year, and temperature >20°C maintained during the whole late spring-summer period when the concomitant elevated sediment oxygen consumption may create favourable conditions for denitrification to occur. Water temperature displayed a clear seasonal cycle, with the highest values recorded in late summer (August, media values 25.9°C) and minimum values in the middle of the winter (January, median value, 5.9°C), resulting in a marked seasonal variation of over 20°C (Fig. 4A). $N\text{-}NO_3^-$ revealed an approximately inverse seasonal pattern to temperature, with median values constantly lower than 3 mg L^{-1} from April to October (Fig. 4B). From winter to summer $N\text{-}NO_3^-$ median concentrations more than halved, passing from a maximum in January of 5.5 mg L^{-1} to a minimum in August of 1.51 mg L^{-1}. A slight decrease from winter to summer was detected also for DO concentrations although they never dropped below 6.6 mg L^{-1} (Fig. 4C). Median BOD_5 concentrations varied between 3.0 and 4.8 mg L^{-1} without a clear seasonal pattern (Fig. 4D).

Using monitoring data of $N\text{-}NO_3^-$ and DO and SOD obtained as a function of temperature, denitrification rates of water column $N\text{-}NO_3^-$ were calculated according to the Christensen model (Christensen et al., 1990; Fig. 5). Ditch network expressed the highest $N\text{-}N_2$ production rates in the middle summer (July, median value 0.53 kg km^{-1} d^{-1}). The lowest rates were observed in winter months (0.25-0.34 kg km^{-1} d^{-1}, from December to February), despite the highest $N\text{-}NO_3^-$ availability at this time of the year because the water temperature was low. However, denitrification did not show a pronounced seasonal pattern, with the lowest median winter rates being about half of the highest summer rates. With the exclusion of winter months, $N\text{-}N_2$ production rates were constantly higher than 0.40 kg km^{-1} d^{-1} in the rest of the year and a clear increase in the spring/summer shift followed by a decrease at the onset of autumn was not observed. This is probably due to the lower $N\text{-}NO_3^-$ availability when bacterial activity might be promoted by water temperature (Fig. 4A, B). For the irrigation period (May−September), $N\text{-}N_2$ production rates calculated according to the Christensen model (monthly median values varying from 0.47 to 0.53 kg km^{-1} d^{-1}) overlapped the range of $N\text{-}NO_3^-$ removal rates experimentally measured by reach-scale $N\text{-}NO_3^-$-budgets in selected unvegetated ditches (range: 0.02−3.10 kg km^{-1} d^{-1}; median value: 0.50 kg km^{-1} d^{-1}), while resulted significantly lower than the experimental values obtained for vegetated ditches (range: 0.01−20.52 kg km^{-1} d^{-1}; median value: 1.94 kg km^{-1} d^{-1}) (Fig 2, Fig. 5).

Shallow aquatic ecosystems colonised by rooted macrophytes are usually considered hotspots of denitrification due to the development of multiple biological active surfaces both in the water column and in the rhizosphere (Pierobon et al., 2013; Taylor et al., 2015). Submerged portions like stems and leaves provide physical support for the growth of epiphytic biofilms, complex matrices of bacteria, microalgae and debris, that promote reactions of nutrient retention and transformations whose kinetics are usually maximised due to the constant renewal of the water in contact with them (Toet et al., 2003; Soana et al., 2018). Due to the ability to transport and release oxygen in the sediment, rooted plants allow the establishment of a mosaic of oxic and anoxic microniches. This extensive development of oxic-anoxic interfaces promotes the aerobic degradation of the organic matter and the coupling of aerobic and anaerobic processes, e.g. denitrification coupled to nitrification, that remove N from the system (Vila-Costa et al., 2016; Roley et al., 2018). The influence of vegetation on denitrification occurs also by increasing the availability of labile organic compounds which act as substrates for heterotrophic denitrifying communities, such as radical exudates, decaying plant litter and suspended particles trapped as a consequence of low hydrodynamics among the vegetation stands (Hang et al., 2016; Srivastava et al., 2017).

3.2. Predicted N-NO_3^- removal in the ditch network: scenarios of vegetation maintenance and comparison to agricultural N surplus

The removal of macrophytes from the ditch network causes a simplification of the agricultural landscapes in terms of denitrification hotspots, due to the loss of multiple interfaces among water, sediment and vegetation itself, a basic requirement for the microbial processes underlying the depuration capacity (Boerema et al., 2014; Vymazal and Březinová, 2018). Considering the typical range of N-NO_3^- availability (0.5−8 mg L^{-1}) in the Po River lowlands waterways, the current N-NO_3^- removal performed by the ditch network, where vegetation cover is limited to 5% of its extension, was estimated to vary between 3,300 and 4,900 t yr^{-1} (Fig. 6A). This buffer capacity represents an irrelevant fraction of the dissolved inorganic N load that the Po River exports to the Adriatic Sea, on average ~110,000 t yr^{-1} (Viaroli et al., 2018). The predicted N-NO_3^- mitigation potential of the ditch network would increase up to 4,000−33,600 t yr^{-1} in case of restoring vegetation on 90% of its extension, albeit maintaining the current management practice, i.e. the mowing operations performed in the middle of the summer (Fig. 6A). Differently, a more conservative management of vegetation with the cutting postponed to the end of the growing season, enhances the N-NO_3^- mitigation potential of the ditch network in terms of 1-26% and 16-168%, for the 5 and 90% scenarios, respectively (Fig. 6B). Supposing the highest N-NO_3^- availability commonly found in the Po River lowlands waterways (up to 8 mg L^{-1}), the ditch network would at present express a potential N-NO_3^- removal of 6,200 t yr^{-1}, if only its 5% extension where vegetation is present was managed conservatively. This key ecosystem function would increase almost tenfold (up to 56,600 t yr^{-1}) in case of restoring vegetation on 90% of the total length like in the past. This amount almost equals the N load needing treatment (supposing a depuration target of 75% of the incoming N) produced annually by the whole Po River basin population of over 16 million inhabitants.

The N budget revealed a N surplus in the agricultural soils of the investigated area (~46,000 t yr^{-1}), due to N inputs (~163,000 t yr^{-1}) exceeding N outputs (~117,000 t yr^{-1}). Total inputs were almost equally divided among synthetic fertilizers (34%), livestock manure (30%), and biological fixation (33%), with atmospheric depositions accounting for the remaining 3% (Table 2). Total N input in the municipalities of the studied area ranged from 112 to 817 kg ha^{-1} yr^{-1} and overall the average input rate was 260 kg ha^{-1} yr^{-1}. Livestock manure was produced mainly by cattle (60%) and swine (32%) farming, sustained by large portions of agricultural lands devoted to fodder crops (mainly N-fixing alfalfa, 25% of total agricultural surfaces), maize (23%), and other cereals (26%). The main N output term was crop harvest, mainly as feed for livestock which accounted for over 81% of the total N removal from agricultural lands. The remaining portion was estimated to be lost as N gases to the atmosphere by ammonia volatilization (10%) and denitrification (9%). Total N output in the municipalities of the studied area ranged from 123 to 362 kg ha^{-1} yr^{-1} and overall the average output rate was 186 kg ha^{-1} yr^{-1}.

More than one quarter (~28%) of total N input remained in the agricultural soils as N surplus, representing a risk of N runoff and potential pollution of aquatic ecosystems. However, N input and output patterns varied spatially within the area, together with the resulting N surplus (on average 74 kg ha^{-1} yr^{-1}), reflecting the heterogeneous distribution of agricultural and farming activities across the territory, in specific three different agro-environments (Fig. C1, Appendix C). The first is the zone on the Po River hydrographic left side (Fig. 1B) where livestock manure was the biggest N source in most of the municipalities. This area was characterized by a dramatically high density of cows and pigs (equivalent livestock units: up to 15-20 per ha of agricultural surfaces) generating N inputs up to 500-600 kg ha^{-1} yr^{-1} that, despite the widespread occurrence of high N-demanding crops (i.e. maize), were generally in excess compared to uptake and accumulation in crop biomass. The second is the Po River hydrographic right side zone (Fig. 1B), with the exclusion of the deltaic sub-basin of the Po di Volano, where the alfalfa dominated among crops resulting in N input from biological fixation. The last is the Po di Volano sub-basin (Fig. A1, Appendix A) characterized by low livestock densities and cereals as major crops amended almost exclusively by synthetic fertilizers.

In the current management situation of vegetation on 5% of ditch length which is cut in the middle of the growing season, N abatement performed by ditch network accounted for only 7-11% of the N surplus (Fig. 6A). This percentage would increase by only a small amount (7-13%) if vegetation was managed conservatively. Differently, for the 90% vegetation scenario with the current management, the $N\text{-}NO_3^-$ mitigation potential of the ditch network represented from 9% to 73% of the N excess in agricultural lands (Fig. 6A), due to the presence of vegetation in ditch stretches characterised by different degrees of $N\text{-}NO_3^-$ contamination. If vegetation would be maintained with a conservative management on 90% of ditch length having $N\text{-}NO_3^-$ concentrations > 6 mg L^{-1}, N abatement would potentially overcome the N surplus by about 20% (Fig. 6B) and reduce the dissolved inorganic N load exported annually by the Po River to the Northern Adriatic Sea.

4. Management and research perspectives

The outcomes of the up-scale model presented here highlight that routine ditch network management practices involving aquatic vegetation removal deeply affect its capacity to process N-NO_3^-, with cascading implications for N dynamics at the watershed scale. One of the most valuable ecosystem service provided by macrophytes, i.e. the mitigation of N-NO_3^- pollution, may play a considerable role in water quality improvement in agricultural landscapes. Thus proper management of the interfaces between croplands and surface water bodies may potentially reduce an appreciable amount of the N surplus generated by farming activities and ultimately delivered to the coastal zones. Even if tightly intertwined by water management practices, agricultural surfaces and river networks are commonly managed and investigated separately, with consequent limitations in solving nutrient excess issues across agricultural landscapes. An increasing amount of research is promoting the use of vegetated ecological ditches (eco-ditches) as an agricultural best management practice. Eco-ditches have been proven to be effective in reducing diffuse and point source pollution in several human-impacted catchments in China and the United States. Eco-ditches are engineered systems that are constructed ex novo or transformed from conventional agricultural drainage ditches. They are designed to mimic the depuration processes occurring in surface-flow wetlands and are considered an attractive alternative to traditional technologies for wastewater treatments (Chen et al., 2015; Xiong et al., 2015; Faust et al., 2018; Kalcic et al., 2018). In Italy, vegetated ditches have never been implemented as an agricultural best management practice (Provolo et al., 2016), as the study of pollutant mitigation in this type of aquatic ecosystem is just beginning. A few studies have investigated how processes and functions ascribed to the ditch networks may affect hydrochemical quality and ecosystem quality (e.g. biodiversity, resilience) on a broader scale (Bolpagni et al., 2013; Castaldelli et al., 2015; Otto et al., 2016; Soana et al., 2018). In recent years, some regions have tried to fill this knowledge gap by developing guidelines for ecological restoration of agricultural ditches. However, the qualitative nature of the provided indications and in particular the lack of parameterization regarding the self-depurative processes have led to a poor level of implementation of the suggested interventions, limited to a few canal stretches in the form of pilot actions (Bischetti et al., 2008; Emilia-Romagna Region, 2012).

In agricultural impacted watersheds, restoration or construction of wetlands is often impracticable due to limitations in available area and funds (Verhoeven, 2014; Hansen et al., 2018). However, ditches and canals are already present on croplands and their management could be aimed at harmonizing hydraulic functionality as well as ecological issues. Actions targeting $N\text{-}NO_3^-$ mitigation and sustainable agricultural practices might for example consider how to manipulate those widespread landscape elements through a design aimed at promoting biogeochemical processing, in particular creating conditions that maximize denitrification (e.g. organic matter availability, high water retention times) (Soana et al., 2017; Vymazal and Březinová, 2018; Schilling et al., 2018).

Uncertainty exists in the up-scale presented here, likely due to the high spatial and temporal heterogeneity of biogeochemical processes, which in turn are affected by several hydrological, geomorphological, and biological drivers. Nevertheless, scenario investigations clearly showed that vegetation restoration in ditch network of the Po River lowlands could be an effective tool to decrease $N\text{-}NO_3^-$ loads in surface waters, but only if implemented at a larger scale with respect to the present situation. Restoring vegetation on 90% of the ditch length, as prior to the introduction of mechanical mowing, appears unrealistic. However, the denitrification potential could be increased by identifying ditch stretches with appropriate chemico-physical features (e.g. Castaldelli et al., 2018; Soana et al., 2018) and low hydraulic risk. Here, interventions like reshaping and reduction of the slope banks could be carried out to restore suitable sites for macrophyte maintenance throughout the growing season. For example, our simulations demonstrated that a removal of up to 35-40% of the N surplus generated in the surrounding agricultural lands could be expected if vegetation is maintained in one quarter of the total ditch length. In order to preserve water transport capacity, canal sections would also be enlarged accordingly to the increase in hydraulic impedance due to the presence of vegetation. These actions would ultimately create a mosaic of heterogeneous habitats, and increase the water retention time and the surface area for microbial biofilms, which would maximize the depuration potential. Moreover, frequency and extension of the vegetation cutting could be planned to avoid complete and simultaneous removal of vegetation in all ditches, using maintenance practices that could ensure denitrification functionalities are maintained in time and space. Lastly, the effectiveness of restoration actions would be maximized if they are placed where the vegetated ditches intercept N surplus derived from manure or synthetic fertilizers since these are more prone to be leached towards surface waters than the organic N pool from N-fixing crops.

Further research is necessary to gain insights into the mechanisms underlying the depurative capacity of canals and ditches. The mitigation potential of vegetated ditch towards N has been less extensively studied compared to vegetated wetlands and the relative contribution of denitrification has been scarcely investigated, especially when assessed by the direct measurement of N_2 production (Taylor et al., 2015; Castaldelli et al., 2015; Speir et al., 2017). A more quantitative understanding of the processes accounting for N retention in vegetated ditches is crucial to develop management strategies to reduce eutrophication. Studies where the relative contributions of permanent N dissipation processes (nitrification coupled denitrification) and temporary storage mechanisms (plant uptake) are simultaneously assessed are still scarce for vegetated ditched (Castaldelli et al., 2015; Soana et al., 2017; Vymazal and Březinová, 2018). Conventional biogeochemical techniques (e.g. isotope pairing, acetylene inhibition) based on intact sediment core, microcosm or slurry approaches have been recently adopted to measure denitrification in ditches and have contributed to the understanding of specific controls of the process (Kröger et al., 2014; Soana et al., 2017; Veraart et al., 2017). However, severe uncertainty arises in scaling up data from the laboratory to the watershed, in particular measurements of potential denitrification activity where the structural integrity of sediment with associated biogeochemical gradients is altered and the direct influence of root activity is eliminated. Robust datasets should be collected, not only spanning a variety of environmental conditions (e.g. hydraulic parameters, solute concentrations, plant type), but also at spatial and temporal scales relevant to N pollution issues and appropriate to be extrapolated up to the watershed level. The parameterization of the N-NO_3^- mitigation capacity should be obtained by the application of whole-reach approach (e.g. N_2 open-channel) that capture small-scale spatial and temporal heterogeneity of environmental conditions and metabolic processes occurring in shallow aquatic ecosystems, such as vegetated ditches, where multiple riverine habitats and interfaces exist (i.e. biofilms, oxic-anoxic microniches in the rhizosphere) (Castaldelli et al., 2015; Taylor et al., 2015; Soana et al., 2017). The actual efficiency of small artificial waterways to buffer N pollution can be really appreciated if they are considered as a whole, i.e. sediment, water, vegetation, biofilms and their multiple interactions. Reach-scale methods integrate water column and benthic compartment dynamics and overcomes the limitations inherent in the upscaling of results from the laboratory to the field (e.g. measurements performed over small surfaces, incubation artifacts, etc.) (Reisinger et al., 2016; O'Brien et al., 2017). The scientific outcomes will be instrumental to produce predictive tools and management guidelines aimed at maximizing the natural depuration capacity of the ditch network in agricultural landscapes.

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Tables

Table 1. Main chemico-physical and hydraulic features of vegetated and unvegetated ditches of the Po River lowlands for which experimentally measured values of reach-scale N-NO_3^- removal rate are available. These datasets were acquired in previous experimental campaigns (sampling events along the irrigation period: N=44 for vegetated sites and N=61 for unvegetated sites) reported by Pierobon et al. (2013) and Castaldelli et al. (2015), and were used to build the N-NO_3^- concentrations–N-NO_3^- removal relationships shown in Fig. 2.

		Vegetated ditches		**Unvegetated ditches**	
		Median value	Range	Median value	Range
Water T	(°C)	23	17 – 31	23	16 – 29
N-NO_3^-	(mg L^{-1})	1.17	0.01 – 7.94	1.20	0.71 – 7.05
DO	(mg L^{-1})	7.23	3.34 – 13.86	7.68	6.23 – 8.24
Discharge	(L s^{-1})	31	5 – 158	39	3 – 171

Table 2. N budget in the lowland of the Po River below 50 m a.s.l. Data are expressed as tons of N produced or consumed per year (t yr^{-1}) in the whole area and normalized for the utilized agricultural area (kg ha^{-1} yr^{-1}). The percentage of each input or output with respect to the total is reported in brackets.

	N budget			
	Average	Min – Max	Average	Min – Max
INPUT	(t yr^{-1})	(t yr^{-1})	(kg ha^{-1} yr^{-1})	(kg ha^{-1} yr^{-1})
Livestock manure	48,838 (30)	39,554 – 58,123	77.8	63.0 – 92.6
Synthetic fertilizers	55,245 (34)	54,140 – 56,350	88.1	86.3 – 89.8
Biological fixation by N-fixing crops	45,815 (28)	29,486 – 65,395	73.0	47.0 – 104.2
Biological fixation by natural surfaces*	7,971 (5)	4,042 – 12,297	12.7	6.4 – 19.6
Atmospheric deposition	5,019 (3)	4,266 – 5,772	8.0	6.8 – 9.2
Σ input	**162,889**	131,488 – 197,937	**259.6**	209.6 – 315.5
OUTPUT				
Harvest by feed crops	70,231 (60)	44,832 – 100,859	111.9	71.5 – 160.8
Harvest by food crops	24,305 (21)	15,649 – 34,679	38.7	24.9 – 55.3
NH_3 volatilization	11,763 (10)	1,430 – 24,986	18.7	2.3 – 39.8
Denitrification in agricultural soils	10,408 (9)	4,685 – 17,171	16.6	7.5 – 27.4
Σ output	**116,707**	66,596 – 177,695	**186.0**	106.1 – 283.2
Σ input – Σ output = Surplus	**46,182**	20,242 – 64,893	**73.6**	32.3 – 103.4

*The term includes N fixed by permanent grassland and pastures and by non-symbiotic process in arable land and permanent crops.

Figure captions

Fig. 1. Study area: A) location of the Po River basin in Europe; B) location of the Po plain below 50 m a.s.l. within the Po River basin; C) land use (Corine Land Cover inventory 2012) and D) hydrographic network of the studied area (Bing Aerial Maps Baselayer for QGIS, www.bing.com/maps).

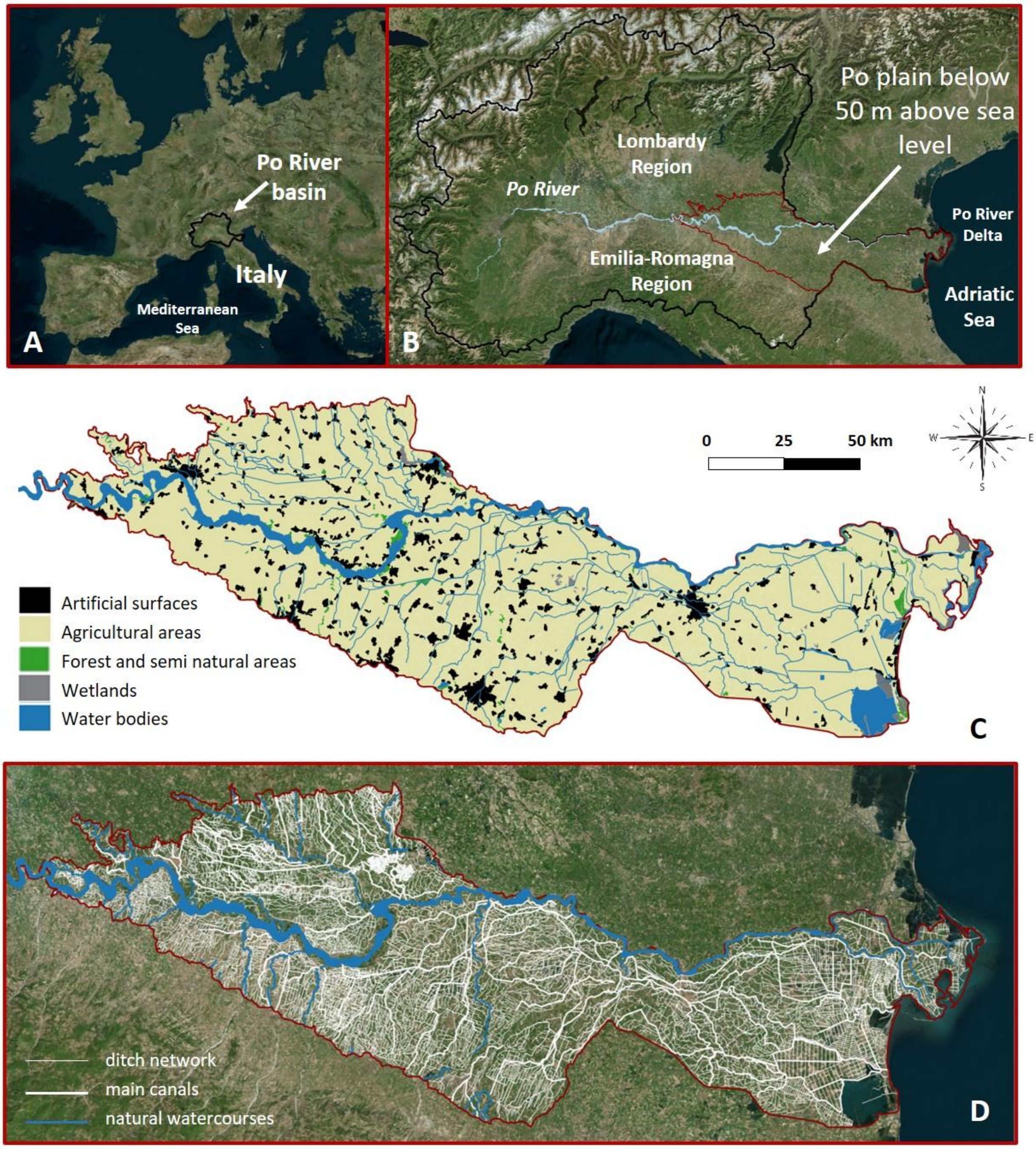
Fig. 2. Relationship between incoming N-NO_3^- concentrations (mg L^{-1}) and reach-scale N-NO_3^- removal rates (kg km^{-1} d^{-1}) measured along the irrigation period in vegetated (panel A) and unvegetated ditches (panel B) of the Po River lowlands. The regression line (solid line) is presented along with the 95% confidence interval (dotted line). Data from Pierobon et al. (2013) and Castaldelli et al. (2015).

Fig. 3. Log-log relationship between reach-scale N-NO_3^- removal and N-N_2 production rates (g km^{-1} d^{-1}) in vegetated ditch sediments obtained by the simultaneous application of in-out NO_3^- budget and N_2 open channel method. The regression line (solid line) is presented along with the 95% confidence interval (dotted line). Data from Castaldelli et al. (2015), Castaldelli et al. (2018) and Soana et al. (2018).

Fig. 4. Seasonal variations of water T (°C, panel A), N-NO_3^- (mg L^{-1}, panel B), DO (mg L^{-1}, panel C), and BOD_5 (mg L^{-1}, panel D) in the ditch network of the Po River lowlands. Box and Whisker plots include monthly data (period 2009-2014) for 70 stations located within the 50 m a.s.l. area and belonging to official surface water monitoring network of the Regional Agency for the Environmental Protection of Lombardy and Emilia-Romagna Regions. Central horizontal line in the box is the median, top and bottom boxes are 25^{th} and 75^{th} percentiles, and whiskers are 10^{th} and 90^{th} percentiles. Outliers are showed as open circles.

Fig. 5. Seasonal variations of N-N_2 production rates (kg km^{-1} d^{-1}) predicted by the model proposed by Christensen et al. (1990) for the ditch network of the Po River lowlands. Box and Whisker plots include the predicted rates for 70 stations located within the 50 m a.s.l. area and belonging to official surface water monitoring network of the Regional Agency for the Environmental Protection of Lombardy and Emilia-Romagna Regions. Christensen model was applied for all the official monthly surveys (period 2009-2014) for which values of water T, DO and N-NO_3^- were concomitantly available. Central horizontal line in the box is the median, top and bottom boxes are 25^{th} and 75^{th} percentiles, and whiskers are 10^{th} and 90^{th} percentiles. Outliers are showed as open circles.

Fig. 6. Predicted N-NO_3^- removal (t yr^{-1}) with vegetation restoration on variable extension of the ditch network (5%, 25%, 50% and 90% of the total ditch network length). Different water N-NO_3^- availability together with two options of vegetation management were simulated, i.e. the *current management* (mowing performed in the middle of the summer, panel A) and the *conservative management* (mowing performed at the end of the growing season, panel B). The corresponding percentage of N surplus in the agricultural land is also reported.


Po River
basin
Italy
Mediterranean
Sea
A
Po plain below
50 m above sea
level
Lombardy
Region
Po River
Emilia-Romagna
Region
Po River
Delta
Adriatic
Sea
B
0
25
50 km
N
W
E
S
Artificial surfaces
Agricultural areas
Forest and semi natural areas
Wetlands
Water bodies
C
ditch network
main canals
natural watercourses
D

862

863 Fig. 1

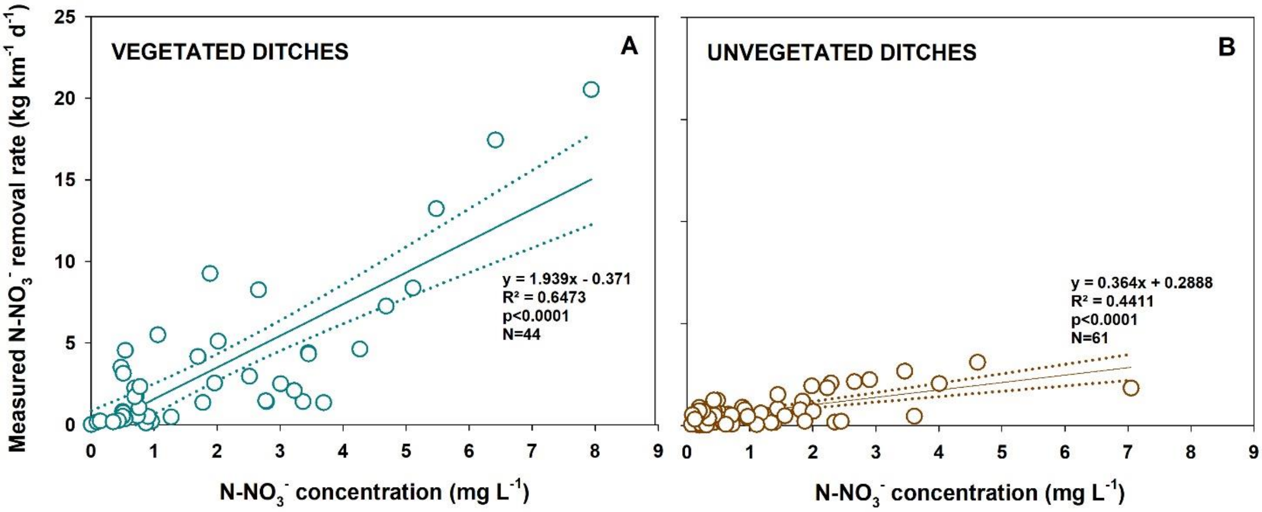
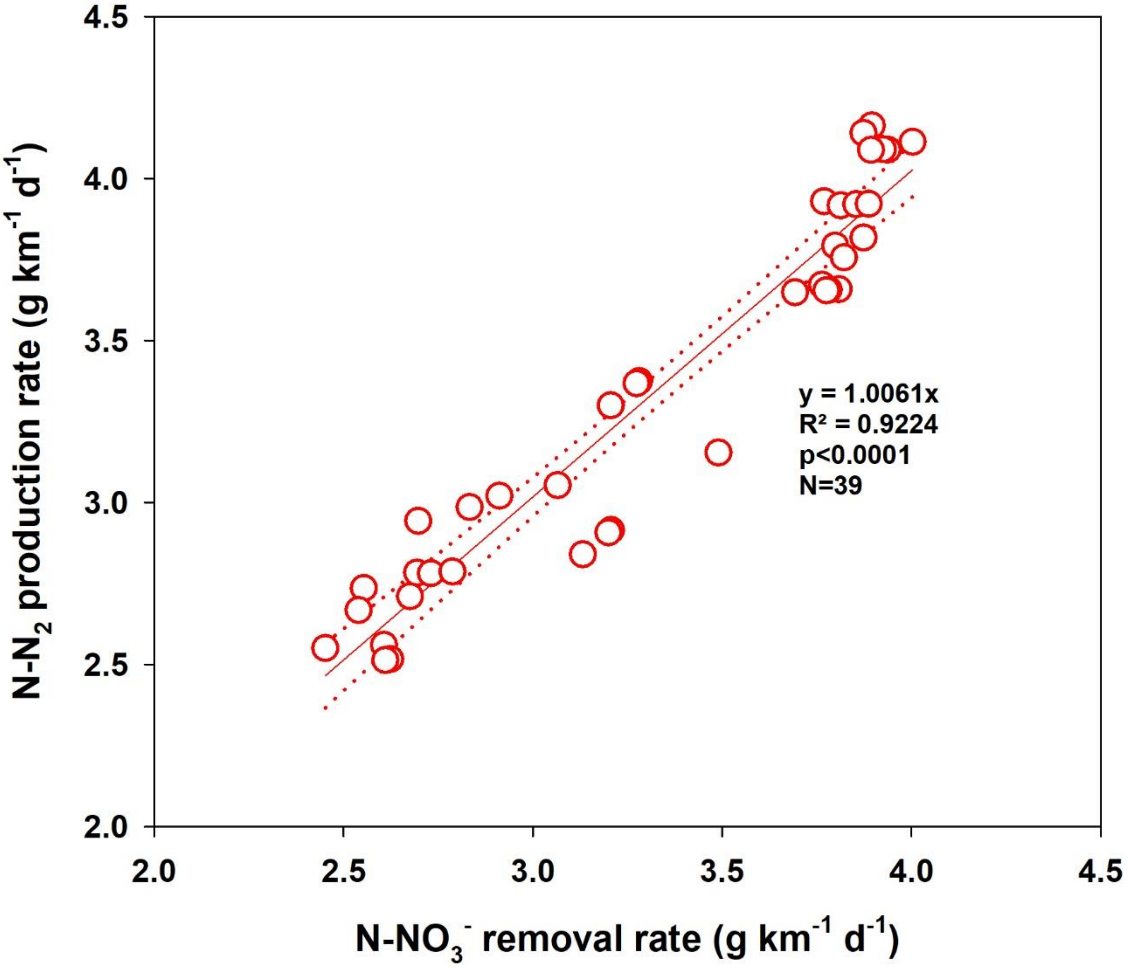

VEGETATED DITCHES
A
y = 1.939x - 0.371
R² = 0.6473
p<0.0001
N=44
UNVEGETATED DITCHES
B
y = 0.364x + 0.2888
R² = 0.4411
p<0.0001
N=61
Measured N-NO3- removal rate (kg km-1 d-1)
N-NO3- concentration (mg L-1)
N-NO3- concentration (mg L-1)

Fig. 2


4.5
4.0
3.5
3.0
2.5
2.0
N-N2 production rate (g km-1 d-1)
y = 1.0061x
R² = 0.9224
p<0.0001
N=39
2.0
2.5
3.0
3.5
4.0
4.5
N-NO3- removal rate (g km-1 d-1)

866 Fig. 3

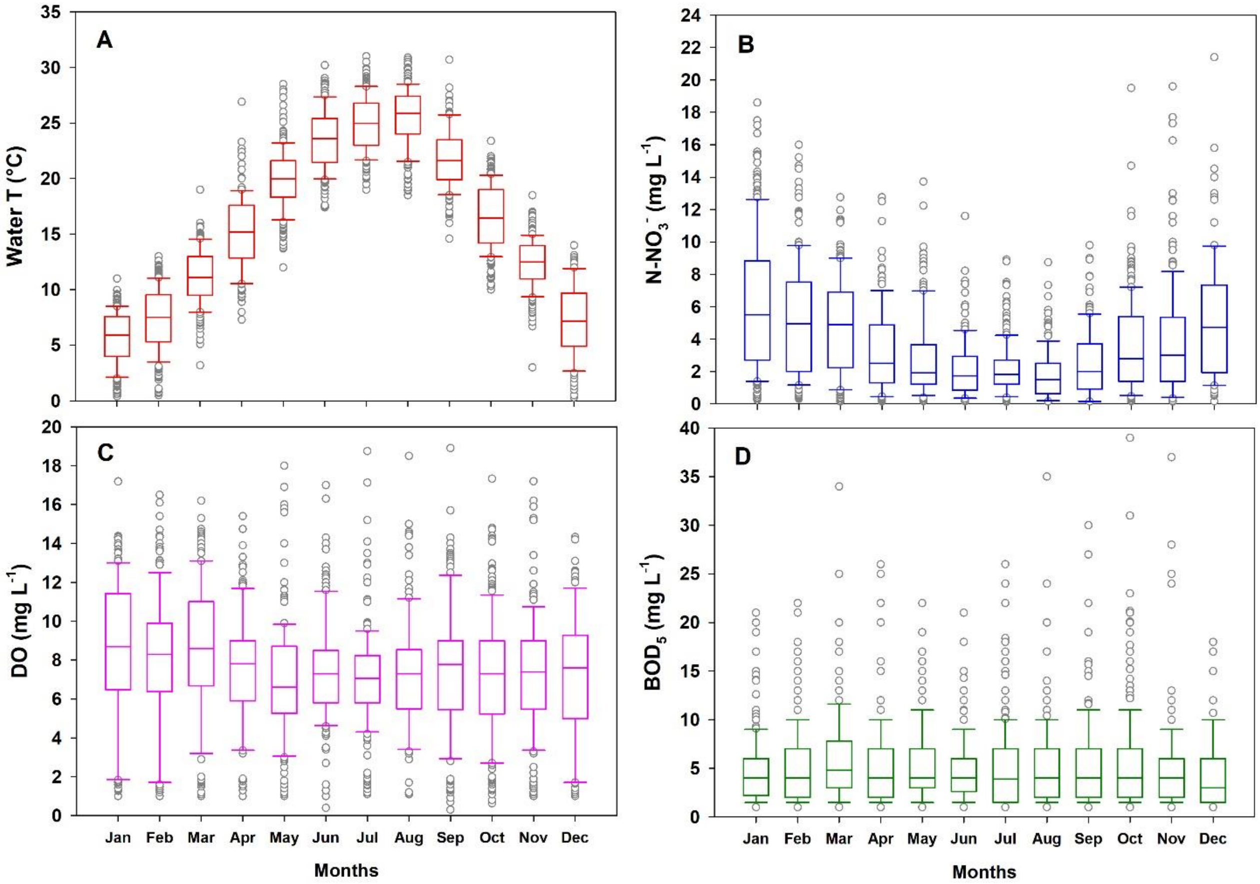
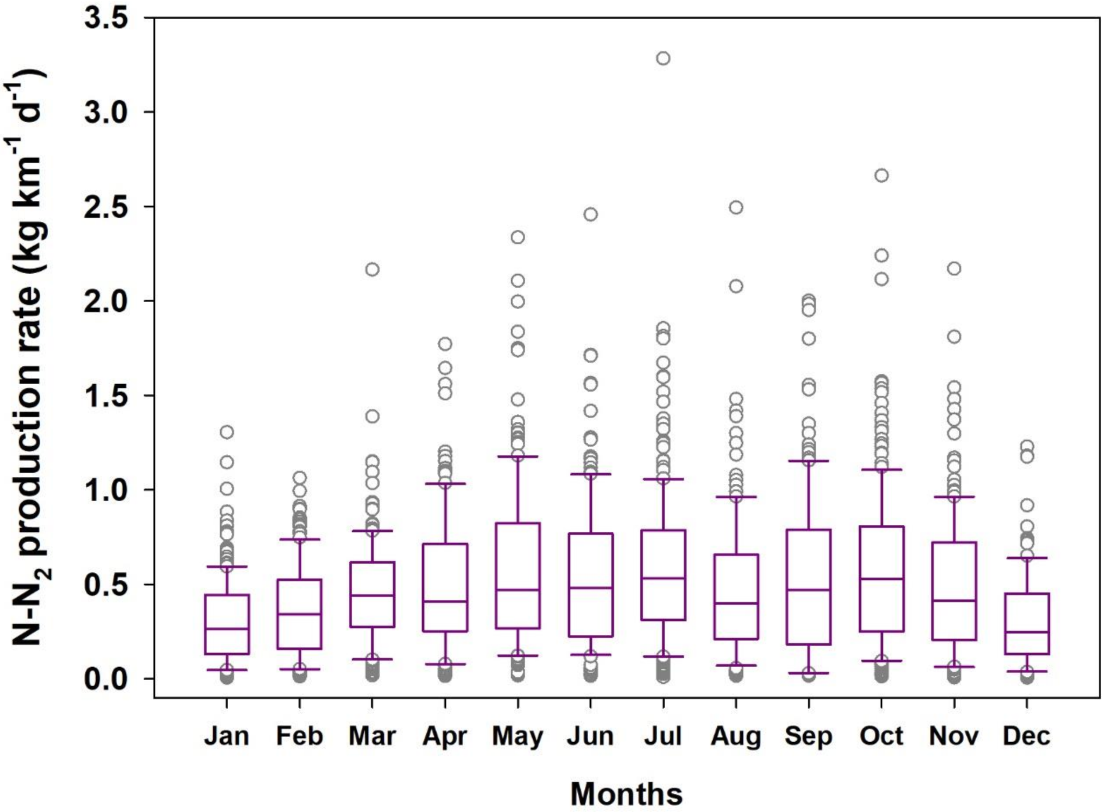

A
Water T (°C)
B
N-NO3- (mg L-1)
C
DO (mg L-1)
D
BOD5 (mg L-1)
Jan Feb Mar Apr May Jun Jul Aug Sep Oct Nov Dec
Months
Jan Feb Mar Apr May Jun Jul Aug Sep Oct Nov Dec
Months

Fig. 4


N-N_2 production rate (kg km^{-1} d^{-1})
3.5
3.0
2.5
2.0
1.5
1.0
0.5
0.0
Jan
Feb
Mar
Apr
May
Jun
Jul
Aug
Sep
Oct
Nov
Dec
Months

869 Fig. 5

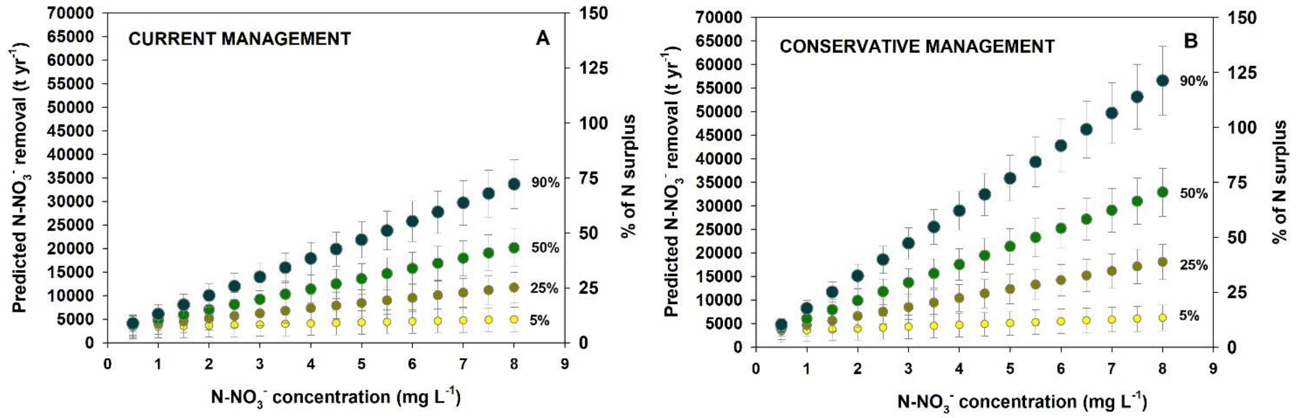

CURRENT MANAGEMENT
A
Predicted N-NO$_3^-$ removal (t yr^{-1})
% of N surplus
N-NO$_3^-$ concentration (mg L^{-1})
90%
50%
25%
5%
CONSERVATIVE MANAGEMENT
B
Predicted N-NO$_3^-$ removal (t yr^{-1})
% of N surplus
N-NO$_3^-$ concentration (mg L^{-1})
90%
50%
25%
5%

Fig. 6