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Looking back to move forward: Restoring vegetated canals to meet missing Water Framework Directive goals in agricultural basins

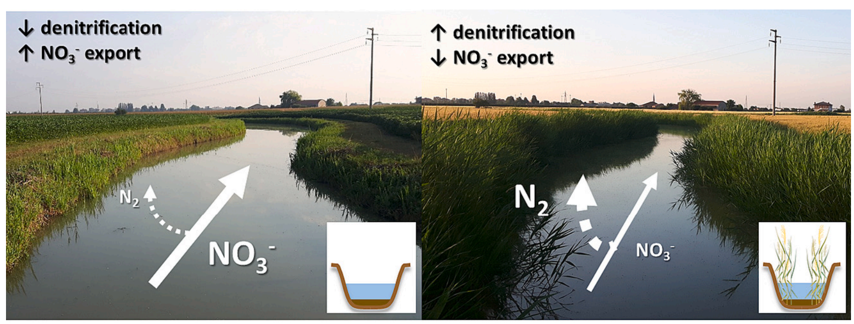
Elisa Soana [a,*], Anna Gavioli [a], Federica Neri [b], Giuseppe Castaldelli [a]

[a] Department of Environmental and Prevention Sciences, University of Ferrara, Via Luigi Borsari 46, 44121 Ferrara, Italy
[b] Ferrara Plain Reclamation Consortium, Via Borgo dei Leoni, 28, 44121 Ferrara, Italy

HIGHLIGHTS

- Nitrogen removal potential of vegetated canals was assessed in an irrigated basin.
- Postponing vegetation mowing contributes to lower NO_3^- export in spring and summer.
- Vegetated canals help meet NO_3^- reduction targets in agricultural basins.
- Restoration of in-stream vegetation is an effective strategy to improve water quality.

GRAPHICAL ABSTRACT



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ABSTRACT

Nitrate pollution and eutrophication remain pressing issues in Europe regarding the quality of aquatic ecosystems and the safety of drinking water. Achieving water quality goals under the Water Framework Directive (WFD) has proven to be particularly challenging in agricultural catchments, where high nitrate concentrations are the main reason for the failure of many water bodies to meet a good ecological status. Canals and ditches are common man-made features of irrigated and drained landscapes and, when vegetated, have recently been identified as denitrification hotspots.

By combining experimental data and GIS-based upscaling estimation, the potential capacity of the canal network to reduce nitrate loads was quantified in several scenarios differing in the level of nitrate pollution and in the extent of the canal network length where conservative management practices are implemented. The analysis was carried out in the irrigated lowlands of the Po River basin, which is the largest hydrographic system in Italy and a global hotspot for nitrogen inputs and eutrophication.

Scenario simulations showed that maintaining aquatic vegetation in at least 25 % of the canal network length, selecting sites with high nitrate availability (>2.4 mg N L^{-1}), would promote a greater potential for permanent N removal. The increased denitrification capacity would meet the load reduction target required to achieve a WFD good ecological status in waters draining into the Adriatic Sea during the spring-summer months, when the eutrophication risk is higher. Promoting denitrification in the canal network by postponing the mowing of in-

* Corresponding author.
E-mail address: elisa.soana@unife.it (E. Soana).

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stream vegetation to the end of the growing season could be an effective mitigation strategy to improve water quality in agricultural basins and contribute to achieving the WFD goals.

1. Introduction

Over the last century, agriculture has profoundly altered the nutrient balance of cropland and is now globally recognized as a major source of diffuse pollution, especially nitrogen (N), in waters (Battye et al., 2017; Galloway et al., 2021). Excess nitrate (NO_3^-) from agriculture contributes to the deterioration of surface and groundwater quality and has been identified as a serious threat to human health by contaminating drinking water and to aquatic ecosystems through eutrophication (Le Moal et al., 2019; Nieder and Benbi, 2022). Agricultural activities and food production will likely continue to follow the intensification paradigm to meet the increasing population by the middle of the 21st century, with negative ecological impacts in terms of water pollution, greenhouse gas emissions, ecosystem functioning, and biodiversity loss (Stoate et al., 2009; Leip et al., 2015; Beltran-Peña et al., 2020). In recent decades, a number of environmental policies, such as the Nitrates Directive (91/676/EEC) and the Water Framework Directive (WFD, 2000/60/EC), have been adopted in Europe to promote good agricultural practices aimed at controlling NO_3^- losses from agriculture to water bodies, i.e., optimising the use and application of fertilizer to better match crop needs, introducing cover crops and treating animal waste (Velthof et al., 2014; Garnier et al., 2016; Malagó et al., 2019). Since the 1990s, NO_3^- concentrations in many European rivers have steadily decreased because of reduced fertilizer inputs (Vigiak et al., 2023). However, the apparent stabilisation in recent years calls for further measures to be taken as concentrations still do not meet the environmental targets needed to ensure water quality and the functioning of aquatic ecosystems (Bouraoui and Grizzetti, 2011; Vigiak et al., 2021). High NO_3^- concentrations remain a pressing issue in Europe and are in many cases the main reason for the failure of surface waters to achieve a good ecological status according to the WFD. According to the most recent estimate, more than half of the European rivers still deviated from a good ecological status (EEA, 2018; Poikane et al., 2019), and there are concerns that the new deadline of 2027 will be missed again (Vigiak et al., 2021). Nitrogen legacy, i.e., storage in groundwater as NO_3^- and in soils as organic N, may make it impossible to achieve good water quality goals in the short to medium term (Basu et al., 2022) and may become increasingly difficult also in the context of climate change (Glibert, 2020; Costa et al., 2022).

In recent decades, agricultural, industrial, and urban development, together with hydro-morphological modifications, have contributed to the landscape simplification in human-impacted basins, which have lost the aquatic ecosystems acting as natural nutrient filters (Beaulieu et al., 2015; Frascaroli et al., 2021; Cuenca-Cambronero et al., 2023). Simultaneously, agricultural catchments have progressively lost a key component of their buffer capacity against NO_3^- pollution, namely the aquatic vegetation. Eutrophication has caused the decline or dieback of submerged macrophytes in several freshwater ecosystems worldwide, owing to a combination of increased turbidity in the water column and chemical conditions in the interstitial water that have become hostile to roots (McGlathery et al., 2007; Ribaudo et al., 2023).

The sustainable intensification of agriculture and a 50 % reduction in nutrient losses have been recently introduced as targets by the European Green Deal in the Farm to Fork Strategy and in the Biodiversity Strategy 2030 (European Commission, 2020, 2021). Mitigating NO_3^- pollution and controlling eutrophication will require both strengthening existing good agricultural practices and water policies (Grizzetti et al., 2021), and testing and implementing new remediation measures, such as the restoration of biogeochemical processes and functions in aquatic ecosystems. Nevertheless, this approach is generally economically unsustainable, time consuming and ineffective for streams and rivers due to the size of such water bodies (Petersen et al., 2021; Zhou et al., 2023) and needs to target the most metabolically reactive zones along the land-sea continuum that are capillary distributed across the landscapes.

Eutrophication is a relevant and long-standing water quality problem in the Po River Basin (Italy), one of the most agriculturally productive areas in Europe, generating approximately 40 % of Italy's Gross Domestic Product (Viaroli et al., 2018; Po River District Authority, 2021). The implementation of environmental policies and legislation aimed at controlling nutrient emissions from point sources (e.g., the construction of wastewater treatment plants, and bans on polyphosphates in detergents) has led to significant reductions in nutrient loads from urban areas (de Wit and Bendoricchio, 2001; Palmeri et al., 2005). However, over the last three decades, the preventive and remedial measures introduced by the European Directives to control widespread agricultural and livestock sources have been largely ineffective and NO_3^- pollution has become a major concern for water quality (Viaroli et al., 2018; Musacchio et al., 2020).

In agricultural landscapes, such as the Po Plain, agricultural canals and ditches are ubiquitous man-made linear elements that comprise most of the hydrographic networks and, as ecotones, play a crucial role in intercepting N leaching from agricultural land (Goeller et al., 2020; Moore and Locke, 2020; Rizzo et al., 2023). Canals and ditches have traditionally been designed, constructed, and managed for their surface drainage and irrigation water transfer functions (Dollinger et al., 2015; Rudi et al., 2020). Recently, complementary functions of nutrient buffer zones have been identified, making canal networks the quantitatively most important metabolic reactors in agricultural landscapes (Lassaletta et al., 2012; Romero et al., 2016; Soana et al., 2019; Hill, 2023). Research over the last decade has shown that canals are more than simple conduits for water transfer, but act as natural wetlands, providing key biogeochemical services, such as N removal through denitrification, i.e., microbial anaerobic respiration of NO_3^- (Castaldelli et al., 2015; Kumwimba et al., 2023; Shen et al., 2021).

The presence and abundance of aquatic macrophytes are considered key factors in determining the water depuration potential and, in particular, the removal of excess NO_3^-, through a complex synergistic action with bacterial communities (Veraart et al., 2017; Soana et al., 2020; Taylor et al., 2020). However, the hydraulic efficiency of irrigation and drainage has been enabled by the straightening of canal networks and, in most cases, this has corresponded to the quantitative removal of aquatic vegetation (Errico et al., 2019; Kalinowska et al., 2023).

Although canals and ditches have received increasing attention as biogeochemical hotspots in recent years, the full impact of these artificial ecosystems on watershed N dynamics is not yet fully understood, nor is their effectiveness as a management option to reduce N excess in agricultural watersheds and to meet nutrient reduction targets. There is a need to assess whether conservative vegetation-preserving management practices can help achieve surface water quality targets by exploiting the natural depuration capacity of vegetated canals, which can be considered as nature-based solutions (Mancuso et al., 2021; Ferreira et al., 2023). To support and guide their implementation on a wider scale, a better understanding of how these edge-of-field tools can improve inland and coastal water quality is needed.

The aim of the present study was to compare the treatment performance of the canal network in the Po River lowland with the reduction target required to achieve a good ecological status for NO_3^-, as defined by the WFD, in the drainage waters delivered to the Adriatic Sea. Through a combination of experimental data and GIS-based upscaling estimation, the potential capacity of the canal network to reduce NO_3^- loads was quantified in several scenarios that differed in the level of NO_3^-

pollution and in the percentage of the canal network length where conservative management practices are implemented.

2. Material and methods

2.1. Study area

The study area covers the Po Plain that surrounds the river for the last 250 km of its course, from the city of Cremona to the Delta in the Adriatic Sea, within the administrative boundaries of three Italian Regions (Lombardy, Emilia-Romagna, and Veneto) (Fig. 1A, B). The territory (~6300 km^2) includes the last part of the largest Italian alluvial plain, at an altitude of <50 m above sea level, and where crops are almost exclusively irrigated with water withdrawn from the Po River. The study area was delimited using information on the location of water diversion points and drainage facilities and the boundaries of the irrigation districts provided by the Land Reclamation Authorities management plans.

The climate is warm temperate (Type Cfa, Köppen–Geiger classification; Beck et al., 2018) with an average annual precipitation of 700–800 mm and peaks in spring and autumn (Vezzoli et al., 2015). Land use is predominantly agricultural (>85 % of the total surface, Fig. 1), dominated by wheat, fodder (silage maize, alfalfa, and temporary grassland), industrial and horticultural crops, fertilized with both synthetic compounds and livestock manure (National Institute of Statistics, 2022a). Following the adoption of the European Directives (91/676/EEC and 2000/60/EC), >50 % of the study area has been declared as nitrate vulnerable zone from agricultural sources (European Commission, 2022), mainly covering the deltaic sub-basin of the Burana–Po di Volano–Navigabile (BVN, 41 % of the total study area) and the territory on the left bank of the Po River. <3 % of the study area was classified as forest and semi-natural surfaces, whereas artificial surfaces

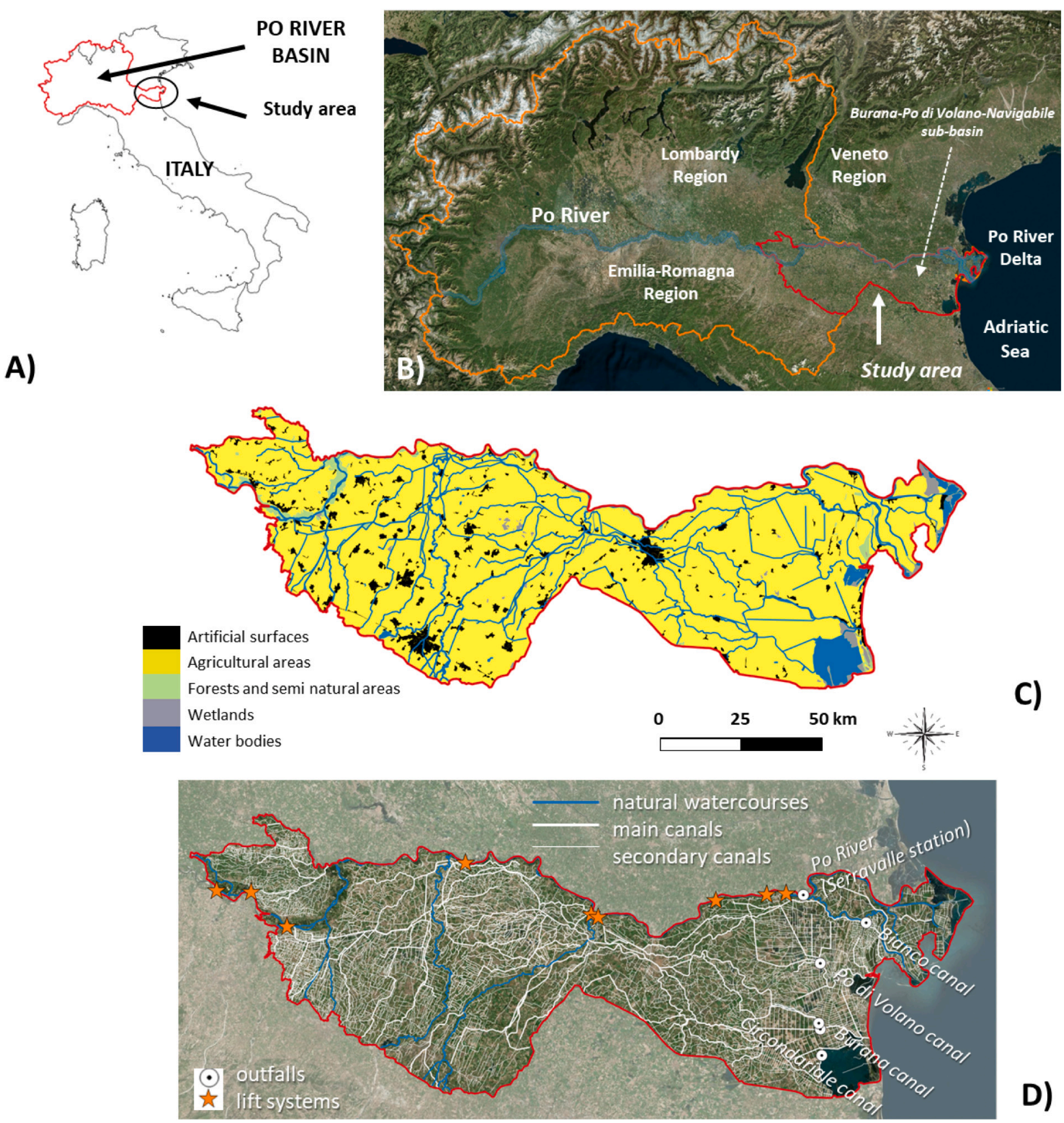

Fig. 1. Study area: a) Po River basin; b) location of the study area (red) within the Po River basin (orange); c) land use (Corine Land Cover inventory 2018); d) hydrological network (Bing Aerial Maps Baselayer for QGIS, Microsoft, USA).

covered approximately 6 % of the study area (Fig. 1C). The urban pattern is dispersed, with the population concentrated in a few medium-sized cities (Reggio nell'Emilia, Modena, and Ferrara) and many small towns (National Institute of Statistics, 2022b).

The Po River lowland is crossed by a capillary network of artificial canals and ditches built over the centuries to distribute water for irrigation, to buffer fluctuations in the river's flow and to minimize the hydraulic risk to several towns and villages along the main river course. The hydrological structure of the canal network is the result of a long reclamation process that began with the ancient Etruscans and continued until the 1960s (Marchetti, 2002; Frascaroli et al., 2021). The drainage system, which is managed by six Land Reclamation Authorities (*Consorzi di Bonifica* in Italian), consists of a network of approximately 13,000 km of canals (average density of ~2.5 km per km^2 of agricultural land, Fig. 1D), hundreds of pumping stations and thousands of hydraulic adjustment structures that are in constant operation to ensure hydraulic safety conditions for the entire territory and water supply for agriculture from May to September. The water requirements of the crops during the spring and summer months are met by irrigation water supplied almost exclusively by the Po River (>90 % of the total volume, Ferrara Plain Reclamation Consortium, personal communication), which enters the irrigated area through lift systems and mechanically controlled water diversion points on both the right and left banks (Fig. 1D), and are then distributed over the farmland thanks to a large number of hydraulic works scattered throughout the area. Several drainage systems pump the excess water back into the Po River, or into the end sections of two tributaries to the Po River (Oglio River and Secchia River), or directly into the final receptor (the Adriatic Sea) through several collectors draining the BVN basin. From the end of September, large parts of the canal network are left dry until the next irrigation season and are used for flood control in the event of heavy rainfall.

2.2. Vegetation management in the canal network

The Land Reclamation Authorities routinely perform maintenance operations on the canal network (e.g., vegetation cutting, dredging, section reshaping, and bank reinforcing) to optimise water conveyance capacity. In the 1970s and 1980s, when severe eutrophication phenomena occurred (Palmeri et al., 2005; Viaroli et al., 2018), the overgrowth of aquatic vegetation caused hydraulic problems in the canal network of the Po River lowland, thus the local water authorities initiated extensive efforts to mechanically remove it. Since the late 1980s, with the introduction of mower arms and mowing buckets, aquatic vegetation has been much more efficiently controlled and, in some cases, completely removed from the bottom and the banks of the canals. Mowing is carried out two or three times a year, the first generally June (Fig. 2), in some cases repeated at the end of July, and in all the canals in October, before the heavy rains and the increase in hydraulic risk. As a result, the canal network of the Po River lowland is currently almost completely unvegetated, except for isolated stands of *Phragmites australis* (Cav.) Trin. Ex Steud. And *Typha latifolia* L., which are maintained in <5 % of the network, in sections that are difficult to reach with hydraulic excavators and are not crucial for hydraulic functions (Pierobon et al., 2013; Soana et al., 2019; Tamburini et al., 2020).

2.3. Riverine loads exported from the Po River basin: baseline vs good ecological status for NO_3^-

Nitrate loads exported to the Adriatic Sea were calculated using databases of flows and NO_3^- concentrations collected over the last two decades (period 2000–2021) by the Environmental Protection Agency of the Emilia-Romagna Region (ARPAE) and the Ferrara Plain Reclamation Consortium, the local authority managing the drainage system of the BVN sub-basin. The total NO_3^- transport to the sea was obtained by summing up the load conveyed at the main outfall, the course of the Po River just before the beginning of the Po Delta (Serravalle station, 44°58′43.6″ N, 11°59′51.9″ E), and at four minor outfalls draining the BVN sub-basin (Fig. 1D).

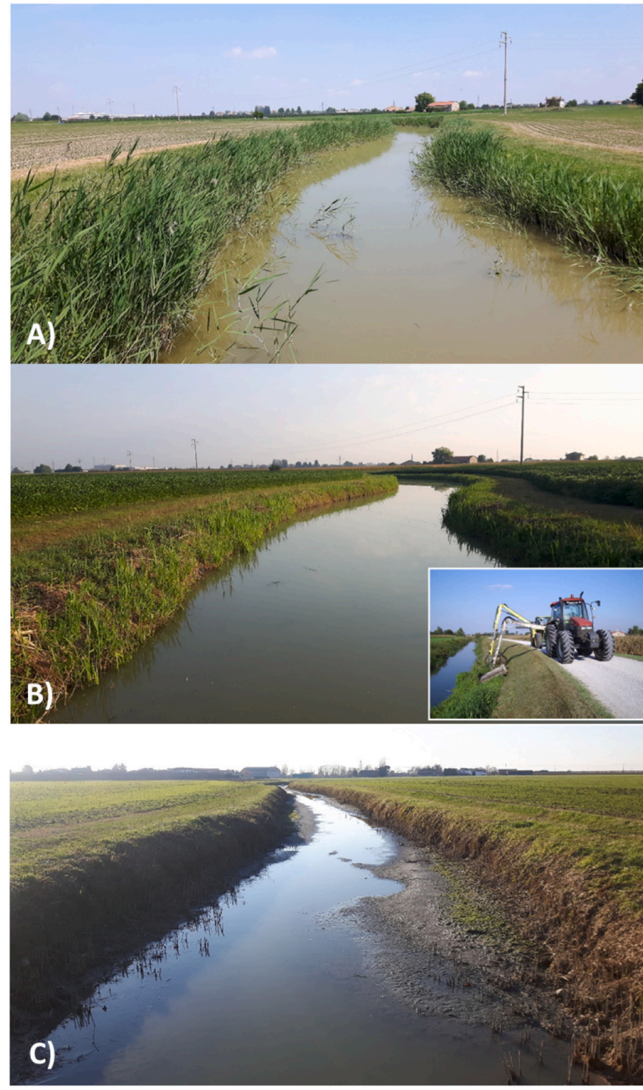

Fig. 2. Typical seasonal evolution of a canal in the Po River lowland (Brusabò Basso canal, Gradizza, Ferrara; photo by Anna Gavioli). Beginning of the irrigation period (A), mid-summer (C), non-irrigation period (C).

Measured daily flow at Serravalle gauge station were retrieved from the Hydrological Annals published by ARPAE-SIMC (Regional Hydro-Meteo-Climatological Service of the Emilia-Romagna Region) in the Open Data Portal (ARPAE-SIMC, 2021). The river water level was measured at a resolution of 15 min and then converted into flows using a rating curve specific for the cross section of interest (Domeneghetti et al., 2012). Daily flows for minor outfalls draining the BVN sub-basin were calculated from raw data (i.e., pumping station operating time and hydraulic characteristics of the pumps) collected by the Ferrara Plain Reclamation Consortium through a continuous and real-time data acquisition system.

Datasets of NO_3^- concentrations were collected monthly (or bimonthly in a few cases), for five stations, one on the Po River (Serravalle) and four located on the main artificial canals draining the BVN basin (Fig. 1D). The stations belong to the ARPA network of sampling sites for monitoring the ecological status of the surface water bodies in the framework of the WFD implementation (Emilia-Romagna Region, 2021a). Sample collection and analytical determinations were performed according to standard methods and analytical protocols outlined by national environmental regulations and adopted by the Regional

Environmental Agencies in Italy (APAT-IRSA/CNR, 2003). Nitrate was measured on filtered water samples (Whatman GF/F filters) using a standard colorimetric method based on the azo dye reaction (detection limit >0.2 mgN L^{-1}, precision ±3 %).

Nitrate load is the product of flow and NO_3^- concentration integrated over time. For the study system, flow was measured continuously with daily resolution, while concentration data were collected at monthly (for the Po River) or bimonthly (in some canals of the BVN basin) frequency. The NO_3^- concentration of non-sampled days was established by linear interpolation between sampled concentrations (Moatar and Meybeck, 2005; Nava et al., 2019). The interpolation procedure assumes that the concentration values obtained from instantaneous samples (i.e., NO_3^- concentrations measured once or twice a month) are representative of a much longer period. Previous studies have demonstrated that this method is accurate and precise for assessing large rivers with recurring seasonal variations in NO_3^-concentrations (Moatar and Meybeck, 2005; Viaroli et al., 2018). The procedure was validated for the Po River by Gervasio et al. (2022) and Soana et al. (2023). Daily loads were calculated as the product of the measured daily flow and NO_3^- concentration (measured or interpolated), and then aggregated to a monthly scale (t N $month^{-1}$) using the following equation:

$$NO_3^- \text{load} = k \bullet \sum_{j=1}^{n} C_i \bullet Q_I \tag{1}$$

where:

C_i is the daily NO_3^- concentration (mg N m^{-3}) on day *i*, measured or linearly interpolated between two consecutive sampling events to represent unsampled days; Q_i is the average daily flow ($m^3\ s^{-1}$) on day *i*; n is the number of days in a month, and k is the unit conversion factor (mg s^{-1} to t d^{-1}, 86.4 × 10^{-6}). The annual NO_3^- load was calculated by summing all monthly loads within each year and separating the contribution of the irrigation period (from May to September) and the non-irrigation period (from October to April).

The target NO_3^- loads exported by the Po River basin were calculated using time series of daily flows and concentrations capped at a threshold of 1.2 mg N L^{-1}. This concentration is the upper limit of the "good ecological status" according to the LIMeco index (Macro-descriptors Pollution Level for the Ecological Status) (Erba et al., 2022), developed for the classification of the trophic status of Italian watercourses according to the WFD (Table S1 and Table S2, Supplementary Material 1). In practice, a fixed value of 1.2 mg N L^{-1} was set when the measured NO_3^- concentrations exceeded the upper limit of the "good" class. This concentration approximates the threshold (~1 mg N L^{-1}) proposed to prevent the risk of eutrophication in freshwaters and to protect sensitive freshwater species from nitrate toxicity (Camargo and Alonso, 2006; Grizzetti et al., 2011).

2.4. GIS-based upscaling estimation of NO_3^- removal capacity: from field measurements to the canal network scale

The denitrification model was based on large datasets of in situ measurements collected in several vegetated canals and ditches of the Po River lowland during the irrigation season (from May to September). The canal network in the study area is almost completely unvegetated due to current management, so the experimental measurements were taken on pilot sites that were kept temporarily vegetated as part of research projects. The following empirical relationship was developed:

$$Dv = \frac{82 \bullet [NO_3^-]}{25 + [NO_3^-]} \tag{2}$$

where:

D_V = daily rates of denitrification (i.e., NO_3^- removal) in vegetated sediments (kg N km^{-1} d^{-1}).

$[NO_3^-]$ = water NO_3^- availability (mgN L^{-1}).

The regression equation predicting NO_3^- removal rates in vegetated sediments as a function of water NO_3^- concentration was obtained (Fig. 3A; $R_{adj}^2 = 0.85$; $p < 0.001$) by pooling field measurements (n = 48) from previous experimental campaigns using a reach-scale in-out NO_3^- balance approach (Pierobon et al., 2013; Castaldelli et al., 2015; Castaldelli et al., 2018; Soana et al., 2020). The investigated canals were representative of the dominant agricultural waterways of the Po River lowlands (e.g., hydraulic regime, maintenance operations, bottom sediment features, and vegetation density), thus the extensive datasets of denitrification rates were reasonably robust for use in the upscale model extended to the entire canal network. Denitrification rates were estimated using whole-system methods that integrate N processes occurring in different compartments (i.e., sediment, biofilms, and water column) and under natural conditions. This overcomes the limitations associated with the upscaling of results from laboratory experiments (Castaldelli et al., 2015; Soana et al., 2019).

When water temperature is >20 °C, the presence of emergent vegetation and water NO_3^- availability have been shown to be the most important factors affecting denitrification rates during the growing season (Soana et al., 2019). The obtained regression Eq. (2) was used to predict the NO_3^- removal in vegetated canal sediments as a function of water NO_3^- availability, covering the common concentration range found in drainage waters affected by diffuse pollution in the Po River lowlands (0.5–11.5 mg N L^{-1}) (Pierobon et al., 2013; Soana et al., 2019). Previous studies have provided independent evidence to support the hypothesis that denitrification alone accounts for the main share of the total N removal in vegetated canal sediments, firstly the 1:1 ratio between NO_3^- consumption and N_2 production rates (Fig. 3B). Furthermore, Pierobon et al. (2013) showed that plant N uptake and sequestration in biomass represent only a small fraction of total NO_3^- consumption (<5 %), confirming that emergent macrophytes play a crucial indirect role in quantitatively supporting relevant bacterial dissipation processes.

The capacity of the canal network to remove NO_3^- by denitrification was estimated under two conditions that differed in the overall extent of the network where in-stream vegetation was maintained throughout the growing season. It was assumed that vegetation cutting would be postponed from the current mid-summer to the end of the growing season (i.e., the end of September), in 25 % (~3250 km) and 50 % (~6500 km) of the network length. This would make it possible to take advantage of the effect of in-stream vegetation in reducing NO_3^- loads during the spring-summer months (150 days). To maximise the drainage capacity of the network, it was assumed that vegetation mowing would occur before the autumn period of heavy rainfall, as per current practice (Soana et al., 2019; Tamburini et al., 2020; Soana et al., 2021).

In order to extrapolate NO_3^- removal measurements to the entire study area, a detailed map of the canal network (Fig. 1D) was created in a GIS environment (QGIS Development Team, 2023) by merging vector data downloaded from the geoportals of the local water authority (Po River District Authority, 2023) and the Italian Regions included in the study area (Emilia-Romagna Region, 2021b; Veneto Region, 2022; Lombardy Region, 2023). Water quality datasets (period 2000–2021, monthly or bimonthly surveys; Fig. S1, Fig. S2, Supplementary Material 2) were provided by the Environmental Protection Agencies of the Emilia-Romagna and Lombardy Regions for 50 sites belonging to the institutional monitoring network and located on the main canals collecting drainage water from agricultural land (Emilia-Romagna Region, 2021; Lombardy Region, 2021). The predictive Eq. (2) was applied to all official monitoring surveys conducted from May to September for which measurements of water NO_3^- concentrations were available and it resulted in explaining the 85 % of variance (Fig. 3A). A dataset of daily denitrification rates, expressed per unit of canal length (kg N km^{-1} day^{-1}), was estimated for each month during the irrigation period (Fig. S3, Supplementary Material 2), according to the concentration NO_3^- ranges found in drainage waters. Nitrate concentrations and denitrification rates were spatially analysed in the canal network using

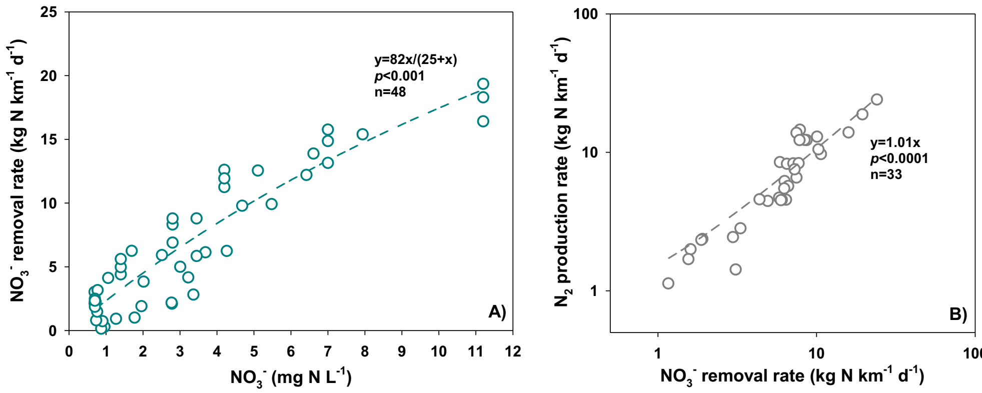

Fig. 3. A) NO_3^- removal rates as a function of water NO_3^- concentration measured in vegetated canals of the Po River lowlands during the irrigation period; B) log-log relationship between NO_3^- removal and N_2 production rates. Raw data were extracted from Pierobon et al. (2013), Castaldelli et al. (2015), Castaldelli et al. (2018) and Soana et al. (2020).

the Kriging interpolation method with a spherical semivariogram model (Papritz and Stein, 1999).

Six scenarios of in-stream vegetation maintenance were simulated, differing for 1) the percentage of canal network length (25 % and 50 %) where vegetation mowing is postponed to the end of the growing season, 2) the interventions located in canal reaches with variable water NO_3^- availability (1.2–2.4 mg N L^{-1}, 2.4–4.8 mg N L^{-1}, >4.8 mg N L^{-1}). The adopted ranges of NO_3^- concentrations correspond to the threshold values of the classes “moderate”, “poor” and “bad” according to the LIMeco index.

The following equation was used to estimate the monthly NO_3^- removal capacity of the canal network (t N month^{-1}):

$$R = \frac{V\% \bullet L \bullet D_V \bullet n}{1000} \quad (3)$$

where:

V% = scenario of vegetation maintenance (25 % and 50 % of canal network length).

L = total canal network length (km).

D_V = daily rates of denitrification in vegetated sediments calculated as a function of water NO_3^- availability (kg N km^{-1} d^{-1}).

n = number of days of each month of the irrigation period (i.e., from May to September).

For each month from May to September, the predicted N removal capacity in the canal network under the different scenarios was compared to the “distance to target”, i.e., the difference between the current NO_3^- load (baseline) and the target NO_3^- load that needs to be achieved in order to have concentrations constantly lower than 1.2 mg N L^{-1} in the draining waters conveyed to the sea.

3. Results

3.1. Nitrate loads exported from the Po River basin: baseline vs good ecological status

During the period 2000–2021, NO_3^- concentrations in the Po River water, upstream and downstream of the irrigated zone, displayed a clear seasonal pattern, with lower values recorded in summer and higher ones in winter (Fig. 4). In all months, the river water was enriched with NO_3^- as it flowed through the irrigated zone for the last 250 km of its course. Nitrate concentrations in water exported from the basin showed a marked monthly variation (Fig. 4B), with the minimum and maximum values measured in July (1.1 mg N L^{-1}) and January (3.0 mg N L^{-1}), respectively. During the irrigation months, from 50 % (in July) to 100 % (in September) of the NO_3^- data exceeded the threshold of 1.2 mg N L^{-1}, that is, the upper limit of the good status according to the LIMeco index.

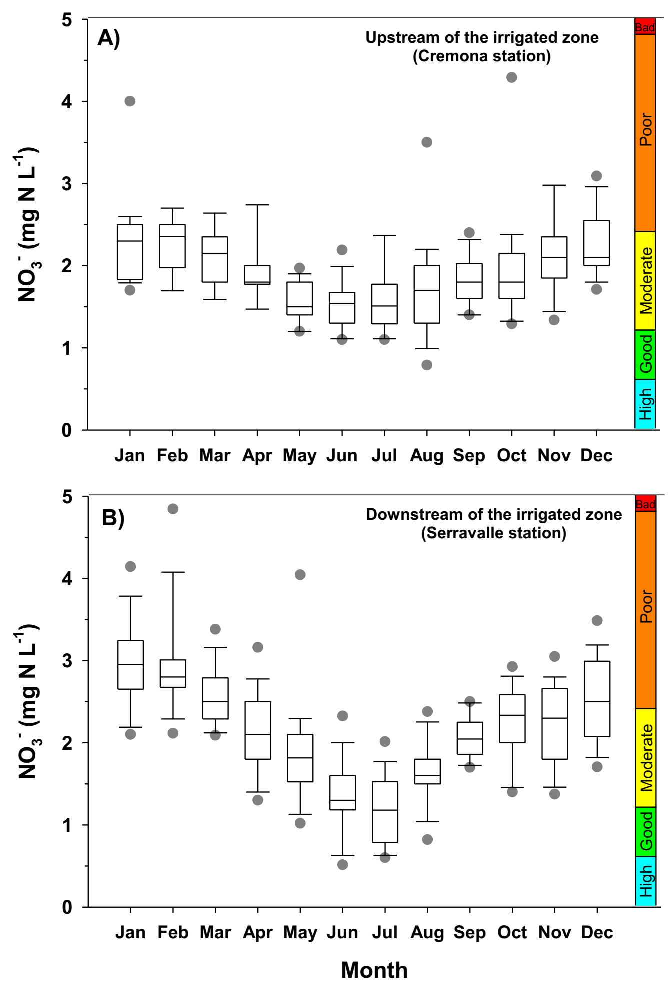

Fig. 4. Pattern of monthly NO_3^- concentrations in the waters of the Po River upstream and downstream of the irrigated zone. Threshold values for the assessment of ecological status according to the LIMeco index are shown. The central horizontal line in the box is the median, the upper and lower line of the box are the 25th and 75th percentiles, respectively, and the whiskers are the 10th and 90th percentiles, respectively. Outliers are shown as solid grey dots.

Otherwise, NO_3^- concentrations were systematically above the threshold value in all the other months (from October to April).

In the last two decades, the annual NO_3^- load exported from the Po

River basin to the Adriatic Sea showed a high inter-annual variability, ranging from ~53,000 t N yr^{-1} (in 2007 and 2017) to ~150,000 t N yr^{-1} (in 2009 and 2014), depending on the hydrological conditions (Fig. 5). In contrast, NO_3^- concentrations did not change significantly over the same period ($p > 0.05$; Fig. S4 and Fig. S5, Supplementary Material 2). According to previous studies (Viaroli et al., 2018; Gervasio et al., 2022), NO_3^- is the dominant form of N in the waters of the Po River at the basin closing section, accounting on average for >75 % of the total N load, which is a common feature of agricultural watersheds (Pina-Ochoa and Álvarez-Cobelas, 2006; Hill, 2023). Nitrate export during the irrigation period, which averaged 28 % (range 12 %–38 %) of the corresponding annual value, displayed no significant trend over the last two decades when tested by linear regression, but varied greatly from year to year ($p > 0.05$; Fig. 5). Indeed, the period 2000–2021 includes years with rather extreme hydrological conditions in the late spring/summer months, e.g., 2002 and 2014 were very wet, while the years from 2003 to 2007 were characterized by frequent and persistent summer droughts (Fig. 5). Similar to the NO_3^- concentrations, the monthly NO_3^- loads exported from the Po River basin followed a seasonal pattern (Fig. 6), with late autumn/early winter peaks (up to 12,600 t N $month^{-1}$ in November and December) and a mid-summer minimum (3000 t N $month^{-1}$ in July). The monthly target NO_3^- loads ranged from 2400 to 6800 t N $month^{-1}$, with the higher values calculated for spring and autumn, because of greater precipitation and runoff. The distance to target, that is, the amount of NO_3^- load required to meet the target threshold varied over one order of magnitude, ranging from ~630 t N $month^{-1}$ in mid-summer to ~6900 t N $month^{-1}$ in mid-winter (Fig. 6). The current NO_3^- load reaching the coastal zone systematically exceeded the target NO_3^- load in all months, ranging from +26 % (July) to +148 % (February), with an overall average of +45 % during the irrigation period (Fig. 6).

3.2. Denitrification potential of the canal network and good ecological status for NO_3^-

The denitrification rates predicted if the canals were kept vegetated were found to be spatially heterogeneous (Fig. 7), depending on NO_3^- availability in water over the irrigation period (Fig. S2, Supplementary Material 2). Postponing the cutting of in-stream vegetation to the end of the growing season in 25 % and 50 % of the canal network length, respectively, would guarantee a denitrification capacity of 2500–8000 t N yr^{-1} and 5000–17,600 t N yr^{-1} throughout the irrigation period (Fig. 8). Variable denitrification performance can be obtained by maintaining vegetation in stretches with different water NO_3^-

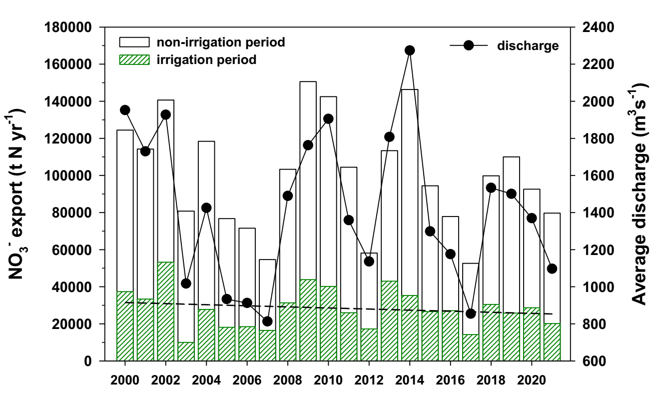

Fig. 5. Trajectory of annual NO_3^- loads exported from the Po River basin during the last two decades, divided into the contribution of irrigation (May–September) and non-irrigation period. Temporal trend of NO_3^- loads during the irrigation period resulted not significant ($p > 0.05$). Average annual discharge at the basin closing section is also reported.

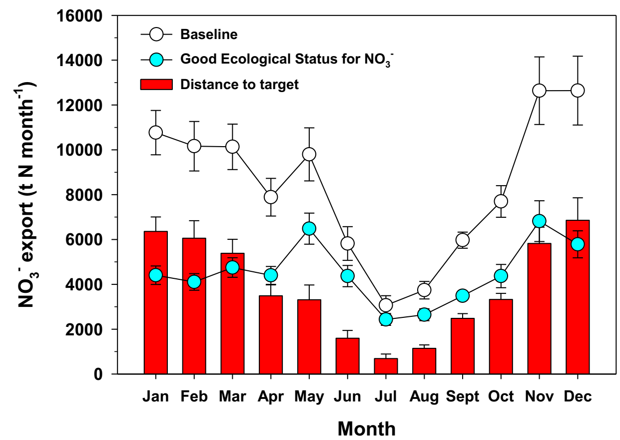

Fig. 6. Pattern of monthly NO_3^- loads (baseline and target) exported from the Po River basin (average ± standard deviation, years from 2000 to 2021). The distance to target is the difference between the current load (baseline) and the target load, i.e., the delta that needs to be reduced to achieve concentrations at the basin outlet constantly lower than the threshold for the good ecological status (1.2 mg N L^{-1}).

availability, with the highest mitigation potential achieved by postponing vegetation mowing in canals where NO_3^- exceeds 4.8 mg N L^{-1}. The predicted denitrification capacity would match the NO_3^- load reduction target required to achieve good ecological status as defined by the WFD, if mowing was postponed to the end of September in 25 % or 50 % of the canal network length, with $NO_3^- > 4.8$ mg N L^{-1} and in the range 2.4–4.8 mg N L^{-1}, respectively. Looking at the mid-summer months (July and August), when the risk of eutrophication in coastal zones is the highest, the reduction target can be achieved with even less restrictive criteria, i.e., by maintaining vegetation in a quarter of the network length by selecting canals and ditches with a wider range of NO_3^- concentrations (>2.4 mg N L^{-1}).

4. Discussion

4.1. Vegetated canals as a nature-based solution to boost the denitrification capacity of agricultural landscapes

A major challenge for landscape managers, landowners, and local stakeholders is the mitigation of excess reactive N in agricultural watercourses. Runoff and leaching of excess N can be directly prevented by targeting the source of the pollution, either by reducing the need for fertilizer or by improving its application and uptake in soils (Grizzetti et al., 2021; Vigiak et al., 2023). Although this is the preferred solution, effective strategies to meet basin water quality goals would require the use of multiple and combined N-attenuation tools located at the aquatic interfaces (Goeller et al., 2020; Ferreira et al., 2023). Edge-of-field mitigation tools in irrigated catchments should seek to intercept the hydraulic pathways of NO_3^- delivery and create conditions that enhance its attenuation along the landscape-river-sea continuum.

Irrigated and artificially drained agricultural catchments are relevant sources of diffuse N contributing to coastal eutrophication, but at the same time also sites where N removal processes, especially denitrification, occur and can potentially be enhanced by acting on the management of certain landscape features. A growing number of studies have demonstrated that Mediterranean basins subjected to intense pressure from agriculture and livestock farming exhibit lower N export to coastal areas when intensified irrigated agriculture is dominant (Lassaletta et al., 2012; Romero et al., 2016; Soana et al., 2021). The high density of irrigation channels and reservoirs leads to high N retention in catchments and hinders N transfer. Nitrogen dynamics are closely linked to water management practices, particularly irrigation

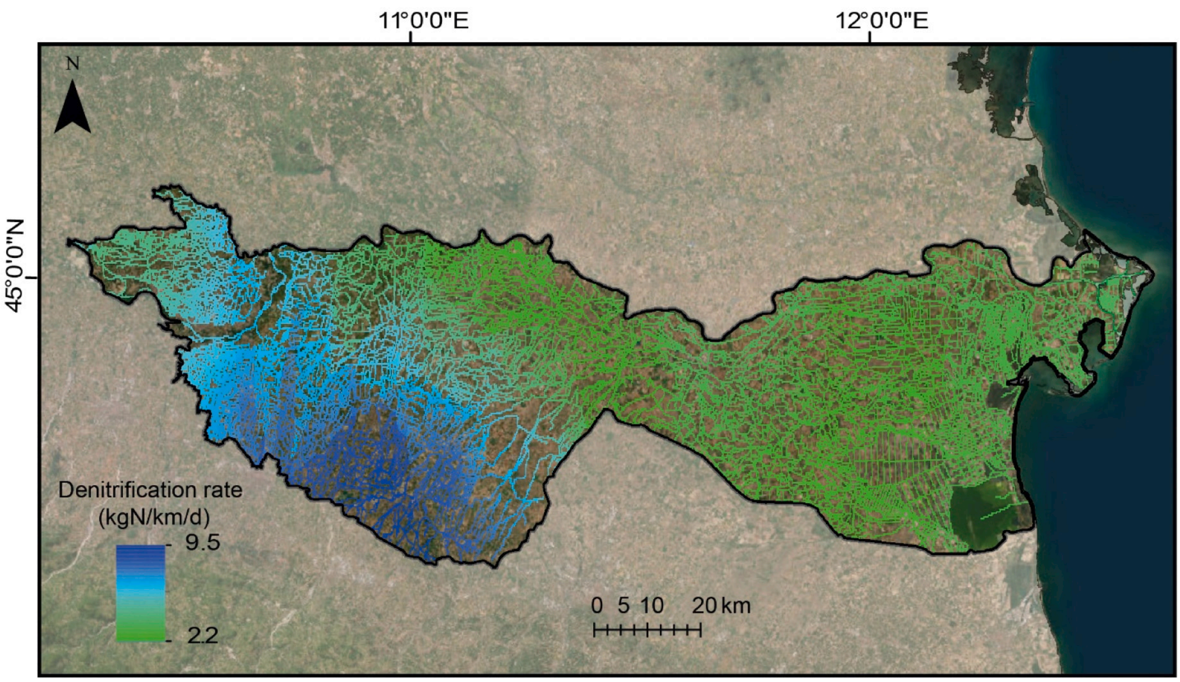

Fig. 7. Spatial distribution of denitrification rates predicted if the canals were kept vegetated (Kriging interpolation method). Average values over the irrigation period are reported.

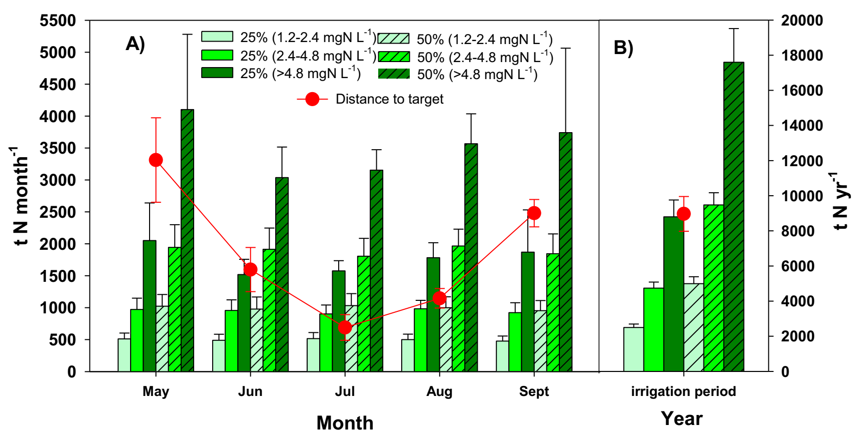

Fig. 8. Predicted NO_3^- removal via denitrification under scenarios of vegetated canals implemented in the area irrigated by River Po water. The scenarios differ in 1) the percentage of canal network length (25 % and 50 %) where vegetation mowing is postponed to the end of the growing season, 2) the NO_3^- availability in water (three classes according to LIMeco index).

practices (Serra et al., 2023), which can create favourable conditions for N processing, i.e., by increasing the retention time of water volumes withdrawn from major water bodies and distributed to agricultural land through capillary networks of canals and ditches during periods when water temperatures are more favourable for microbially-mediated N dissipation (Törnqvist et al., 2015; Barakat et al., 2016). Similar to wetlands, canals and ditches are denitrification hotspots because of the intensive exchange of labile organic matter and nutrients from the surrounding cropland, anoxic or hypoxic conditions in sediments, long water residence time, and high ratio of bioreactive surface area to water volume (e.g., Castaldelli et al., 2015; Veraart et al., 2017; Taylor et al., 2020).

Canal networks can boost denitrification in irrigated basins and prevent water quality degradation in receiving terminal water bodies if conservative aquatic vegetation management practices are implemented on a large scale. Several ecological practices fulfilling the nature-based solution criteria to address nutrient pollution control are based on the principles of phytodepuration (e.g., constructed wetlands, buffer strips, stormwater biofiltration systems) (Goeller et al., 2020; Kumwimba et al., 2023; Rizzo et al., 2023). The shallow water depth of the canals supports the role of aquatic vegetation as an "ecosystem engineer" in regulating biogeochemical reactions and affecting denitrification in several ways, both directly and indirectly. Macrophytes directly extend bioactive surfaces by providing physical attachment areas for attached biofilms, which have been identified as biogeochemical hotspots and represent the core system that maintains the depuration capacity of canal networks affected by diffuse sources of NO_3^- pollution (Soana et al., 2018; Mu et al., 2020). Indeed, 1 m^2 of canal area with dense stands of emergent macrophytes (*Phragmites australis* or *Typha latifolia*) can provide an additional surface area of 3–8 m^2 to host bacterial consortia, resulting in denitrification rates up to an order of magnitude higher than those in unvegetated sediments (Pierobon et al., 2013). Indirectly, aquatic vegetation affects denitrification rates by altering solute concentrations through uptake and release during growth and senescence, i.

e., by increasing the availability of labile organic compounds, substrates for heterotrophic denitrifying bacteria (root exudates, decaying plant litter, and suspended particles trapped as a consequence of low hydrodynamism among the vegetation stands), and by releasing oxygen from roots, which promotes the development of oxic/anoxic niches where coupled redox reactions (i.e. nitrification and denitrification) can occur (Alldred and Baines, 2016; Srivastava et al., 2017).

A side effect of NO_3^- removal via denitrification is the production of nitrous oxide (N_2O), which has a high global warming potential, approximately 300 times that of carbon dioxide (Burgin et al., 2013; Webb et al., 2021). Nitrous oxide analyses were not performed extensively for all the monitored times and sediments whose denitrification rates were used in the upscale model, but only just in a few selected samplings to assess the relevance of N_2O with respect to N_2 emission. Both at the reach-scale and in laboratory mesocosms, rates of N_2O production resulted negligible, being on average < 10 ‰ of the corresponding N_2 production (Castaldelli et al., 2015; Castaldelli et al., 2018; Soana et al., 2018), demonstrating that complete denitrification was the main process responsible for NO_3^- removal (Fig. 3B). Although studies focusing on biogeochemical N dynamics in agricultural canals have increased in the last years (e.g., Castaldelli et al., 2015; Veraart et al., 2017; Soana et al., 2020; Taylor et al., 2020), there is still a lack of quantitative understanding of the environmental drivers that favour N_2O emissions at the expense of NO_3^- conversion to N_2. The comparative assessment of ecosystem services (i.e., water depuration) and disservices (i.e., greenhouse gas emissions) in agricultural landscapes remains an open question that needs be explored in the future.

4.2. Conservative management of aquatic vegetation as a WFD implementation measure in the Po River lowlands

The simulated scenarios presented in this study demonstrated that the maintenance of vegetation in the canal network could be an effective strategy for reducing NO_3^- loads in irrigated watersheds, contributing to a partial mitigation of N export during the most sensitive period for eutrophication (Romero et al., 2013; Ricci et al., 2022), with positive effects on water quality in the terminal water bodies, such the coastal lagoons of the Po Delta and the northern Adriatic Sea. This system is one of the most important chlorophyll hotspots in the Mediterranean area and has been recognized as dependent on freshwater and nutrient inputs from the Po River (Cozzi and Giani, 2011; Viaroli et al., 2018). As in other watersheds affected by human activity, intensive agricultural practices and urbanization have greatly simplified the Po River landscape, by eliminating natural buffers (e.g., riparian vegetated zones, wetlands and riverine lakes, and temporary ponds), accelerating the nutrient transport from the arable lands to the main water bodies and exacerbating eutrophication and NO_3^- pollution of inland waters (Racchetti et al., 2011; Frascaroli et al., 2021). Opportunities for restoring the biogeochemical functionality of lost natural river environments can be found in the secondary hydrographic networks of irrigation and drainage canals and ditches that already exist in agricultural landscapes. By their very nature, canals are positioned to interact with large volumes of water moving from agricultural land to terminal water bodies, and the adoption of conservative practices of in-stream vegetation would enable them to operate as free water surface wetlands with linear development (Kumwimba et al., 2023; Wu et al., 2023), offering the potential to improve water quality while maintaining water transport capacity (Errico et al., 2019; Kalinowska et al., 2023). This could assist and complement other measures, such as the design and creation of constructed wetlands (Kumwimba et al., 2023; Rizzo et al., 2023), which is frequently unfeasible because of space limitations and budget constraints for environmental programs (Shortle et al., 2012). Vegetated canals provide the same removal conditions as constructed wetlands, but their capital cost is lower because these systems can develop naturally.

Conservative management of aquatic vegetation in canal networks can be seen as an opportunity to help solve, albeit partially, the persistent NO_3^- pollution problem in the Po River basin, bacause current policies and management options have failed to control excess N emissions from diffuse sources (Palmeri et al., 2005; Viaroli et al., 2018). Recent assessments indicate a 60–80 % probability of failing to achieve a WFD good ecological status in the waterbodies of the Po Plain (Vigiak et al., 2021). Persistently high N loads in the decades following N fertilizer reduction (Gervasio et al., 2022) are due to hydrologic legacy, i.e., NO_3^- storage in groundwater and unsaturated zone followed by its release to surficial waters (Racchetti et al., 2019; Musacchio et al., 2020). Water used for irrigation in the study area, withdrawn from the Po River, conveys a NO_3^- load, partially a result of a long-term contamination, which vegetated canals may contribute to buffering.

To promote effective strategies to mitigate N excess and to address sustainable agricultural practices, research efforts should identify both the pathways of N load generation in the watersheds and the buffering mechanisms along the land-sea continuum. This issue is challenging in a time of climate change, which is predicted to modify the timing and magnitude of nutrient export, due to extreme rainfall events or prolonged drought, thereby affecting water management and irrigation practices (Costa et al., 2022; Gervasio et al., 2022; Soana et al., 2023). At the same time, the extreme hydrological conditions associated with climate change may hinder the large-scale implementation of vegetation conservation measures. In this scenario, the hydrological aspects are given priority in the management of canal networks, with all interventions being finalised to reduce friction and optimise water transfer capacity. In fact, the resistance to flow is increased by the presence of extensive stands of macrophytes, which slows runoff and raises the water level. To prevent this, local Land Reclamation Authorities conduct routine mowing operations to control vegetation growth and maximise conveyance (Errico et al., 2019; Kalinowska et al., 2023). Therefore, the conservation management option presented here needs to be necessarily accompanied by a site-specific hydraulic risk assessment and should be prioritised for canals where sections can be widened in order to accommodate the increased hydraulic impedance due to the presence of in-stream vegetation throughout the irrigation season. The widening of canal sections can be combined with dredging operations, which are generally carried out by the consortia to remove the deposited sediments every 5 to 10 years, depending on the pedoclimatic context (Dollinger et al., 2015; Rudi et al., 2020).

In order to maintain the conveyance efficiency, canal sections must be widened and redesigned– which cannot be carried out on the entire network for economic reasons. Therefore, it is necessary to study the boundary parameters that have the greatest influence on treatment efficiency to draw a list of criteria for selecting the most suitable sections for the restoration of vegetation. It is important to prioritize areas for vegetated canal implementation by considering the spatial distribution of criteria that maximise depuration efficiency (e.g., hydraulic risk water retention time, and chemical-physical features). The results of the present study showed that delaying aquatic vegetation mowing in a quarter of the canal network length of the Po River irrigated zone would provide the denitrification capacity to meet the NO_3^- load reduction target required to achieve a WFD good ecological status in waters draining into the Adriatic Sea from May to September. This could be met by selecting canal stretches with high NO_3^- availability (i.e., >2.4 mg N L^{-1}), such as those located in the zone of the right bank of the Po River (Fig. 7; Fig. S2, Supplementary Material 2), where the N surplus in the cropland reached the highest values for the entire basin (Viaroli et al., 2018; Racchetti et al., 2019; Soana et al., 2019). The selection of canals must be made by the local Land Reclamation Authorities in the first instance, as this is primarily dictated by hydraulic safety. However, work such as this can guide the choice of priority areas by suggesting that interventions should be targeted where NO_3^- availability, and therefore denitrification potential, is highest (Fig. 7).

Recent research and a number of demonstration activities carried out in Italy have shown that in many cases it is possible to achieve a steady lowering of water levels and a consequent reduction in the risk of flooding without completely cutting the vegetation in the watercourse and along the banks, but by limiting the cutting to a more or less wide area, depending on the hydraulic and environmental conditions (Errico et al., 2019; Veneto Agricoltura, 2020). These experiences have paved the way for the experimentation of the so-called "gentle maintenance" methods of aquatic vegetation, which now need to be adapted to the different types of canals and different situations and implemented on a wider scale, finding the right balance between the objectives of hydraulic safety and those of ecosystem functioning. Ultimately, the general perspective is the definition of a new role for the local Land Reclamation Authorities as providers of multiple environmental benefits to the society.

5. Conclusion

The present results highlight that promoting denitrification in the canal networks of irrigated watersheds by implementing vegetation-preserving management practices may be an effective tool to attenuate the N loads exported to the coastal zones during the period that is most prone to eutrophication. This study provides essential information for landscape managers and policy makers across Europe who develop and implement effective measures to reduce nutrient flows through river systems to achieve the WFD water quality goals, which were set two decades ago and are still not being met. The postponement of vegetation mowing could be promoted as a nature-based solution for inclusion in restoration strategies to achieve NO_3^- reduction targets in agricultural catchments.

Despite recent advances in the quantification of denitrification in lowland canals, the variables that explain the process need to be investigated at spatial and temporal resolution, and particularly at landscape scale, to better capture the variability in rates and denitrification efficiency, including the occurrence of hot spots and hot moments in the context of climate change. Understanding the complex biogeochemical N dynamics in irrigated catchments and the links between the drivers regulating N removal (e.g., plant species, hydrology, organic carbon availability, and nutrient pulses) is essential to optimise nature-based solution design for improved water treatment performance. Detailed investigations of canal N metabolism, and in particular the parameterisation of their ability to remove excess nutrients, need to be integrated into large-scale studies (e.g., priority management areas with elevated N surplus) in order to inform effective remediation measures.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Data availability

Data will be made available on request.

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CRediT authorship contribution statement

Elisa Soana: Conceptualization, Methodology, Investigation, Data curation, Formal analysis, Writing–Original draft preparation, Visualization, Funding acquisition; Anna Gavioli: Investigation, Data curation, Visualization, Writing–Review and editing; Federica Neri: Investigation, Data curation; Giuseppe Castaldelli: Conceptualization, Writing–Review and editing, Funding acquisition, Supervision.

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.scitotenv.2023.167331.

References

Alldred, M., Baines, S.B., 2016. Effects of wetland plants on denitrification rates: a meta-analysis. Ecol. Appl. 26 (3), 676–685. https://doi.org/10.1890/14-1525.

APAT—IRSA/CNR. 2003. Analytical methods for water. Manuals and guidelines, 29/2003 (in Italian) https://www.isprambiente.gov.it/it/pubblicazioni/manuali-e-linee-guida/metodi-analitici-per-le-acque.

ARPAE-SIMC, 2021. Hydrological Annals. Dexter Portal. https://simc.arpae.it/dext3r/ (Accessed 17 April 2023).

Barakat, M., Cheviron, B., Angulo-Jaramillo, R., 2016. Influence of the irrigation technique and strategies on the nitrogen cycle and budget: a review. Agric. Water Manag. 178, 225–238. https://doi.org/10.1016/j.agwat.2016.09.027.

Basu, N.B., Van Meter, K.J., Byrnes, D.K., Van Cappellen, P., Brouwer, R., Jacobsen, B.H., Olsen, S.B., 2022. Managing nitrogen legacies to accelerate water quality improvement. Nat. Geosci. 15 (2), 97–105. https://doi.org/10.1038/s41561-021-00889-9.

Battye, W., Aneja, V.P., Schlesinger, W.H., 2017. Is nitrogen the next carbon? Earth's Future 5 (9), 894–904. https://doi.org/10.1002/2017EF000592.

Beaulieu, J.J., Golden, H.E., Knightes, C.D., Mayer, P.M., Kaushal, S.S., Pennino, M.J., Hill, B.H., 2015. Urban stream burial increases watershed-scale nitrate export. PLoS One 10 (7), e0132256. https://doi.org/10.1371/journal.pone.0132256.

Beck, H.E., Zimmermann, N.E., McVicar, T.R., Vergopolan, N., Berg, A., Wood, E.F., 2018. Present and future Köppen-Geiger climate classification maps at 1-km resolution. Scientific data 5 (1), 1–12. https://doi.org/10.1038/sdata.2018.214.

Beltran-Peña, A., Rosa, L., D'Odorico, P., 2020. Global food self-sufficiency in the 21st century under sustainable intensification of agriculture. Environ. Res. Lett. 15 (9), 095004 https://doi.org/10.1088/1748-9326/ab9388.

Bouraoui, F., Grizzetti, B., 2011. Long term change of nutrient concentrations of rivers discharging in European seas. Sci. Total Environ. 409 (23), 4899–4916. https://doi.org/10.1016/j.scitotenv.2011.08.015.

Burgin, A.J., Lazar, J.G., Groffman, P.M., Gold, A.J., Kellogg, D.Q., 2013. Balancing nitrogen retention ecosystem services and greenhouse gas disservices at the landscape scale. Ecol. Eng. 56, 26–35. https://doi.org/10.1016/j.ecoleng.2012.05.003.

Camargo, J.A., Alonso, Á., 2006. Ecological and toxicological effects of inorganic nitrogen pollution in aquatic ecosystems: a global assessment. Environ. Int. 32 (6), 831–849. https://doi.org/10.1016/j.envint.2006.05.002.

Castaldelli, G., Soana, E., Racchetti, E., Vincenzi, F., Fano, E.A., Bartoli, M., 2015. Vegetated canals mitigate nitrogen surplus in agricultural watersheds. Agric. Ecosyst. Environ. 212, 253–262. https://doi.org/10.1016/j.agee.2015.07.009.

Castaldelli, G., Aschonitis, V., Vincenzi, F., Fano, E.A., Soana, E., 2018. The effect of water velocity on nitrate removal in vegetated waterways. J. Environ. Manag. 215, 230–238. https://doi.org/10.1016/j.jenvman.2018.03.071.

Costa, D., Sutter, C., Shepherd, A., Jarvie, H., Wilson, H., Elliott, J., Liu, J., Macrae, M., 2022. Impact of climate change on catchment nutrient dynamics: insights from around the world. Environ. Rev. 31 (1), 4–25. https://doi.org/10.1139/er-2021-0109.

Cozzi, S., Giani, M., 2011. River water and nutrient discharges in the Northern Adriatic Sea: current importance and long-term changes. Cont. Shelf Res. 31 (18), 1881–1893. https://doi.org/10.1016/j.csr.2011.08.010.

Cuenca-Cambronero, M., Blicharska, M., Perrin, J.A., Davidson, T.A., Oertli, B., Lago, M., Brucet, S., 2023. Challenges and opportunities in the use of ponds and pondscapes as nature-based solutions. Hydrobiologia 1–15. https://doi.org/10.1007/s10750-023-05149-y.

Dollinger, J., Dagès, C., Bailly, J.S., Lagacherie, P., Voltz, M., 2015. Managing ditches for agroecological engineering of landscape. A review. Agron. Sustain. Dev. 35, 999–1020. https://doi.org/10.1007/s13593-015-0301-6.

Domeneghetti, A., Castellarin, A., Brath, A., 2012. Assessing rating-curve uncertainty and its effects on hydraulic model calibration. Hydrol. Earth Syst. Sci. 16 (4), 1191–1202. https://doi.org/10.5194/hess-16-1191-2012.

Emilia-Romagna Region, 2021a. Open data portal of the Emilia-Romagna Region. https://dati.arpae.it/group/acqua.

Emilia-Romagna Region, 2021b. Regional Canal network. https://geoportale.regione.emilia-romagna.it/catalogo/dati-cartografici/acque-interne.

Erba, S., Buffagni, A., Cazzola, M., Balestrini, R., 2022. Italian reference rivers under the Water Framework Directive umbrella: do natural factors actually depict the observed nutrient conditions? Environ. Sci. Eur. 34 (1), 63. https://doi.org/10.1186/s12302-022-00642-y.

Errico, A., Lama, G.F.C., Francalanci, S., Chirico, G.B., Solari, L., Preti, F., 2019. Flow dynamics and turbulence patterns in a drainage channel colonized by common reed (*Phragmites australis*) under different scenarios of vegetation management. Ecol. Eng. 133, 39–52. https://doi.org/10.1016/j.ecoleng.2019.04.016.

European Commission, 2020. Farm to fork strategy. For a fair, healthy and environmentally friendly food system. European Union, 22 p. https://food.ec.europa.eu/system/files/2020-05/f2f_action-plan_2020_strategy-info_en.pdf.

European Commission, 2021. EU Biodiversity Strategy for 2030: Bringing nature back into our lives, Directorate-General for Environment, Publications Office of the European Union. 34 p. https://op.europa.eu/en/publication-detail/-/publication/31e4609f-b91e-11eb-8aca-01aa75ed71a1/language-en.

European Commission, 2022. Nitrate Vulnerable Zones. ArcGIS Web Application. https://water.jrc.ec.europa.eu/portal/apps/webappviewer/index.html?id=d651ecd9f5774080aad738958906b51b (Accessed 20 March 2023).

European Waters. Assessment of Status and Pressures, 2018. EEA Report 7/2018 - Publications Office of the European Union, Luxembourg (2018). https://www.eea.europa.eu/publications/state-of-water.

Ferreira, C.S., Kašanin-Grubin, M., Solomun, M.K., Sushkova, S., Minkina, T., Zhao, W., Kalantari, Z., 2023. Wetlands as nature-based solutions for water management in different environments. Curr. Opin. Environ. Sci. Health 100476. https://doi.org/10.1016/j.coesh.2023.100476.

Frascaroli, F., Parrinello, G., Root-Bernstein, M., 2021. Linking contemporary river restoration to economics, technology, politics, and society: perspectives from a historical case study of the Po River Basin, Italy. Ambio 50 (2), 492–504. https://doi.org/10.1007/s13280-020-01363-3.

Galloway, J.N., Bleeker, A., Erisman, J.W., 2021. The human creation and use of reactive nitrogen: a global and regional perspective. Annu. Rev. Environ. Resour. 46, 255–288. https://doi.org/10.1146/annurev-environ-012420-045120.

Garnier, J., Anglade, J., Benoit, M., Billen, G., Puech, T., Ramarson, A., Tallec, G., 2016. Reconnecting crop and cattle farming to reduce nitrogen losses to river water of an intensive agricultural catchment (Seine basin, France): past, present and future. Environ. Sci. Pol. 63, 76–90. https://doi.org/10.1016/j.envsci.2016.04.019.

Gervasio, M.P., Soana, E., Granata, T., Colombo, D., Castaldelli, G., 2022. An unexpected negative feedback between climate change and eutrophication: higher temperatures increase denitrification and buffer nitrogen loads in the Po River (Northern Italy). Environ. Res. Lett. 17 (8), 084031 https://doi.org/10.1088/1748-9326/ac8497.

Glibert, P.M., 2020. Harmful algae at the complex nexus of eutrophication and climate change. Harmful Algae 91, 101583. https://doi.org/10.1016/j.hal.2019.03.001.

Goeller, B.C., Febria, C.M., McKergow, L.A., Harding, J.S., Matheson, F.E., Tanner, C.C., McIntosh, A.R., 2020. Combining tools from edge-of-field to in-stream to attenuate reactive nitrogen along small agricultural waterways. Water 12 (2), 383. https://doi.org/10.3390/w12020383.

Grizzetti, B., Bouraoui, F., Billen, G., Van Grinsven, H., Cardoso, A., Thieu, V., Garnier, J., Curtis, C., Howarth, R., Johnes, P., 2011. Nitrogen as a threat to European water quality. In: Sutton, Mark A., Howard, Clare M., Erisman, Jan Willem, Billen, Gilles, Bleeker, Albert, Grennfelt, Peringe, van Grinsven, Hans, Grizzetti, Bruna (Eds.), European Nitrogen Assessment. Cambridge University Press, pp. 379–404. http://www.nine-esf.org/node/204/ENA.html.

Grizzetti, B., Vigiak, O., Udias, A., Aloe, A., Zanni, M., Bouraoui, F., Bielza, M., 2021. How EU policies could reduce nutrient pollution in European inland and coastal waters. Glob. Environ. Chang. 69, 102281. https://doi.org/10.1016/j.gloenvcha.2021.102281.

Hill, A.R., 2023. Patterns of nitrate retention in agriculturally influenced streams and rivers. Biogeochemistry 1–29. https://doi.org/10.1007/s10533-023-01027-w.

Kalinowska, M.B., Västilä, K., Nones, M., Kiczko, A., Karamuz, E., Brandyk, A., Krukowski, M., 2023. Influence of vegetation maintenance on flow and mixing: case study comparing fully cut with high-coverage conditions. Hydrol. Earth Syst. Sci. 27 (4), 953–968. https://doi.org/10.5194/hess-27-953-2023.

Kumwimba, M.N., Huang, J., Dzakpasu, M., Soana, E., Zhu, B., Wang, T., Ajibade, F.O., 2023. Advances in ecotechnological methods for diffuse nutrient pollution control: wicked issues in agricultural and urban watersheds. Front. Environ. Sci. 11, 1199923 https://doi.org/10.3389/fenvs.2023.1199923.

Lassaletta, L., Romero, E., Billen, G., Garnier, J., García-Gómez, H., Rovira, J.V., 2012. Spatialized N budgets in a large agricultural Mediterranean watershed: high loading and low transfer. Biogeosciences 9 (1), 57–70. https://doi.org/10.5194/bg-9-57-2012.

Le Moal, M., Gascuel-Odoux, C., Ménesguen, A., Souchon, Y., Étrillard, C., Levain, A., Pinay, G., 2019. Eutrophication: a new wine in an old bottle? Sci. Total Environ. 651, 1–11. https://doi.org/10.1016/j.scitotenv.2018.09.139.

Leip, A., Billen, G., Garnier, J., Grizzetti, B., Lassaletta, L., Reis, S., Westhoek, H., 2015. Impacts of European livestock production: nitrogen, sulphur, phosphorus and greenhouse gas emissions, land-use, water eutrophication and biodiversity. Environ. Res. Lett. 10 (11), 115004. https://doi.org/10.1088/1748-9326/10/11/115004.

Lombardy Region, 2021. Open Data Portal of the Lombardy Region. https://www.dati.lombardia.it/ (Accessed 17 April 2023).

Lombardy Region, 2023. Regional Hydrographic Network. https://www.dati.lombardia.it/Territorio/RETICOLO-IDROGRAFICO-REGIONALE-UNIFICATO/2398-zrim (Accessed 17 April 2023).

Malagó, A., Bouraoui, F., Pastori, M., Gelati, E., 2019. Modelling nitrate reduction strategies from diffuse sources in the Po River basin. Water 11 (5), 1030. https://doi.org/10.3390/w11051030.

Mancuso, G., Bencresciuto, G.F., Lavrnić, S., Toscano, A., 2021. Diffuse water pollution from agriculture: a review of nature-based solutions for nitrogen removal and recovery. Water 13 (14), 1893. https://doi.org/10.3390/w13141893.

Marchetti, M., 2002. Environmental changes in the central Po Plain (northern Italy) due to fluvial modifications and anthropogenic activities. Geomorphology 44 (3–4), 361–373. https://doi.org/10.1016/S0169-555X(01)00183-0.

McGlathery, K.J., Sundbäck, K., Anderson, I.C., 2007. Eutrophication in shallow coastal bays and lagoons: the role of plants in the coastal filter. Mar. Ecol. Prog. Ser. 348, 1–18. https://doi.org/10.3354/meps07132.

Moatar, F., Meybeck, M., 2005. Compared performances of different algorithms for estimating annual nutrient loads discharged by the eutrophic River Loire. Hydrol. Process. 19 (2), 429–444. https://doi.org/10.1002/hyp.5541.

Moore, M.T., Locke, M.A., 2020. Experimental evidence for using vegetated ditches for mitigation of complex contaminant mixtures in agricultural runoff. Water Air Soil Pollut. 231 (4), 140. https://doi.org/10.1007/s11270-020-04489-y.

Mu, X., Lv, X., Liu, W., Qiu, C., Ma, Y., Zhang, S., Jeppesen, E., 2020. Biofilms attached to Myriophyllum spicatum play a dominant role in nitrogen removal in constructed wetland mesocosms with submersed macrophytes: evidence from ^{15}N tracking, nitrogen budgets and metagenomics analyses. Environ. Pollut. 266, 115203. https://doi.org/10.1016/j.envpol.2020.115203.

Musacchio, A., Re, V., Mas-Pla, J., Sacchi, E., 2020. EU Nitrates Directive, from theory to practice: environmental effectiveness and influence of regional governance on its performance. Ambio 49, 504–516. https://doi.org/10.1007/s13280-019-01197-8.

National Institute of Statistics, 2022a. Annals of agrarian statistics. http://dati.istat.it/ (Accessed 20 March 2023).

National Institute of Statistics, 2022b. Population statistics. http://dati.istat.it/ (Accessed 20 March 2023).

Nava, V., Patelli, M., Rotiroti, M., Leoni, B., 2019. An R package for estimating river compound load using different methods. Environ. Model Softw. 117, 100–108. https://doi.org/10.1016/j.envsoft.2019.03.012.

Nieder, R., Benbi, D.K., 2022. Reactive nitrogen compounds and their influence on human health: an overview. Rev. Environ. Health 37 (2), 229–246. https://doi.org/10.1515/reveh-2021-0021.

Palmeri, L., Bendoricchio, G., Artioli, Y., 2005. Modelling nutrient emissions from river systems and loads to the coastal zone: Po River case study, Italy. Ecol. Model. 184 (1), 37–53. https://doi.org/10.1016/j.ecolmodel.2004.11.007.

Papritz, A., Stein, A., 1999. Spatial prediction by linear kriging. In: Spatial Statistics for Remote Sensing. Springer Netherlands, Dordrecht, pp. 83–113.

Petersen, R.J., Blicher-Mathiesen, G., Rolighed, J., Andersen, H.E., Kronvang, B., 2021. Three decades of regulation of agricultural nitrogen losses: experiences from the Danish Agricultural Monitoring Program. Sci. Total Environ. 787, 147619. https://doi.org/10.1016/j.scitotenv.2021.147619.

Pierobon, E., Castaldelli, G., Mantovani, S., Vincenzi, F., Fano, E.A., 2013. Nitrogen removal in vegetated and unvegetated drainage ditches impacted by diffuse and point sources of pollution. CLEAN Soil, Air, Water 41 (1), 24–31. https://doi.org/10.1002/clen.201100106.

Pina-Ochoa, E., Álvarez-Cobelas, M., 2006. Denitrification in aquatic environments: a cross-system analysis. Biogeochemistry 81, 111–130. https://doi.org/10.1007/s10533-006-9033-7.

Po River District Authority, 2021. Management Plan of the hydrographic district of the River Po. In Italian. https://pianoacque.adbpo.it/piano-di-gestione-2021/.

Po River District Authority, 2023. Surface water bodies. https://webgis.adbpo.it/catalogue/#/map/1207 (Accessed 17 April 2023).

Poikane, S., Kelly, M.G., Herrero, F.S., Pitt, J.A., Jarvie, H.P., Claussen, U., Phillips, G., 2019. Nutrient criteria for surface waters under the European Water Framework Directive: current state-of-the-art, challenges and future outlook. Sci. Total Environ. 695, 133888. https://doi.org/10.1016/j.scitotenv.2019.133888.

QGIS Development Team, 2023. QGIS Geographic Information System, Long Term Release Version 3.28. Open-Source Geospatial Foundation Project. http://qgis.osgeo.org.

Racchetti, E., Bartoli, M., Soana, E., Longhi, D., Christian, R.R., Pinardi, M., Viaroli, P., 2011. Influence of hydrological connectivity of riverine wetlands on nitrogen removal via denitrification. Biogeochemistry 103, 335–354. https://doi.org/10.1007/s10533-010-9477-7.

Racchetti, E., Salmaso, F., Pinardi, M., Quadroni, S., Soana, E., Sacchi, E., Bartoli, M., 2019. Is flood irrigation a potential driver of river-groundwater interactions and diffuse nitrate pollution in agricultural watersheds? Water 11 (11), 2304. https://doi.org/10.3390/w11112304.

Ribaudo, C., Benelli, S., Bolpagni, R., Darul, R., Bartoli, M., 2023. Macrophyte growth forms and hydrological connectivity affect greenhouse gas concentration in small eutrophic wetlands. Aquat. Bot. 188, 103660. https://doi.org/10.1016/j.aquabot.2023.103660.

Ricci, F., Capellacci, S., Campanelli, A., Grilli, F., Marini, M., Penna, A., 2022. Long-term dynamics of annual and seasonal physical and biogeochemical properties: role of minor river discharges in the North-western Adriatic coast. Estuar. Coast. Shelf Sci. 272, 107902. https://doi.org/10.1016/j.ecss.2022.107902.

Rizzo, A., Sarti, C., Nardini, A., Conte, G., Masi, F., Pistocchi, A., 2023. Nature-based solutions for nutrient pollution control in European agricultural regions: a literature review. Ecol. Eng. 186, 106772. https://doi.org/10.1016/j.ecoleng.2022.106772.

Romero, E., Garnier, J., Lassaletta, L., Billen, G., Le Gendre, R., Riou, P., Cugier, P., 2013. Large-scale patterns of river inputs in southwestern Europe: seasonal and interannual variations and potential eutrophication effects at the coastal zone. Biogeochemistry 113, 481–505. https://doi.org/10.1007/s10533-012-9778-0.

Romero, E., Garnier, J., Billen, G., Peters, F., Lassaletta, L., 2016. Water management practices exacerbate nitrogen retention in Mediterranean catchments. Sci. Total Environ. 573, 420–432. https://doi.org/10.1016/j.scitotenv.2016.08.007.

Rudi, G., Bailly, J.S., Belaud, G., Dagès, C., Lagacherie, P., Vinatier, F., 2020. Multifunctionality of agricultural channel vegetation: a review based on community functional parameters and properties to support ecosystem function modeling. Ecohydrol. Hydrobiol. 20 (3), 397–412. https://doi.org/10.1016/j.ecohyd.2020.03.004.

Serra, J., Marques-dos-Santos, C., Marinheiro, J., Aguilera, E., Lassaletta, L., Sanz-Cobeña, A., do Rosário Cameira, M., 2023. Nitrogen inputs by irrigation is a missing link in the agricultural nitrogen cycle and related policies in Europe. Sci. Total Environ. 889, 164249. https://doi.org/10.1016/j.scitotenv.2023.164249.

Shen, W., Li, S., Mi, M., Zhuang, Y., Zhang, L., 2021. What makes ditches and ponds more efficient in nitrogen control? Agric. Ecosyst. Environ. 314, 107409. https://doi.org/10.1016/j.agee.2021.107409.

Shortle, J.S., Ribaudo, M., Horan, R.D., Blandford, D., 2012. Reforming agricultural nonpoint pollution policy in an increasingly budget-constrained environment. Environ. Sci. Technol. 46 (3), 1316–1325. https://doi.org/10.1021/es2020499.

Soana, E., Gavioli, A., Tamburini, E., Fano, E.A., Castaldelli, G., 2018. To mow or not to mow: reed biofilms as denitrification hotspots in drainage canals. Ecol. Eng. 113, 1–10. https://doi.org/10.1016/j.ecoleng.2017.12.029.

Soana, E., Bartoli, M., Milardi, M., Fano, E.A., Castaldelli, G., 2019. An ounce of prevention is worth a pound of cure: managing macrophytes for nitrate mitigation in irrigated agricultural watersheds. Sci. Total Environ. 647, 301–312. https://doi.org/10.1016/j.scitotenv.2018.07.385.

Soana, E., Gavioli, A., Vincenzi, F., Fano, E.A., Castaldelli, G., 2020. Nitrate availability affects denitrification in Phragmites australis sediments. J. Environ. Qual. 49 (1), 194–209. https://doi.org/10.1002/jeq2.20000.

Soana, E., Fano, E.A., Castaldelli, G., 2021. The achievement of Water Framework Directive goals through the restoration of vegetation in agricultural canals. J. Environ. Manag. 294, 113016. https://doi.org/10.1016/j.jenvman.2021.113016.

Soana, E., Gervasio, M.P., Granata, T., Colombo, D., Castaldelli, G., 2023. Climate change impacts on eutrophication in the Po River (Italy): temperature-mediated reduction in nitrogen export but no effect on phosphorus. J. Environ. Sci. https://doi.org/10.1016/j.jes.2023.07.008 (In press).

Srivastava, J.K., Chandra, H., Kalra, S.J., Mishra, P., Khan, H., Yadav, P., 2017. Plant–microbe interaction in aquatic system and their role in the management of water quality: a review. Appl Water Sci 7, 1079–1090. https://doi.org/10.1007/s13201-016-0415-2.

Stoate, C., Baldi, A., Beja, P., Boatman, N.D., Herzon, I., Van Doorn, A., Ramwell, C., 2009. Ecological impacts of early 21st century agricultural change in Europe–a review. J. Environ. Manag. 91 (1), 22–46. https://doi.org/10.1016/j.jenvman.2009.07.005.

Tamburini, E., Soana, E., Monti, M., Fano, E.A., Castaldelli, G., 2020. Introducing Life Cycle Assessment in costs and benefits analysis of vegetation management in drainage canals of lowland agricultural landscapes. Water 12 (8), 2236. https://doi.org/10.3390/w12082236.

Taylor, J., Moore, M., Speir, S.L., Testa III, S., 2020. Vegetated ditch habitats provide net nitrogen sink and phosphorus storage capacity in agricultural drainage networks despite senescent plant leaching. Water 12 (3), 875. https://doi.org/10.3390/w12030875.

Törnqvist, R., Jarsjö, J., Thorslund, J., Rao, P.S.C., Basu, N.B., Destouni, G., 2015. Mechanisms of basin-scale nitrogen load reductions under intensified irrigated agriculture. PLoS One 10 (3), e0120015. https://doi.org/10.1371/journal.pone.0120015.

Velthof, G.L., Lesschen, J.P., Webb, J., Pietrzak, S., Miatkowski, Z., Pinto, M., Oenema, O., 2014. The impact of the Nitrates Directive on nitrogen emissions from agriculture in the EU-27 during 2000–2008. Sci. Total Environ. 468, 1225–1233. https://doi.org/10.1016/j.scitotenv.2013.04.058.

Veneto Agricoltura, 2020. Manuale per la gestione ambientale dei corsi d'acqua. Seconda edizione. L'esperienza dei Consorzi di Bonifica, ISBN 9788863372595. In Italian. https://www.venetoagricoltura.org/2021/06/editoria/manuale-per-la-gestione-ambientale-dei-corsi-dacqua-seconda-edizione-lesperienza-dei-consorzi-di-bonifica/.

Veneto Region, 2022. Regional Hydrographic Network. https://idt2.regione.veneto.it/idt/webgis/viewer?previewLayerId=13556 (Accessed 17 April 2023).

Veraart, A.J., Dimitrov, M.R., Schrier-Uijl, A.P., Smidt, H., de Klein, J.J., 2017. Abundance, activity and community structure of denitrifiers in drainage ditches in relation to sediment characteristics, vegetation and land-use. Ecosystems 20, 928–943. https://doi.org/10.1007/s10021-016-0083-y.

Vezzoli, R., Mercogliano, P., Pecora, S., Zollo, A.L., Cacciamani, C., 2015. Hydrological simulation of Po River (North Italy) discharge under climate change scenarios using the RCM COSMO-CLM. Sci. Total Environ. 521, 346–358. https://doi.org/10.1016/j.scitotenv.2015.03.096.

Viaroli, P., Soana, E., Pecora, S., Laini, A., Naldi, M., Fano, E.A., Nizzoli, D., 2018. Space and time variations of watershed N and P budgets and their relationships with reactive N and P loadings in a heavily impacted river basin (Po river, Northern Italy). Sci. Total Environ. 639, 1574–1587. https://doi.org/10.1016/j.scitotenv.2018.05.233.

Vigiak, O., Udias, A., Pistocchi, A., Zanni, M., Aloe, A., Grizzetti, B., 2021. Probability maps of anthropogenic impacts affecting ecological status in European rivers. Ecol. Indic. 126, 107684. https://doi.org/10.1016/j.ecolind.2021.107684.

Vigiak, O., Udías, A., Grizzetti, B., Zanni, M., Aloe, A., Weiss, F., Pistocchi, A., 2023. Recent regional changes in nutrient fluxes of European surface waters. Sci. Total Environ. 858, 160063. https://doi.org/10.1016/j.scitotenv.2022.160063.

Webb, J.R., Clough, T.J., Quayle, W.C., 2021. A review of indirect N_2O emission factors from artificial agricultural waters. Environ. Res. Lett. 16 (4), 043005 https://doi.org/10.1088/1748-9326/abed00.

de Wit, M., Bendoricchio, G., 2001. Nutrient fluxes in the Po basin. Sci. Total Environ. 273 (1–3), 147–161. https://doi.org/10.1016/S0048-9697(00)00851-2.

Wu, H., Wang, R., Yan, P., Wu, S., Chen, Z., Zhao, Y., Zhang, J., 2023. Constructed wetlands for pollution control. Nat. Rev. Earth Environ. 4 (4), 218–234. https://doi.org/10.1038/s43017-023-00395-z.

Zhou, X., Jomaa, S., Yang, X., Merz, R., Wang, Y., Rode, M., 2023. Stream restoration can reduce nitrate levels in agricultural landscapes. Sci. Total Environ. 164911 https://doi.org/10.1016/j.scitotenv.2023.164911.