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Limited N removal by denitrification in agricultural drainage ditches in the Taihu Lake region of China

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Abstract
Purpose Agricultural drainage ditches constitute corridor wetlands that facilitate agricultural non-point nitrogen (N) load transportation into river systems. Quantifying sediment denitrification in ditches provides relationships between N losses from agriculture and water quality. However, high denitrification rate potential and limited N residence time make the total denitrification removal capacity in ditches uncertain. The purpose of this work was to identify N removal by denitrification in agricultural ditches in the Taihu Lake region of China.
Materials and methods A field investigation and laboratory analyses were conducted to investigate the sediment denitrification rate of ditches in areas under different crops, including vegetable, rice-wheat fields, and a peach orchard, between June 2014 and October 2015. At each sampling, concentrations of dissolved inorganic nitrogen (DIN, NO_3^--N, NH_4^+-N), water temperature, pH, Eh, dissolved oxygen, and dissolved organic carbon of the overlying water and the DIN and total carbon of the sediment were examined.
Results and discussion Sediment denitrification rates in all the ditches exhibited high spatial and temporal heterogeneity. Concentrations of DIN and temperature of the overlying water were key factors controlling denitrification in the ditch system. The sediment denitrification rate in the ditches could be estimated by a nonlinear mixed-effect model. Based on estimating data on N concentrations and temperature of overlying water and our established nonlinear mixed model, less than 1% of N was removed by denitrification annually in the ditches. The limited N removal by sediment denitrification was attributed to short retention times and the low area of the ditch system.
Conclusions Dissolved inorganic nitrogen concentrations and temperature of the overlying water were key factors controlling denitrification in the ditch system. High density of ditches did not lead to high N removal by denitrification due to short retention times and low areal coverage in the Taihu Lake region.

Keywords Denitrification · Non-point pollution · Water quality · Retention time

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1 Introduction

Extensive agricultural development and nitrogen (N) fertilizer use over the past 50 years have led to substantial quantities of N in water bodies worldwide. Due to the heavy N pollution, water deterioration and eutrophication has been observed in > 80% of the lakes and rivers in China (Novotny et al. 2010). This is especially true of the Taihu Lake region of southeastern China where farming is intensive and the practice has been to use excessive amounts of N-fertilizer such that the land has received as much as 550–600 kg N ha^{-1} $year^{-1}$ (Xing and Zhu 2002). However, since the farmland N-fertilizer utilization rate is only 20–50%, high large amounts of N have entered the surface water via runoff and farmland drainage, and the resulting water pollution presents an increasingly serious environmental problem (Ni et al. 2007; Ju et al. 2009). This has been evident when occurrences of eutrophication have resulted in seasonal incidents of algal blooms (*Microcystis* spp.) in Taihu Lake and the Yangtze River estuary (Zhong et al. 1999; Qin et al. 2007). Therefore, it is necessary to quantify the impact of the agricultural N cycle on surface water quality in order to determine ways by which the pollution can be mitigated.

In order to reduce river water pollution, it is necessary to first identify the contributions of various N pollution sources. However, there is controversy over the main pollution sources for the N loading of the river network N that drains into Taihu Lake. For example, Xing and Zhu (2001) suggested that anthropogenic reactive N (including industrial, urban, and rural wastes) was the main source of pollution based on ^{15}N isotope tracing. However, Gao et al. (2004) estimated that agricultural sources of N could explain 75% of the variance in the river N concentration variability based on a regression model.

Ignoring the N removal in ditches is probably the main reason causing the controversy. Currently, the estimation of agricultural non-point sources load is mainly dependent on field measurements of N drainage losses and watershed characteristics such as those of its hydrology, e.g., rainfall and runoff, that can then be used to develop empirical models that can estimate the N load. Simpler empirical models are often preferred because they require fewer parameters as input data (Liu et al. 2015). For example, the export coefficient model uses only land use, livestock numbers, and export coefficient to predict N load (Johnes and Heathwaite 1997). The soil and water assessment tool (SWAT), a mechanistic model used for modeling non-point source pollution, has hundreds of parameters that can lead to unexpected uncertainties in the model outputs (Kim et al. 2017). An alternative approach is to use mass balances to determine N sources (Jain et al. 1998). However, most of these approaches ignore the N removal during the water flow process through the drainage ditch network. Within irrigated agricultural watersheds, denitrification in the canal varied with rates at the reach-scale (5–25 mmol N m^{-2} d^{-1}) up to one order of magnitude higher than in sediment alone (3–7 mmol N m^{-2} d^{-1}) (Castaldelli et al. 2015). Therefore, without considering the drainage ditches, it is neither possible to quantify accurately the river network N loading from agricultural runoff nor to estimate accurately the contribution of agricultural runoff N to water pollution.

The drainage ditch system is a type of wetland within the agricultural ecosystem, and it is the first waterway system for agricultural non-point N flowing from the source towards the river system. It has been estimated that more than 65% of the agricultural non-point N enters the ditch system (Alexander et al. 2007). Although the individual ditches have relatively small dimensions (width 0.1–15.0 m; depth 0.1–5.0 m), collectively, the ditches are distributed extensively for irrigation and drainage purposes in the agricultural catchment. The total length of the ditches can account for 85% of the total length of the waterways in an agricultural catchment (Peterson et al. 2001). Extensively distributed ditch networks may play a crucial role as N sinks before the agricultural loading N enters into the river system.

Denitrification has been recognized as the dominant pathway for N removal in wetland ecosystems (Kreiling et al. 2011). Sediment denitrification is mediated by facultative bacteria that respire by using NO_3^- as the terminal electron acceptor and organic substrates as the electron donor when oxygen (O_2) is depleted (Seitzinger et al. 1993). Denitrifying systems consist of four steps: $NO_3^- \rightarrow NO_2^- \rightarrow NO \rightarrow N_2O \rightarrow N_2$ (Kreiling et al. 2011). Therefore, the denitrification process can have positive effects on the environment by reducing NO_3^- concentrations and, thereby, the risks of non-point source pollution (Sala et al. 2000; Stevens et al. 2004). Many studies have indicated that sediment denitrification varies spatially and temporally, and that it is mainly regulated by various environmental factors including overlying water NO_3^--N concentrations, water temperature (WT), dissolved oxygen (DO), and dissolved organic carbon (DOC) concentrations. (Li et al. 2013; Zhao et al. 2015). It is important to identify the denitrification controlling factors and the magnitude of their impacts in order to determine the relative roles of the drainage ditch wetland ecosystem in reducing reactive N pollutants.

The agricultural land systems in the Taihu Lake region form land use mosaic patterns in the watershed landscape that include fields for rice, wheat, and various vegetables, as well as orchards. Due to the characteristics of the various crops and the agronomic practices applied, the ditches passing through different land-use types should receive different amounts of nitrate. In addition, there may be other variations in physical and chemical properties such as pH and DOC. These factors may affect the microbial activities, correspondingly generating spatial and temporal variability of the denitrification rates in the drainage ditch sediments. Therefore, in this study, we sought to measure sediment denitrification rates in ditches that were located in different land use areas and to determine the key factors that control the sediment denitrification in the study area. Our hypotheses are that (1) sediment denitrification rates would vary spatially and temporally with associated environmental factors; (2) some environmental factors, such as NO_3^- concentrations in surface water and WT, would control the sediment denitrification rates; and (3) the importance of N removal through sediment denitrification can be estimated from the identified relationships.

2 Materials and methods

2.1 Study site

The study was conducted from June 2014 to October 2015 in the Zhushanwan watershed (31° 22′ 34″ N, 119° 57′ 42″ E), Yixing City, Jiangsu Province, China. The climate is a subtropical, humid climate. Rain occurs in about 137 days each year, mostly in moderate to intense rainstorms during the summer (May to September), resulting in a mean annual precipitation of 1203 mm (see Electronic Supplementary Material, Fig. S1). Temperatures range from −2 °C in January to 41 °C in August, with an annual mean temperature of 15.3 °C.

The area of the watershed is 429 km^2, of which 175.6 km^2 (41%) was cultivated including crops such as rice with wheat, vegetables, and peaches. The areas under these three types of cultivation were in the proportions 0.73: 0.17: 0.10. Within each of these three land-use types, a ditch was selected according to several pilot surveys, with the aim of having the dominant waterway type of this territory. The ditch selected in a rice-wheat field was approximately 140 m long, 0.60 m wide, and 0.16 m deep; the wheat and rice were grown in a two-season rotation. The ditch selected in a vegetable field was approximately 340 m long, 0.30 m wide, and 0.33 m deep; vegetables were grown in this field in a four-season rotation of Pakchoi, Chinese cabbage, Peking cabbage, and Potato. The ditch selected in a peach orchard was approximately 180 m long, 0.21 m wide, and 0.10 m deep. Sampling was carried out at six locations at equidistant intervals established along each ditch.

2.2 Sediment denitrification rates measurement

Sampling was initially conducted two times a month in June, July, and August in 2014, and then once a month from September 2014 to October 2015 on a total of 20 dates. Intact sediment samples were collected in cores (10 cm depth; 8 cm inner diameter) using a Uwitec core sampler (length, 30 cm; Uwitec, Austria) from each of the six sites of each ditch. The overlying water on each sampled core was about 10 cm depth. Once collected, the cores were sealed at the bottom with rubber stoppers, and immediately transported to the laboratory (The Agricultural Water-Soil Engineering Laboratory, Hohai University) for denitrification rate measurements.

An incubation experiment was carried out to measure sediment denitrification rates under the standard procedure of the acetylene (C_2H_2) inhibition method (Seitzinger et al. 1993). Approximately 25% of the core length was reserved for headspace sampling. The core was sealed and the headspace was filled with pure C_2H_2 to 10% saturation (10 kPa). Immediately following the addition of C_2H_2, the headspace was reduced and increased alternately by pumping with a large syringe, which promoted the distribution of C_2H_2 (Zhao et al. 2014). Sediments were incubated in environmental chambers for 4 h at ambient temperatures. Gas samples were taken with disposable plastic syringes and transferred to evacuated gas vials at the time point 0 (before C_2H_2 addition) and at the time point 4 h. Gas was sampled with three replications.

The N_2O concentration in the gas sample was determined using a gas chromatograph (Shimadzu, Gas Chromatograph GC-14B) equipped with an electron capture detector (ECD) and a back-flush controlled by a ten-port valve. Two chromatograph columns of different lengths (1 and 3 m) and 2 mm in diameter, packed with Poropak Q (80/100) were used. The ECD temperature was 330 °C and the column temperature was 55 °C. A pure mixture of Ar (95%) and methane (5%) was used as a carrier gas in the ECD at a flow rate of 35 cm^3 min^{-1} into which the gas sample was injected. The N_2O concentration in the liquid phase of the solution in the flask was then calculated according to Henry's law as described by Terry et al. (1981). A N_2O standard gas, used for calibration, was provided by the National Institute of Agricultural Environment of Japan.

2.3 Environmental factors measurement

At each sampling site, the water temperature (WT), pH, dissolved oxygen (DO; % saturation) content, and oxidation-reduction potential (Eh) of the overlying water were measured using a portable parameter detector (Hach Company, Loveland, CO). Additionally, the ditch-overlying water (100 ml) and sediment (about 500 g) were sampled at each of the six sampling locations on the 20 sampling dates. The water samples were transported to the laboratory on ice, where they were filtered and frozen at −4 °C until analysis. The parameters of dissolved organic carbon (DOC), nitrate (NO_3^--N), ammonium (NH_4^+-N), and total nitrogen (TN) contents were analyzed using the filtered water samples. Well-mixed fresh sediment samples (about 15 g each) were mixed with 50 mL of 2 M KCl, and the resulting suspension was shaken, filtered, and analyzed for NO_3^--N and NH_4^+-N. Sediment (about 50 g) from each assay core was dried (60 °C), pulverized (0.15 mm), and analyzed for the total carbon (TC) and total nitrogen (TN) contents. Concentrations of NO_3^--N and NH_4^+-N were determined using a flow injection analyzer (detection limits were 0.015, 0.046, and 0.04 mg L^{-1}, respectively; Skalar Analytical, Breda, The Netherlands); Sediment TC and TN contents were analyzed using a Vario Max CN (detection limits were 0.20 g kg^{-1}; Elementar Americas, Inc., Mt. Laurel, NJ).

2.4 Statistical analysis

Statistical analyses such as frequency distribution, normality tests, and analysis of variance (ANOVA) were conducted using Microsoft Excel (Version 2007) and SPSS (Version 19.0). For non-normally distributed data, a logarithmic transformation was performed to meet the assumptions of parametric statistics. Regression analyses were conducted to determine the relationships between denitrification rates and the measured independent variables. Multiple nonlinear regression models were established incorporating highly correlated variables to estimate nitrogen removal capacity by sediment denitrification in the drainage ditches using 1stOpt (Version 6.0) software (Table 1).

3 Results

3.1 Spatial and temporal variability of sediment denitrification

The sediment denitrification rates measured within the ditches passing through three different types of land use in the Taihu Lake region exhibited high spatial and temporal heterogeneity (Fig. 1). Denitrification rates were 96.4 ± 7.2~8308.3 ± 250.6 μg m^{-2} h^{-1} for the orchard ditches, 76.6 ± 5.5~1415.0 ± 364.8 μg m^{-2} h^{-1} and 97.0 ± 11.4 ~ 675.3 ± 167.3 μg m^{-2} h^{-1} for the vegetable- and rice-wheat-field ditches, respectively. The differences among the denitrification rates over time were significant (ANOVA, $P < 0.05$), with higher values in summer than in the other seasons, with the notable exception of the highest value that was measured in April 2015 in the peach-orchard ditch. The mean sediment denitrification rates were highest in the orchard ditches (1443.4 ± 306.3 μg m^{-2} h^{-1}), followed by those in the vegetable-field ditches and then by those in the rice-wheat-field ditches; this corresponded to the order of NO_3^--N concentrations measured in the surface water samples.

3.2 Factors controlling sediment denitrification

The relationships between sediment denitrification rates and NO_3^--N, NH_4^+-N, TN, WT, pH, Eh, DO, and DOC of the overlying water were examined (see Electronic Supplementary Material, Fig. S2). Denitrification rates in the orchard ditches were positively and linearly correlated with TN ($R^2 = 0.20$, $P = 0.001$), and exponentially correlated with the WT of the overlying water ($R^2 = 0.29$, $P = 0.002$). Similarly, sediment denitrification rates showed an exponential relationship with the WT of the overlying water in both the vegetable-field ditch ($R^2 = 0.31$, $P = 0.031$) and the rice-wheat field ditches ($R^2 = 0.26$, $P = 0.000$). Denitrification rates were positively and linearly correlated with the NH_4^+-N concentrations in the overlying water in the vegetable-field ditches ($R^2 = 0.53$, $P = 0.000$), and with the NO_3^--N concentrations in the overlying water in the rice-wheat field ditches ($R^2 = 0.22$, $P = 0.000$). No significant correlations between denitrification rates and other measured overlying water properties (pH, Eh, DO, and DOC) were detected in any of the three ditches. The results indicated that concentrations of DIN and overlying WT are probably the key factors that control denitrification in the ditch system of the study region.

Table 1 Predicted nitrogen removal capacity by sediment denitrification in drainage ditches in three land-use types of the study area

Ditch	Mean denitrification rates (μg m^{-2} h^{-1})	Area covered by ditches (ha)	Annual N removed in ditches (ton $year^{-1}$)	Coefficient of N removal by denitrification (%)
Orchard	1576.6	35.1	1.8	0.61
Vegetable field	341.4	59.7	0.7	0.77
Rice-wheat field	300.1	250.4	2.5	0.44

Fig. 1 Changes in denitrification rates of ditch sediments (mean value ± standard deviation) and their relationships with water temperature and nitrate concentrations. R^2 coefficient of determination

3.3 Empirical sediment denitrification model

Based on the exponential correlations between denitrification rates with the WT and the linear correlations with DIN concentrations of the overlying water, the sediment denitrification rate (dN; μg m^{-2} h^{-1}) in the ditches could be estimated by a nonlinear mixed-effect model expressed as the following equation:

$$\mathrm{dN} = \mathrm{a} \times \exp(\mathrm{b} \times \mathrm{WT}) + \mathrm{c} \times \mathrm{N} + \mathrm{d} \tag{1}$$

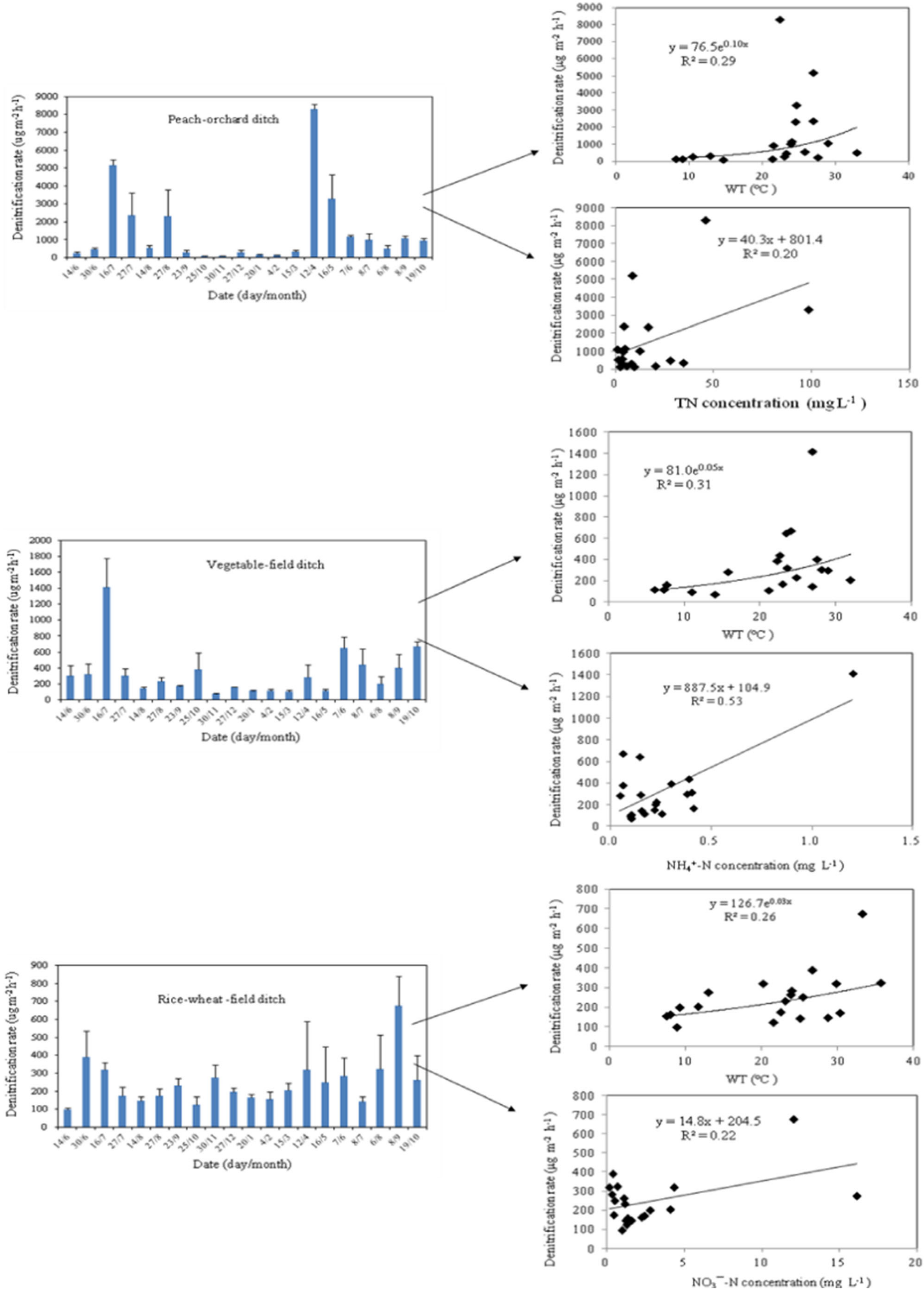
where N is the DIN concentration in the overlying water, and a, b, c, and d are empirically fitted parameters. Using the measured sediment denitrification rate data and those of their key control factors for the period of September 2014 to October 2015, the multiple stepwise regression models of log-transformed denitrification rates were established for each ditch in the three land use types:

$$\log(\mathrm{dN}_{Orchard}) = 1.339 \times \exp.(0.026 \times \mathrm{WT}) + 0.691 \times \log(\mathrm{N}) + 0.059\left(R^2 = 0.69\right) \tag{2}$$

$$\log\left(\mathrm{dN}_{Vegetable}\right) = 0.032 \times \exp(0.108 \times \mathrm{WT}) + 0.594 \times \log(\mathrm{N}) + 1.858\left(R^2 = 0.61\right) \tag{3}$$

$$\log(\mathrm{dN}_{Rice\text{-}wheat}) = 0.042 \times \exp.(0.072 \times \mathrm{WT}) + 0.024 \times \mathrm{N} + 2.097\left(R^2 = 0.45\right) \tag{4}$$

These models accounted for 69, 61, and 45% of the observed variability of sediment denitrification rates in


Peach-orchard ditch
Denitrification rate (ug m-2 h-1)
Date (day/month)
y = 76.5e0.10x
R² = 0.29
Denitrification rate (μg m-2 h-1)
WT (ºC)
y = 40.3x + 801.4
R² = 0.20
TN concentration (mg L-1)
Vegetable-field ditch
Denitrification rate (ug m-2 h-1)
Date (day/month)
y = 81.0e0.05x
R² = 0.31
WT (ºC)
y = 887.5x + 104.9
R² = 0.53
NH4+-N concentration (mg L-1)
Rice-wheat -field ditch
Denitrification rate (ug m-2 h-1)
Date (day/month)
y = 126.7e0.03x
R² = 0.26
WT (ºC)
y = 14.8x + 204.5
R² = 0.22
NO3−-N concentration (mg L-1)

the orchard and vegetable- and rice-wheat-field ditches, respectively (Fig. 2).

To validate how well these models could predict the sediment denitrification rates in the three ditches, the sediment denitrification rate data measured between June 2014 and August 2014 were used for comparison with the model predictions because of their great variation and good representation of the data. The predicted values were calculated using the monitored NO_3^--N concentration and WT data for the three ditches according to the mixed-effect models (Fig. 3). These models accounted for 75, 41, and 51% of the variability in the observed denitrification rates in the orchard, and vegetable- and rice-wheat-field ditches, respectively. Therefore, the empirical models were considered suitable for estimating the sediment denitrification rates of the ditch systems in the study region.

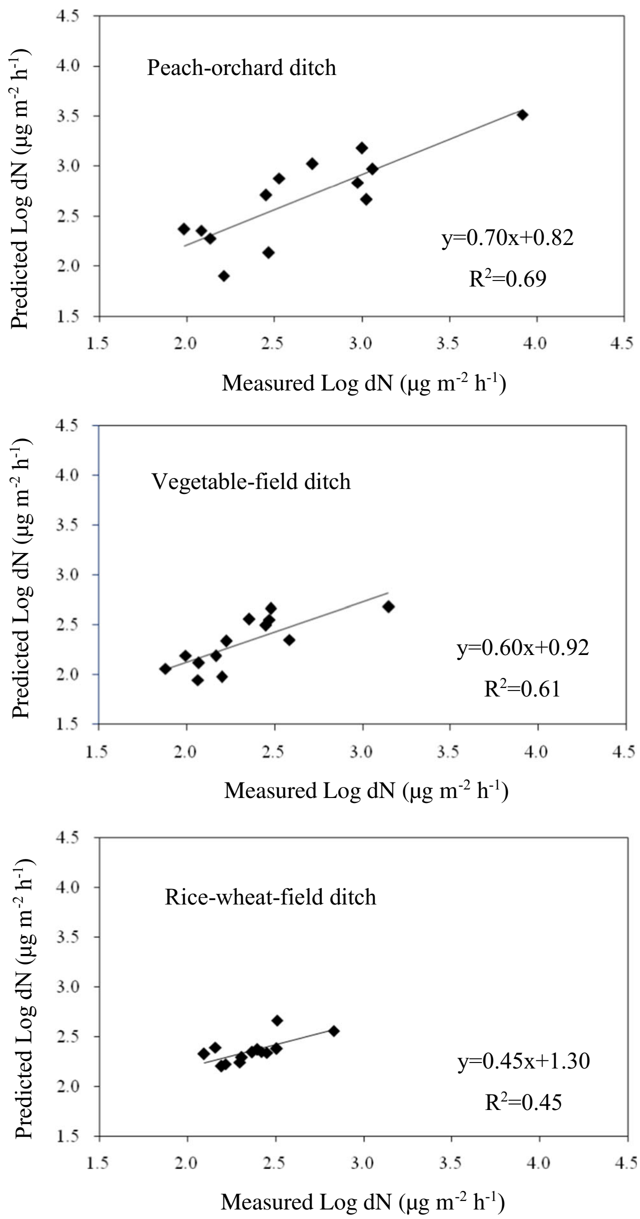


Fig. 2 Measured versus predicted denitrification (dN) rates using established models for ditches in three land use types

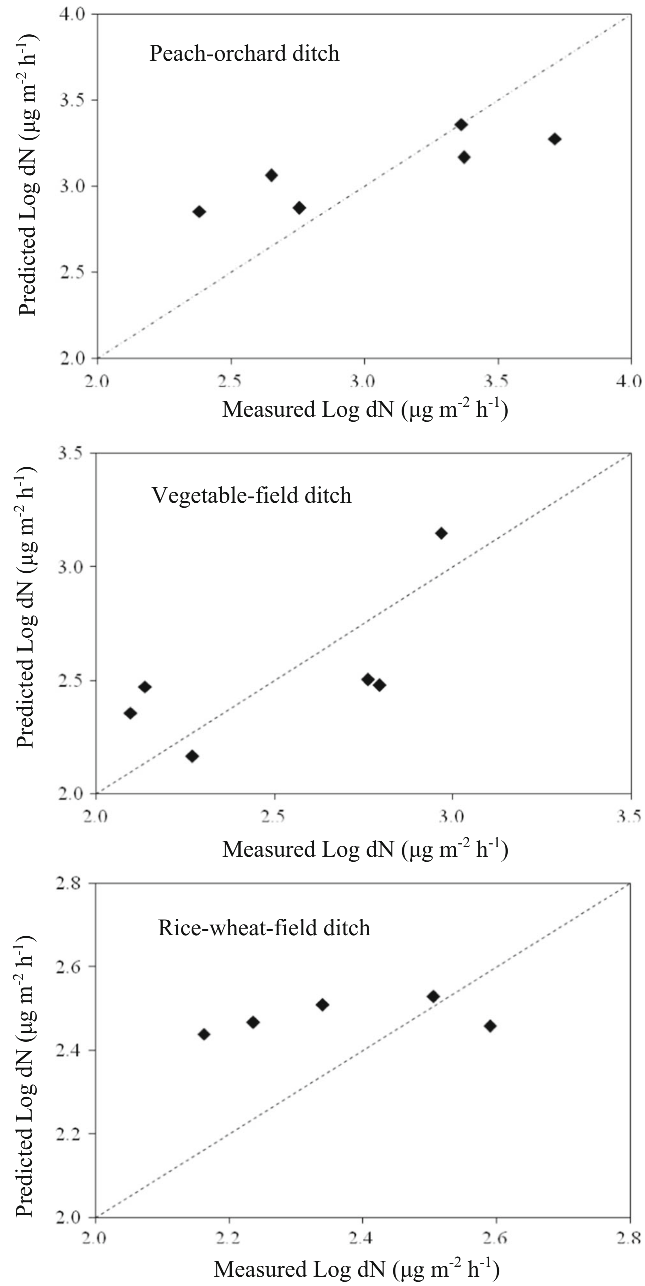


Fig. 3 Validation of denitrification (dN) rate predictions for ditches in three land-use types

3.4 Annual N removal by sediment denitrification in drainage ditches

According to the definition of the denitrification rate, the nitrogen removal capacity by sediment denitrification in the drainage ditch system can be calculated as (Seitzinger et al. 1993):

$$N_{loss} = dN \times A \times D \tag{5}$$

where, N_{loss} is the N removal capacity (kg N year^{-1}); A is the total area of the ditches (ha) (the ditch areas in the Zhushanwan watershed are 35.1, 59.7, and 256.4 ha in the orchard, and in the vegetable and rice-wheat fields, respectively, and were equivalent to 2% of the area of each land-use type); and *D* is the runoff resident time in the ditch system (day) (we set the runoff retention time in the ditch network as 136.6 days annually, which is the same as the annual number of rainy days in the region (Xing and Zhu 2002; Zhao et al. 2014).

To estimate the dN in each ditch using Eqs. (2) to (4), the dissolved inorganic nitrogen concentration (*N*) and the WT data of the overlying water were needed. The annual mean WT of the overlying water in the ditches in the Taihu Lake region is 16.2 °C. However, it is difficult to measure the actual N concentration (*N*) of the overlying water in the ditch system. Even so, the annual mean N in each ditch system can be estimated as follows:

$$N = N_{input} \times E \Big/ (R \times ER \times A) \tag{6}$$

where, N_{input} is the N input in the field (kg ha^{-1}); *E* is the N runoff coefficient of the ditches; *R* is the rainfall (mm); ER is the rainfall runoff coefficient, which is 0.36 in the Taihu Lake

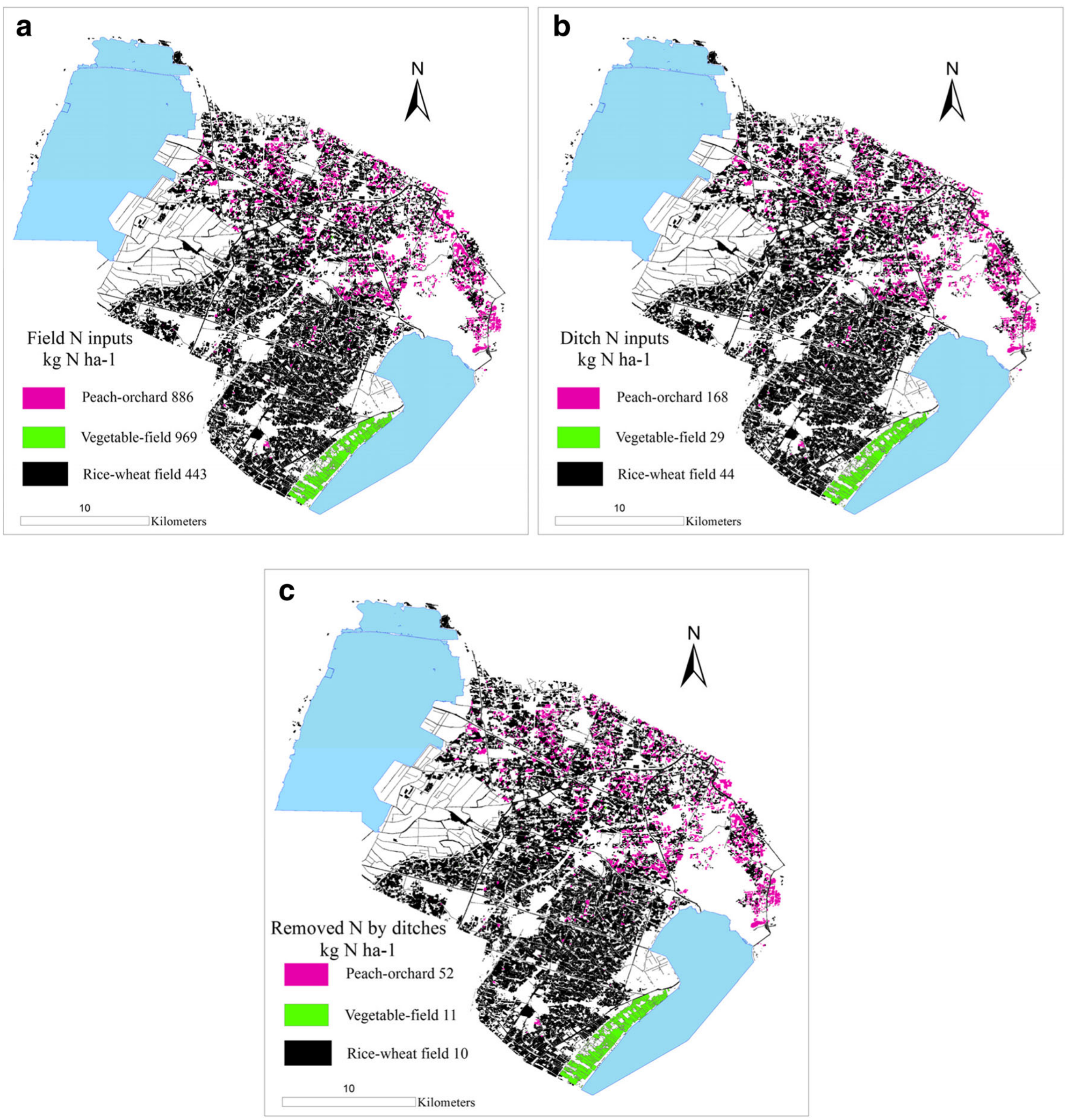


Fig. 4 Distribution of **a** N inputs into the fields, **b** N inputs into the ditches from the fields, and **c** N removal by denitrification per year in the ditches of the study area

region. According to 10 years of field monitoring data, the annual fertilizer N inputs were 886, 969 and 443 kg N ha^{-1} (Fig. 4), and the N runoff coefficients (E) were 0.19, 0.03, and 0.10 for the orchard and the vegetable and rice-wheat fields of the Zhushanwan Watershed, respectively (Xing and Zhu 2001; Zhao et al. 2014) (Table 1).

Based on our established N removal estimation model (Eqs. (2)–(6)), the total annual N removals were 1.8, 0.7 and 2.5 t N year^{-1} via the orchard, and vegetable- and rice-wheat-field ditch systems, respectively (Fig. 5). Thus, the N removal coefficients for the respective ditches were 0.61%, 0.77% and 0.44%. The removal potentials for NO_3^- through flow-through ditch wetlands were very limited. The total N input was as much as 945.3 t into the rivers from the ditch systems in the Zhushanwan Watershed, which was still at a seriously high level.

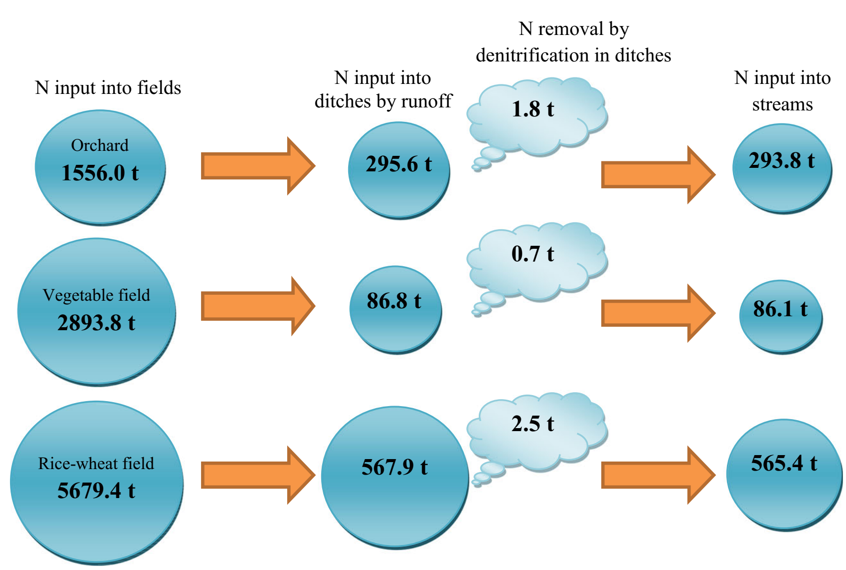
4 Discussion

There were great variations in denitrification rates as reported in wetlands, including ponds, rivers, reservoir, and lakes. For example, Smith et al. (2006) reported denitrification rates ranging from 0 to 4400 μmol m^{-2} h^{-1} in two N-rich streams (range from 0.28 to 14.0 mg l^{-1}) in the upper Mississippi River Basin, USA. Li et al. (2013) measured net sediment denitrification in the pond–stream–reservoir continuum, and found that net denitrification rates ranged between 23.7 ± 23.9 and 674.3 ± 314.5 μmol m^{-2} h^{-1}. Similarly to these studies, our measured denitrification in ditches also exhibited high spatial and temporal heterogeneity. Furthermore, variations of net denitrification rates in ditches (76.6 ± 5.5~8308.3 ± 250.6 μg m^{-2} h^{-1}) were in the range reported in other wetland systems. Thus, large variations may be a characteristic feature of sediment denitrification in aquatic systems.

The great variations in denitrification were attributed to N concentrations and temperature. Our study showed denitrification rates in the ditches had an obvious summer peak and were significantly higher in the peach-orchard ditches than in the other two ditches ($P < 0.05$). These trends were well correlated with the peaks in the WT and NO_3^--N concentrations in the overlying water. In summer, there were heavy runoff and the high application of fertilizers. High field N runoff levels and N enrichment in the drainage water typically occur synchronously with heavy rainstorms (Fig. 1 and Fig. S1, Electronic Supplementary Material). The significance of N to denitrification was verified by the laboratory incubation experiments, which showed that denitrification rates in freshwater sediments responded strongly to the addition NO_3^--N (data not shown). Moreover, the activity of denitrifying bacteria was likely enhanced by the high temperature (Guntinas et al. 2012). Previous studies have also identified the N concentrations and temperature in surface water as the main factors that control denitrification in sediments (Pfenning and McMahon 1997; Zhao et al. 2014). For example, Zhao et al. (2015) reported a summer peak in the net N_2 production rates of a river system in a high N loading region, and indicated that the seasonal pattern primarily corresponded to the interactive effect of NO_3^- levels and WT.

Interestingly, we found the same control factors in the ditch system as in the river system in the Taihu Lake region (Zhao et al. 2015). It appears that rivers and ditches have the same activity of denitrifier bacteria. Moreover, according to the correlation coefficients of independents, it appears that water nitrate concentrations are more important than WT in controlling the seasonal changes in denitrification. In rainy season (from June to September), abundant N runoff together with high temperature enhance N removal by

Fig. 5 Transitive relationships of agricultural N inputs per year into fields, ditches, and streams in the study area



denitrification though residence time is short. However, in dry season (from October to May), N removal in ditches is very limited because of the low N flow even the residence time is long. Therefore, N removal in ditches during the rainy season should be incorporated into estimating models in future studies and used in formulating management techniques to mitigate severe water N pollution.

Wetland systems have been considered to be effective sinks for mitigating agricultural N pollution because of high denitrification rates. However, our results indicated that this potential for removal of NO_3^- through a flow-through ditch system may, in fact, be very limited. Established denitrification rate models incorporating the total area of the ditches and the resident time (Eq. (5)) determined that less than 1% of the total N was removed annually by denitrification in the study ditch system (Fig. 5). This was considerably lower than we had expected. There are main two reasons for the low N removal efficiency in the ditch system in the Taihu Lake region. First, the area of the ditch system is low, only 2% of the total cultivation area. In comparison, Zhao et al. (2015) estimated that removed N accounted for about 43% of the total aquatic N load into the river system, where the river system was about 6.5% of the total area of the region. Second, the residence time is short (2–3 h) because of the high flow rates and short travel distance in the ditches. The water flow in the ditches in our study region ranged from 0.5 to 1.2 m s^{-1} during draining events, much faster than that in the river system (ranging from 0.01 to 0.3 m s^{-1}) (Xia et al. 2013). Alexander et al. (2000) reported that nitrogen removal by denitrification in corridor wetlands, such as ditches and rivers, decreased with increasing stream flow. Yu et al. (2006) indicated that the N removal doubled when retention time increased from 1 to 5 days. BryantMason et al. (2013) also found a limited N removal capacity of river corridor wetlands (7% retention of NO_3^--N) in the Atchafalaya River Basin during the 2011 Mississippi River floods. The low N removal efficiency indicates that the majority of agricultural runoff NO_3^- was transported to the river system.

Management is required to make the ditch system an effective sink for N. This would be achieved by controlling flow rates through the corridor wetland, possibly with a system of sluice gates or ponds, for example, to promote low flow rates and high residence times. In the southeastern USA, when residence time increased to 3 days with a sufficient carbon source, the NO_3^- removal efficiency reached 90% by denitrification in a wetland (Misiti et al. 2011). Another potential way is improving denitrification rate by improving bacteria activity in the ditch system. Within vegetated cannels, canal networks may remove relevant fractions of excess N from agriculture via microbial denitrification, and that vegetation provides multiple interfaces that greatly support the activity of denitrifier (Castaldelli et al. 2015). Thus, ecological remediation of ditches including vegetation and riparian buffer strips should be among the strategies considered for this purpose.

5 Conclusions

In this study, a field investigation and laboratory analyses were conducted to investigate the sediment denitrification characteristics of drainage ditches, which create corridor wetlands, passing through different land-use types, i.e., a vegetable field, a rice-wheat field, and a peach orchard, in the Taihu Lake region. The denitrification rates measured in the sediments from the three types of ditches exhibited high spatial and temporal heterogeneity. Denitrification differed significantly among the land-use types, which was due to large differences in N-fertilization that affected NO_3^- concentrations in the overlying water. The sediment denitrification rate in the ditches could be estimated by a nonlinear mixed-effect model that incorporated the correlations between denitrification rates and their key control factors (NO_3^--N concentrations and WT of the overlying water). Based on estimating data on N concentrations and temperature of overlying water and our established nonlinear mixed model, annual N removal by denitrification was estimated less than 1% of the total N loading in the ditch system. This was much lower than we had expected. We attributed this low removal to the short retention times of the water in the ditches.

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