SEASONAL PATTERN OF N₂O CONCENTRATIONS AND EMISSIONS IN THE SEWAGE-ENRICHED RIVERS: CASE OF CHAOHU LAKE BASIN IN SOUTHEAST CHINA

YANG, L.B.¹,² – LI, F.²* 

¹River and Coastal Environment Research Center, Chinese Research Academy of Environmental Sciences, Beijing 100012, China
²Institute of Geographic Sciences and Natural Resources Research, Chinese Academy of Sciences, Beijing 100101, China

Both authors contributed equally to this work, in no particular order.

*Corresponding author: LI, F.
e-mail: lifcas@gmail.com

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Abstract. Two sewage-enriched urban rivers within Chaohu Lake Basin in Southeast China were selected to study the seasonal variations in Nitrous Oxide (N₂O) concentrations, to improve the information of N₂O concentrations and emissions in the similar areas. The results indicate that N₂O concentrations ranged 16.12~1043.09 nmol/L and averaged 266.27 ± 250.28 nmol/L during the sampling period. N₂O was oversaturated in both rivers ranging 184~12084% (mean 2964%). The significantly higher level of N₂O concentrations and emissions were observed in cold months. It is indicated that the two rivers were net sources of atmospheric N₂O. N₂O emissions ranged 11.32~2920.38 µg N-N₂O/m²/h with an overall mean value of 743.57 ± 831.78 µg N-N₂O/m²/h. A significant negative correlation exists between N₂O concentration and water temperature, which may be regulated by inverse tendencies between temperature and riverine nutrient loadings. Dissolved Oxygen (DO) is a predictor of N₂O in Nanfei River explaining 60% of variability in N₂O; while water temperature and NO³⁻ are better predictors of N₂O in Ershibu River explaining 73% of variability in N₂O. The riverine N₂O may be produced by denitrification in Nanfei River while by coupled nitrification-denitrification in Ershibu River.

Keywords: nitrous oxide, concentration, nitrification, flux, urban river

Introduction

Nitrous Oxide (N₂O) is one of the most significant greenhouse gases contributing to the destruction of stratospheric ozone and climate change (Ravishankara et al., 2009; Wójcik-Gront et al., 2015; Pauleta, et al., 2019). The atmospheric N₂O has risen from about 270 ppb in the pre-industrial era to 319 ppb in the present, mainly due to human perturbations of the global nitrogen (N) cycle (Holland et al., 2005). As N inputs increase, potentially more N₂O is produced (Bouwman, 1995; Wójcik-Gront et al., 2015). In river networks, more than 0.68 Tg/yr of anthropogenic N inputs were converted to N₂O on the global scale, equivalent to 10% of the anthropogenic N₂O emission rate (Beaulieu et al., 2011; Quick et al., 2019; Luo et al., 2019). Therefore, more and more attentions have been paid to the emissions of N₂O from different sources as accurate information is required to determine the contribution of N₂O to global greenhouse gas fluxes (Khalil et al., 2002; Yang and Lei, 2018).

Among the many natural and anthropogenic N₂O sources, most are biological ones (Short et al., 2013). In aquaculture systems, N₂O can be produced by denitrification under anaerobic conditions, where NO³⁻ was converted to N₂O and dinitrogen (N₂)
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(Öquist et al., 2004). Under oxic conditions, N\textsubscript{2}O is also produced as a by-product of nitrification where ammonium (NH\textsubscript{4}\textsuperscript{+}) was oxidized to nitrite (NO\textsubscript{2}\textsuperscript{-}), and subsequently to NO\textsubscript{3}\textsuperscript{-} (Stein and Yung, 2003). Natural rivers have shown large temporal and spatial variations in their N\textsubscript{2}O production and emissions due to the changes of topographic feature and environmental factors (Wilcock et al., 1998; Bansal et al., 2015; Quick et al., 2019). The exact mechanisms of aquatic N\textsubscript{2}O production are related to the various specific environmental conditions, such as N species and loads, dissolved oxygen (DO) concentration, pH, and others. The classification of carbon source also influences N\textsubscript{2}O emission from denitrification and nitrification. Controlling factors related to N\textsubscript{2}O production and emissions in rivers are still needed to fully understand due to lack of \textit{in situ} observation data. Thus, more investigation is needed to determine the underlying mechanism of seasonal variation in N\textsubscript{2}O concentration in rivers.

Emissions of N\textsubscript{2}O from riverine ecosystems take significant influences on the global climate change, since riverine ecosystems are very sensitive to human activities and often receive high loadings of nutrient and organic matter (Richey et al., 2002; Bansal et al., 2015; Yang and Lei, 2018; Quick et al., 2019). Much work has been dedicated to quantifying N\textsubscript{2}O emissions from terrestrial ecosystems; however, emissions of N\textsubscript{2}O from rivers and streams have received much less attention and remain a major source of uncertainty in the global N\textsubscript{2}O budget. Similar to many developing countries, river N pollution is one of the most critical environmental problems in China (Richey et al., 2002; Yang et al., 2006, 2013; Wang et al., 2007; Luo et al., 2019). The related researches on China’s greenhouse gas emission primarily focus on the estuary and coastal areas (Yang et al., 2006; Wang et al., 2007; Liu et al., 2015; Yang and Lei, 2018). However, few studies actually directly measure N\textsubscript{2}O fluxes from N enriched rivers, and that information is sparse.

Chaohu Lake is a shallow lake in Southeast China, which has suffered from serious pollution resulting from substantial discharge of urban wastewater and agricultural runoff. It has been reported that, the annual mean concentration of Total Nitrogen (TN) and Total Phosphorus (TP) over 2001-2016 ranged 0.08-24.60 and 0.02-2.46 mg/L, respectively, meanwhile, the lake water quality showed no substantial improvement (Yang et al., 2013; Yang and Lei, 2018). The Nanfei River (NR) and Ershibu River (ER) are two urban tributaries of Chaohu Lake. These two rivers are narrow and short with low flow speed. Large amounts of N-enriched sewage inputs into the two rivers result in highly eutrophic river water. Significantly, small rivers are thought to be hot spots of nitrogen cycling (Alexander et al., 2000; Bansal et al., 2015; Quick et al., 2019) and can remove a large proportion of nutrient inputs (Wollheim et al., 2006; Luo et al., 2019; Quick et al., 2019), hence could be important sites of N\textsubscript{2}O emissions. Therefore, this study aimed to determine: (1) the presence and extent of seasonal variations in river water chemistry, dissolved N\textsubscript{2}O concentration and emissions in sewage-enriched rivers in the Chaohu Lake region; (2) the potential factors that regulate the seasonal N\textsubscript{2}O variability.

Materials and methods

\textit{Sites description}

The study was carried out in NR and ER, two tributaries of Chaohu Lake. The mean annual temperature and precipitation of this region is 15.5 °C and 950 mm respectively. The NR and ER are small rivers with a total length of 70 and 17 km respectively.
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(Table 1). In this study, the sampling site was set up at the downstream of each river and located at urban areas (Fig. 1). The respective mean water depth of the sampling site is 1.8 m and 3.2 m in NR and ER, and the respective mean flow speed is 0.2 and 0.3 m/s. In recent years, the sewage loading of NR and ER have significantly increased as a result of the effects of local development, which may have become atmospheric N$_2$O sources. Increases in riverine N and organic matter discharge also stimulate microbial processes and associated algae blooming in Chaohu Lake.

Table 1. Physical characteristics of the study rivers

<table>
<thead>
<tr>
<th>River</th>
<th>River length (km)</th>
<th>Mean flow speed (m/s)</th>
<th>Mean water depth (m)</th>
<th>Dominant land use (%)</th>
<th>Sampling location</th>
</tr>
</thead>
<tbody>
<tr>
<td>NR</td>
<td>70</td>
<td>0.2</td>
<td>1.8</td>
<td>Urban (&gt; 70)</td>
<td>Daxing port</td>
</tr>
<tr>
<td>ER</td>
<td>17</td>
<td>0.3</td>
<td>3.2</td>
<td>Urban (&gt; 80)</td>
<td>Longtang bridge</td>
</tr>
</tbody>
</table>

(Diagram of sampling sites and rivers)

Sample collection and chemical analysis

Triplicate surface water (0.2 m depth) samples were collected monthly during Jan. and Dec. in 2017 for the measurement of NH$_4$+, NO$_3$-, TP, BOD$_5$, COD$_{Mn}$, SO$_4^{2-}$, and Cl$. $Dissolved oxygen (DO), pH, and water temperature were measured in situ using a portable meter (HQ30D, USA). Samples for dissolved N$_2$O analysis were collected in 60-ml serum bottles sealed with a butyl-rubber stopper, and preserved after the addition of a few drops of saturated mercuric chloride solution to prevent their biological activities. Water samples were stored in ice box during transport and analyzed within 24 h.
The respective concentration of NH$_4^+$, NO$_3^-$, TP, BOD$_5$, COD$_{Mn}$, SO$_4^{2-}$, and Cl$^-$ in water samples was determined according to the standard methodology of GB3838-2002 promulgated by the China central government. The headspace-equilibrium method was used for measurement of initial sample dissolved N$_2$O concentrations in river water (Huttunen et al., 2002; Liu et al., 2015; Quick et al., 2019; Luo et al., 2019). Twenty milliliters of highly purified N$_2$ (purity > 99.999%) was injected into the serum bottle using an airtight syringe and a 20-ml water sample was displaced. Bottle headspace N$_2$O concentrations were directly analyzed using a gas chromatograph (HP5890 II) equipped with an electron capture detector (ECD) after the bottles were vigorously shaken for 4 h. Initial N$_2$O concentrations (C$_w$) in water samples were calculated (Johnson et al., 1990; Yang and Lei, 2018). The equilibrium concentration (C$_e$) of N$_2$O in river water with atmosphere was calculated using Henry’s first law. Dissolved N$_2$O saturation, expressed in %, was calculated by comparing C$_w$ and C$_e$.

**N$_2$O emission flux**

N$_2$O emission flux across the water-air interface was estimated using the two-layer model:

$$F = k \times (C_w - C_e)$$  \hspace{1cm} (Eq.1)

where $F$ is the N$_2$O emission flux ($\mu$g N-N$_2$O/m$^2$/h); $C_w$ and $C_e$ is the measured and equilibrium concentration of N$_2$O (nmol/L), respectively; $k$ (cm/h) is the gas transfer coefficient, calculated by the equation accounting for both wind speed and flow speed (Borges et al., 2004; Liu et al., 2015):

$$k = [1 + 1.719(w/h)^{0.5} + 2.58\mu_10] (S_e/600)^{-1/2}$$  \hspace{1cm} (Eq.2)

where $w$ and $h$ are flow speed (m s$^{-1}$) and depth (m) of river water column, respectively; $\mu_10$ (m/s) is the instantaneous wind speed at the 10 m height; $S_e$ is the Schmidt number for N$_2$O calculated by the equation proposed firstly by Wanninkhof (1992): $S_e = 2301.1 - 151.1t + 4.7364t^2 - 0.059431t^3$, where $t$ (°C) is in situ river water temperature. In this study, wind speed data was gained from the Hefei Meteorologic Bureau, Anhui Province.

**Results**

**Water chemistry**

NH$_4^+$ and NO$_3^-$ concentrations varied considerably over the sampling time, ranging 1.97~23.75 (mean12.24 ± 5.13) and 0.05~6.98 (mean3.54 ± 4.45) mg/L (Fig. 2a, b), respectively. NH$_4^+$ and NO$_3^-$ concentrations indicated a generally higher level in cold months (P < 0.001). Difference in mean NH$_4^+$ and NO$_3^-$ concentrations was not significant between the two rivers (P > 0.05). DO concentration ranged 0.20~7.70 mg/L with an overall mean of 2.61 ± 2.14 mg/L (Fig. 2c). In NR, DO concentrations were generally lower than 2 mg/L. Similar to the patterns of NH$_4^+$, TP concentrations (ranging 0.20~2.37 mg/L, mean 1.16 ± 0.53 mg/L) showed the marked temporal variations and reached the lowest value in July in both rivers (Fig. 2d). Water temperature ranged 11.0~33.3°C (Fig. 2e), significantly negatively related with DO (P < 0.05). The pH value of river water ranged 6.60~8.56 and decreased dramatically after August (Fig. 2f).
Figure 2. Seasonal variations in \( \text{NH}_4^+ \), \( \text{NO}_3^- \), DO, TP, water temperature (T), and pH value of Nanfei and Ershibu River

Monthly concentrations of BOD\(_5\), COD\(_{\text{Mn}}\), SO\(_4^{2-}\), and Cl\(^-\) in water samples are shown in Table 2. BOD\(_5\) and COD\(_{\text{Mn}}\) concentrations ranged 3.6-23.1 and 7.5-20.4 mg/L, with an overall mean of 9.80 ± 5.37 and 11.97 ± 4.44 mg/L, respectively. Significantly higher BOD\(_5\) and COD\(_{\text{Mn}}\) concentrations were investigated in spring and winter (\( P < 0.001 \)). SO\(_4^{2-}\) and Cl\(^-\) concentrations ranged 45.65~332.50 (mean 101.43 ± 79.06) and 28.55~277.14 (mean 107.16 ± 56.57) mg/L, respectively, generally lower in summer.

Compared to other rivers in this region, the NR and ER showed extremely high level of \( \text{NH}_4^+ \), \( \text{NO}_3^- \), TP, BOD\(_5\), COD\(_{\text{Mn}}\), SO\(_4^{2-}\), and Cl\(^-\) concentrations. The long-term dynamics of water quality in Chaohu river nets were observed over 13 years (Yang et al., 2013; Yang and Lei, 2018), indicating that \( \text{NH}_4^+ \), COD\(_{\text{Mn}}\), and TP concentrations in the Fengle and Hangbu river which drained agricultural runoff averaged 0.25, 0.07, 4.17 mg/L and 0.31, 0.10, 3.47 mg/L, respectively. Based on the latest investigation, the respective mean SO\(_4^{2-}\) and Cl\(^-\) concentration was 29.8 and 23.5 mg L\(^{-1}\) in Fengle River, and 57.1 and 37.9 mg/L in Hangbu River.
Table 2. Monthly concentrations of BOD$_5$, COD$_{Mn}$, SO$_4^{2-}$, and Cl$^-$ in NR and ER

<table>
<thead>
<tr>
<th>River</th>
<th>Conc. mg/L</th>
<th>Sampling time (month)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td>1  2  3  4  5  6  7  8  9  10  11  12</td>
</tr>
<tr>
<td>NR</td>
<td>BOD$_5$</td>
<td>9.7 12.0 13.6 23.1 11 12.8 4.6 4.4 6.4</td>
</tr>
<tr>
<td></td>
<td>COD$_{Mn}$</td>
<td>16.2 15.0 13.7 20.4 19.7 12.3 10 11.3 9.1</td>
</tr>
<tr>
<td></td>
<td>SO$_4^{2-}$</td>
<td>73.4 62.6 51.7 79.8 332.5 125.9 71.5 64.5</td>
</tr>
<tr>
<td></td>
<td>Cl$^-$</td>
<td>138.9 115.8 92.7 138.9 154.4 115.7 33.2 80.8</td>
</tr>
<tr>
<td>ER</td>
<td>BOD$_5$</td>
<td>14.7 16.0 17.8 18.0 12.0 7.5 4.0 8.2</td>
</tr>
<tr>
<td></td>
<td>COD$_{Mn}$</td>
<td>11.96 10.0 8.7 19.8 15.9 18.2 8.7 8.1</td>
</tr>
<tr>
<td></td>
<td>SO$_4^{2-}$</td>
<td>69.6 60.3 51.1 143.2 114.2 168.6 73.98 54.7</td>
</tr>
<tr>
<td></td>
<td>Cl$^-$</td>
<td>94.5 82.3 70.1 117.9 110.5 96.9 28.5 58.9</td>
</tr>
</tbody>
</table>

Dissolved N$_2$O concentration and emission flux

Overall, N$_2$O concentration ranged 16.12~143.09 nmol/L and averaged 226.27 ± 250.28 nmol/L (Fig. 3a, b). A clear seasonal variation in N$_2$O concentration was found in each river. Consistent seasonal trends were apparent between rivers. In general, the higher N$_2$O concentrations were observed in colder months. Difference in mean N$_2$O concentration was not significant between the two rivers (P > 0.05). During the sampling period, both rivers were oversaturated in N$_2$O with a range of 184~12084% (mean 2964%).

Figure 3. Seasonal variations in N$_2$O concentration and percentage saturation
This study indicates that NR and ER were net sources of N$_2$O across all the seasons sampled. N$_2$O emission fluxes also followed a distinct seasonal pattern ranging of 11.32~2920.38 µg N-N$_2$O/m$^2$/h (mean 743.57 ± 831.78 µg N-N$_2$O/m$^2$/h) (*Fig. 4*). Significantly lower N$_2$O emission fluxes were investigated in summer.

![Figure 4. Seasonal variations in N$_2$O emission flux](image)

**Discussion**

Compared to some literatures on the temporal and spatial variation of water pollution in the Zhegao River of Chaohu Lake basin (Chu et al., 2011; Yang et al., 2013), in which NH$_4^+$, NO$_3^-$, and TP showed an annual mean concentration of 1.13, 0.63, and 0.11 mg/L respectively. However, low NO$_3^-$/Cl$^-$ ratios (ranging 0.000~0.21, mean 0.04) were observed both in NR and ER in this study; this result further indicates that high riverine N concentrations in these rivers can be strongly attributed to the direct discharge of untreated sewage. Based on the measurements, the contribution of N$_2$O flux in colder seasons accounted for 90% of the annual budget, similar to results from two subtropical reservoirs (Liu et al., 2011, 2015), with lower fluxes of N$_2$O appearing in summer. However, different seasonal variation patterns on water surface N$_2$O emissions were also reported from the Three Gorges Reservoir in China (Zhu et al., 2013; Zhao et al., 2012). Some sewage-enriched rivers showed higher dissolved N$_2$O concentrations and were net atmospheric sources of N$_2$O. Xin’an Tang river (Taihu Lake basin of China) receiving high untreated sewage inputs indicated high N$_2$O concentration and emission flux with respective mean value of 0.48 µg N-N$_2$O/L and 56.1 µg N-N$_2$O/m$^2$/h (Xia et al., 2013). The overall mean saturation and emission was 770% and 1.91 mg N-N$_2$O/m$^2$/d in the Shanghai city river network (Yu et al., 2013), respectively. The urban rivers in Tianjin city (Haihe basin of China) showed high N$_2$O concentration and emission rate in winter with respective mean value of 88.9 nmol/L and 71.4 N-N$_2$O/m$^2$/h (Hu et al., 2013; Liu et al., 2015). Thus, given the rapid progress of global urbanization, it is urgent to better understand the effect of urban rivers on global N$_2$O budget.
Seasonal variation in N$_2$O concentration and its potential control

Seasonal variations in N$_2$O concentration in two sampling rivers were observed in the study. Generally, the significantly higher N$_2$O concentration was observed in colder months. It has been reported that, seasonal patterns of trace gas concentration and emission in aquatic systems were governed by seasonal variability in temperatures affecting water availability, production of substrate precursors and microbial activity (Whalen, 2005; Quick et al., 2019). For example, water temperature positively correlated to N$_2$O saturation and explained 70% of the seasonal variance of N$_2$O saturation in a large and impounded river (Beaulieu et al., 2010). However, a negative correlation between water temperature and N$_2$O concentration was investigated in this study ($P = 0.03$) (Fig. 5). Some studies (Sun et al., 2013; Liu et al., 2015) also found a similar seasonal pattern in N$_2$O concentration and emission in the Yellow River estuary. It is indicated that this seasonal pattern of N$_2$O concentration in the NR and ER was mainly regulated by the inverse tendencies between temperature and riverine nutrient loadings during the study period. Small river discharge and high riverine nitrogen concentration was generally investigated during the cold seasons in this region. This combination of decreased water discharge and increased nutrient loadings is likely to have sustained high N$_2$O concentration in cold seasons. The correlation between NO$_3^-$ and N$_2$O concentration was significant in NR but not significant in ER. These results suggested that N$_2$O may be produced by denitrification in NR while by coupled nitrification-denitrification in ER, which could be further proved by the significant negative correlation between DO and N$_2$O concentration in NR and significant positive correlation in ER. However, N$_2$O concentration was not significantly correlated with NH$_4^+$ and NO$_3^-$ based on the data collected.

It is suggested that none of the environmental variables stood out as a clearly superior predictor of N$_2$O when considered them alone, though several of them showed some predictive value with respect to dissolved N$_2$O concentration. Herein, a stepwise multiple regression that include DO, NH$_4^+$, NO$_3^-$, TP, BOD$_5$, COD$_{Mn}$, water temperature, and pH was conducted to assess predictors of N$_2$O concentration. The results showed that DO is a better predictor of N$_2$O in NR explaining 60% of variability in N$_2$O, indicating N$_2$O production was limited by DO concentration; while water temperature and NO$_3^-$ are better predictors of N$_2$O in ER explaining 73% of variability in N$_2$O, indicating N$_2$O production was NO$_3^-$ limited. Based on all of the observation data, water temperature and NO$_3^-$ are better predictors of N$_2$O explaining 43% variability in N$_2$O. Other investigations also found similar relationships between NO$_3^-$ and N$_2$O concentration in urban rivers (Wang et al., 2015; Quick et al., 2019). However, Yu et al. (2013) reported that N$_2$O production was greatly controlled by DO and NH$_4^+$ level in some urban rivers. As discussed above, river N$_2$O may be produced by denitrification in NR while by coupled nitrification-denitrification in ER. Meanwhile, we thought that, high NH$_4^+$ come from the decomposition of organic nitrogen in urban effluents in urban rivers, while NH$_4^+$ cannot be oxide to NO$_3^-$ rapidly under hypoxic conditions. As a result, river N$_2$O production was NO$_3^-$ limited based on the data collected.

Uncertainty of N$_2$O emission flux

There are many models for gas emission estimation. However, the overall accuracy of model-based gas estimation in rivers remains uncertain, since models were developed...
in different settings and gas transfer coefficients ($k$) estimated with these models have not satisfied all field conditions (Kremer et al., 2003; Pauleta et al., 2019). In open waters, $k$ is usually parameterized as a function of wind speed. It is reported that the primary driving mechanism that regulates $k$ is presumed to be near-surface turbulence from low to moderate wind speeds; while at higher winds, bubble-mediated exchange produced by breaking waves may play significant roles on $k$ (Borges et al., 2004). It is also reported that bottom-generated turbulence that is transported to the surface can significantly affect gas transfer especially in deep flowing waters (Nimmo-Smith et al., 1999; Zappa et al., 2003; Luo et al., 2019). Thus, choices of the models for estimation of $k$ took a considerable bearing on the estimated gas emissions. Gas transfer velocity was directly measured in some studies to improve accuracy of estimating N$_2$O emission, as is however beyond this study.

Figure 5. Simple linear regression analysis of normalized N$_2$O concentration with (a) normalized NH$_4^+$, DO, (b) normalized NO$_3^-$, (c) normalized water temperature ($T$), and (d) normalized DO for sampling rivers.
Herein, 5 wind-based models (Liss and Merlivat, 1986; Wanninkhof, 1992; Raymond and Cole, 2001; Liu et al., 2015; Yang and Lei, 2018) and 2 wind-currents based models (Borges et al., 2004; Clough et al., 2007; Luo et al., 2019) were selected to compare the disparities in $k_{\text{N}_2\text{O}}$ and eventually N$_2$O fluxes among models. The results showed wide disparities in gas transfer velocity estimates between different models. Overall, $k_{\text{N}_2\text{O}}$ ranged 0.03~29.05 cm/h and averaged 4.26 ± 4.22 cm/h (Fig. 6). LM86a gave the lowest estimates while CL07 gave the highest estimation results. BO04 model expressed by Equation 2 gave an intermediate $k_{\text{N}_2\text{O}}$ value (mean 4.77 ± 1.61 cm/h) relative to the other selected models. Generally, estimated $k_{\text{N}_2\text{O}}$ using wind-based models indicated lower level than wind-current based model. Thus, studies that applied wind speed value only to estimate N$_2$O emissions in rivers may be grossly underestimating emissions and their contribution to the global budgets. The results also indicated that providing a range for the model-based N$_2$O emissions might be more reliable than providing a single flux value by using a single model.

![Figure 6. Comparison of gas transfer velocities estimated](image)

**Conclusions**

Seasonal variation in N$_2$O concentrations and emissions of two urban rivers receiving sewage effluents in Chaohu Lake basin of China was investigated, to provide information on N$_2$O concentrations and emissions in the similar areas. The sampling rivers were oversaturated in N$_2$O and were net sources of atmospheric N$_2$O. Generally, significantly higher N$_2$O concentrations and emissions were observed during cold seasons. The simple linear regression analysis showed a negative correlation between N$_2$O concentration and water temperature based on the data collected, which may be regulated by the inverse tendencies between temperature and riverine N loadings during the study period. The results for the predicted production of riverine N$_2$O showed that DO is a better predictor of N$_2$O in NR explaining 60% of variability in N$_2$O, indicating N$_2$O production was limited by DO concentration; while water temperature and NO$_3^-$ are better predictors of N$_2$O in ER explaining 73% of variability in N$_2$O, indicating N$_2$O production was NO$_3^-$ limited. The riverine N$_2$O may be produced by denitrification in Nanfei River while by coupled nitrification-denitrification in Ershibu River.
Although further investigation is clearly needed to answer the question of how best to proceed before any unquestionable conclusion can be drawn, deriving quantitative estimates and the relative implications such as those presented in this study, is helpful to advance the policy debate. In the future, impacts of economic activities and land use on the concentration and emission of nitrous oxide in urban nitrogen-enriched rivers would be the crux in the relative study.

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