

# 1 Seasonal dynamics of closed **shallow** lakes nutrient status controlled 2 by lacustrine groundwater discharge

3 Xiaoliang Sun<sup>1,2</sup>, Yao Du<sup>1,2\*</sup>, Hao Tian<sup>1,2</sup>, Jiawen Xu<sup>1,2</sup>, Huanhuan Shi<sup>1,2</sup>, Yetong Liu<sup>1,2</sup>, Yamin Deng<sup>1,2</sup>,  
4 Yiqun Gan<sup>1,2</sup>, Yanxin Wang<sup>1,2</sup>

5 <sup>1</sup>Key Laboratory of Groundwater Quality and Health (China University of Geosciences), Ministry of  
6 Education, Wuhan 430078, China.

7 <sup>2</sup>School of Environmental Studies & State Key Laboratory of Geomicrobiology and Environmental  
8 Changes, China University of Geosciences, Wuhan 430078, China.

9 Corresponding: Yao Du (yaodu@cug.edu.cn)

10  
11 **Abstract.** Lacustrine groundwater discharge (LGD) and its associated nitrogen (N) and phosphorus (P)  
12 inputs are increasingly recognized as the critical drivers of lake eutrophication. However, the  
13 intermonthly variability in LGD and its influence on lake nutrient dynamics remain poorly understood.  
14 In this study, high-frequency monitoring and hydrochemical analyses were conducted over a full  
15 hydrological year to investigate LGD-related nutrient fluxes and their effects in a typical oxbow lake in  
16 the central Yangtze Basin. Water level data and <sup>222</sup>Rn tracing revealed a seasonal LGD pattern  
17 characterized by an increase **from summer to winter**, followed by a decline **from winter to spring**, with  
18 LGD rates ranging from 35.36 to 51.71 mm·d<sup>-1</sup>. This pattern was regulated by monthly net  
19 precipitation, which controlled the lake level fluctuations and LGD rates. The corresponding N and P  
20 loads varied synchronously with LGD and showed seasonal synchrony with lake N and P  
21 concentrations. Moreover, variations in the N/P ratio carried by LGD regulate the lake water N/P ratio,  
22 thereby influencing its relationship with the dynamic changes in chlorophyll-a. **A large number of**  
23 **typical closed shallow lakes similar to the studied cases exist worldwide, and therefore, the regulatory**  
24 **role of LGD on lake nutrient status revealed in this study can be reasonably extended to other closed**  
25 **shallow lake systems.** This study provides the first evidence that groundwater-driven nutrient loading  
26 influences lake nutrient status on an intermonthly scale, offering new insights and management  
27 strategies for eutrophication control in closed shallow lake systems worldwide.

## 28 29 **1 Introduction**

30 Lakes are vital terrestrial ecosystems with ecological, environmental, and societal importance  
31 (Zedler and Kercher, 2005). A number of studies have highlighted that lacustrine groundwater  
32 discharge (LGD) not only sustains the lake water balance but also serves as a key pathway for nitrogen  
33 (N) and phosphorus (P) inputs, significantly influencing biogeochemical cycling (Meinikmann et al.,  
34 2015; Rosenberry et al., 2015; Luo et al., 2018; Kazmierczak et al., 2020; Lewandowski et al., 2024).  
35 By altering nutrient concentrations, LGD can regulate lake trophic status and contribute to  
36 eutrophication (Shi et al., 2022; Zheng et al., 2024). Thus, understanding LGD's impact on lake water  
37 environments is critical for elucidating eutrophication mechanisms.

38 The LGD rate exhibits pronounced temporal variability over the hydrological year, primarily  
39 influenced by meteorological and hydrological factors (Burnett et al., 2017; Shi et al., 2022; Sun et al.,  
40 2024). In relatively closed lake systems (lacking perennial surface river inflows or where inflowing  
41 runoff has minimal impact on hydrodynamics, water balance, or residence time), meteorological factors  
42 are generally the dominant drivers of LGD. However, the magnitude and direction of these influences  
43 vary regionally. For instance, in Taihu Lake, LGD shows a positive correlation with evaporation (Shi et  
44 al., 2022), whereas in coal mining subsidence lakes, both precipitation and evaporation exhibit negative  
45 correlations with LGD (Jiang et al., 2024). In permafrost regions, LGD is largely restricted to the  
46 summer thaw period (Walvoord & Striegl, 2007; Olid et al., 2022). Although previous studies have  
47 made progress, understanding of LGD dynamics at the monthly scale remains limited due to  
48 insufficient observational frequency. Most existing studies focus on extreme hydrological periods (wet  
49 and dry seasons) or adopt seasonal observations, thus lacking monitoring at higher temporal resolution.  
50 However, the responses of lake and groundwater levels to precipitation and evaporation typically occur  
51 on monthly timescales, making it difficult for low-frequency or seasonal observations to adequately  
52 capture the processes by which water-level variations regulate LGD. In particular, the mechanistic  
53 linkages along the precipitation/evaporation → lake/groundwater level → LGD pathway require  
54 systematic investigation.

55 In existing studies, groundwater is regarded not only as an important source of lake water recharge,  
56 but also as a major contributor of nitrogen and phosphorus inputs to lakes. In some lakes, LGD  
57 contributes more than 50%, and in certain cases, over 90% of the total nitrogen (N) and phosphorus (P)  
58 inputs often exceed those from surface inflows (Stets et al., 2010; Meinikmann et al., 2015; Shi et al.,  
59 2022). Once discharged, groundwater-derived nutrients undergo biogeochemical transformations or are

60 assimilated by aquatic organisms, thereby affecting the primary productivity and eutrophication (Zheng  
61 et al., 2025). For instance, LGD-driven nutrient inputs have been shown to stimulate chlorophyll a (Chl  
62 *a*) production in closed lakes and alter nutrient limitation patterns via shifts in N/P ratios (Xu et al.,  
63 2025). In Taihu Lake, LGD-derived N inputs were found to alleviate N limitation (Zheng et al., 2025).  
64 However, most studies have focused on nutrient flux quantification over discrete periods, with limited  
65 insight into the temporal variability and trophic responses of lakes. **Understanding the monthly-scale  
66 variations and their controlling mechanisms is essential not only for predicting responses to extreme  
67 hydrological events but also for identifying critical periods of nutrient input that drive eutrophication.**

68 Environmental tracers are increasingly applied in studies of LGD. An ideal tracer typically  
69 exhibits significant concentration differences between lake water and groundwater (often spanning  
70 orders of magnitude) and stable chemical properties (Arnoux et al., 2017; Petermann et al., 2018).  
71 Commonly used tracers include  $^{222}\text{Rn}$ ,  $^{226}\text{Ra}$ , stable hydrogen and oxygen isotopes ( $\delta^2\text{H}$ ,  $\delta^{18}\text{O}$ ),  $\text{Cl}^-$ , and  
72 electrical conductivity (Sun et al., 2021). Among these,  $^{222}\text{Rn}$  and  $^{226}\text{Ra}$  often show concentration  
73 differences of up to two-three orders of magnitude between the two water types, whereas differences in  
74  $\text{Cl}^-$  and electrical conductivity are generally smaller (sometimes only several times). Therefore,  $^{222}\text{Rn}$   
75 and  $^{226}\text{Ra}$  are frequently the preferred tracers in LGD studies, with other indicators used as auxiliaries  
76 when conditions permit (Dimova & Burnett., 2011). The applicability of stable hydrogen and oxygen  
77 isotopes is strongly influenced by hydrological stability; in lakes with pronounced seasonal  
78 hydrological fluctuations, their quantitative accuracy may be significantly reduced (Sun et al., 2025a).  
79 Regarding radioactive tracers,  $^{226}\text{Ra}$  primarily desorbs from particles into the water phase in brackish or  
80 saline environments (Webster et al., 1995; Gonneea et al., 2008), and its concentration is typically low  
81 in freshwater lakes. Consequently, in freshwater lake LGD studies,  $^{222}\text{Rn}$  is more commonly used and  
82 effective due to its high solubility, large concentration gradient, and ease of detection.

83 The middle Yangtze River plain is characterized by a high density of lakes, and previous studies  
84 have shown that lakes of different types commonly exhibit groundwater discharge of varying  
85 intensities, along with associated nitrogen and phosphorus input fluxes (Jiang et al., 2022; Hu et al.,  
86 2023; Gan et al., 2023). However, its specific role in nutrient loading and eutrophication remains  
87 poorly understood. Heiwawu Oxbow Lake (HW lake), situated along a former Yangtze River channel,  
88 provides a valuable setting to examine the temporal dynamics of LGD due to its seasonal hydrological  
89 connectivity. Typically isolated from the Yangtze by November, HW lake experienced an earlier

90 disconnection in August 2022 due to extreme drought (Sun et al., 2024). During this period, the lake's  
91 water and nutrient balance depended almost entirely on precipitation and groundwater, offering an ideal  
92 natural experiment to investigate LGD-driven nutrient loading and eutrophication over a complete  
93 hydrological year.

94 In this study, we quantified LGD and its associated nitrogen and phosphorus fluxes by conducting  
95 bi-monthly, high-frequency monitoring of groundwater and lake water levels, and by applying a  $^{222}\text{Rn}$   
96 mass balance model to a typical closed lake in the middle reaches of the Yangtze River. Furthermore,  
97 we examined the influence of meteorological factors on LGD and the dynamic effects of LGD-driven  
98 nutrient inputs on lake trophic status. The results elucidate the role of precipitation and evaporation in  
99 regulating the seasonal dynamics of LGD and highlight the critical influence of LGD on the nutrient  
100 regime of closed lakes. These findings provide a valuable scientific basis for improving water quality  
101 management and mitigating eutrophication in closed-lake systems worldwide.

102

## 103 **2 Materials and Methods**

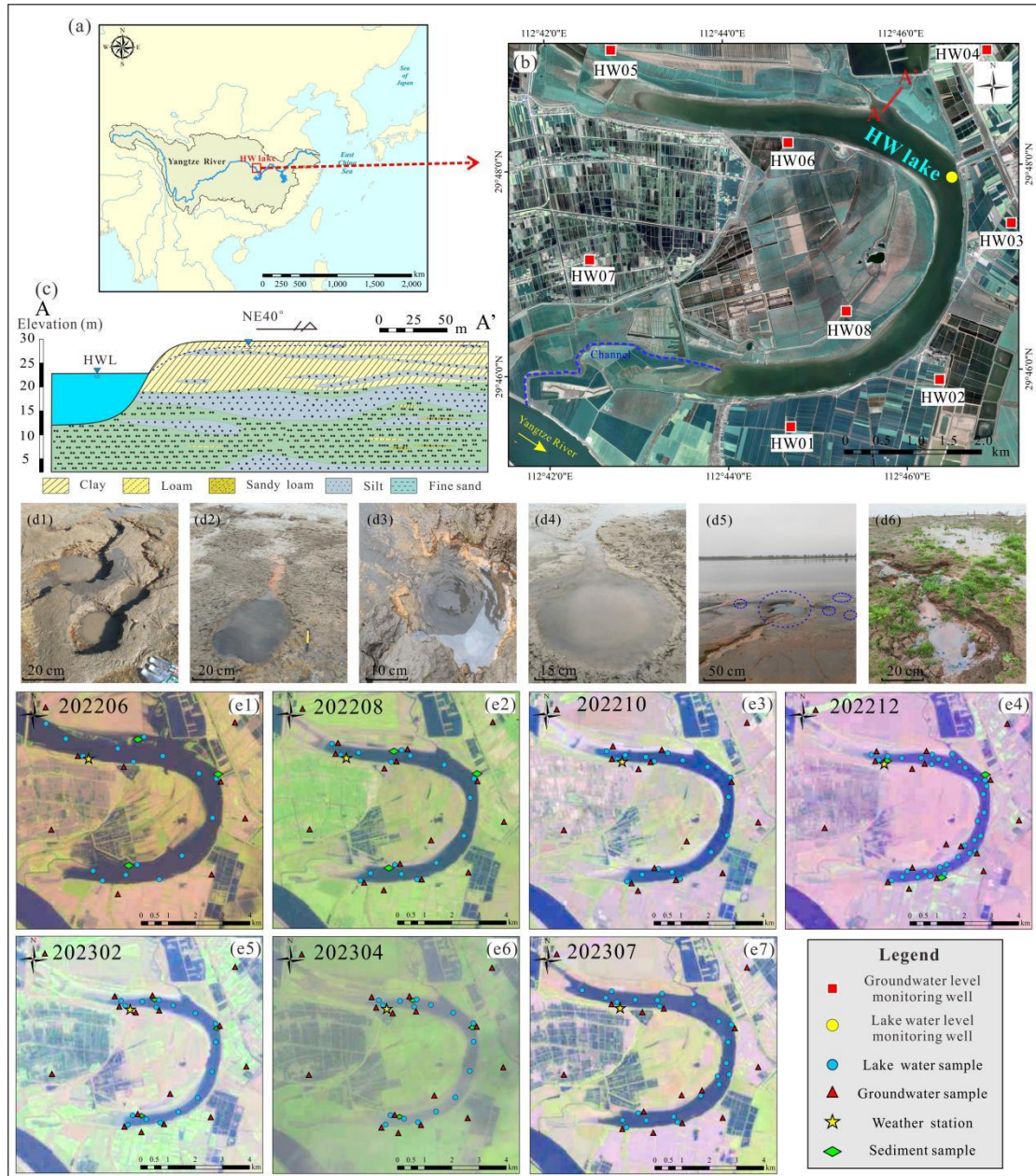
### 104 **2.1 Study area**

105 **HW lake** is situated in the southern Jiangnan Plain on the northern bank of the middle reaches of  
106 the **Yangtze River**, between  $29^{\circ} 45' 10.98'' - 29^{\circ} 48' 52.10''$  N and  $112^{\circ} 41' 35.90'' - 112^{\circ} 46' 32.16''$   
107 E (Figure 1a and b). The region has a subtropical monsoon climate, with an annual mean temperature  
108 of  $15-17^{\circ}\text{C}$ , average annual precipitation of approximately 1400 mm, and average annual evaporation  
109 of approximately 1100 mm. **HW lake** was formed in 1967 following the artificial cutoff of a meander  
110 of the **Yangtze River**, with a downstream channel providing hydrological connectivity between the lake  
111 and the river. During the wet season, elevated **Yangtze River** water levels induce the inflow into **HW**  
112 **lake**, whereas in the dry season, declining **Yangtze River** levels result in backflow from the lake into the  
113 river, ultimately leading to complete disconnection. **In a typical hydrological year, HW lake**  
114 **experiences substantial water level fluctuations between the wet and dry seasons, with a magnitude of**  
115 **up to 8 m.**

116 The aquifer interacting with the oxbow lake comprises two distinct layers (Figure 1c). The upper  
117 layer with 5–15 m thickness consists of low-permeability clay and silty clay with a hydraulic  
118 conductivity of approximately  $1 \text{ m}\cdot\text{d}^{-1}$ . In contrast, the lower layer is a confined aquifer 50–80 m thick,  
119 composed mainly of fine to coarse sand, and exhibits a higher hydraulic conductivity of approximately  
120  $5-15 \text{ m}\cdot\text{d}^{-1}$ . Owing to its greater permeability, the lower aquifer functions as the primary groundwater

121 source directly connected to the lake. Previous studies have confirmed substantial LGD in HW lake,  
122 predominantly occurring as springs (Figure 1 d1–d6) and seepage flows (Sun et al., 2025b). The area  
123 surrounding HW lake is characterized by flat terrain with a slope of less than 0.1‰, and the dominant  
124 land use types include forest and farmland. This indicates that, except for runoff through artificial  
125 channels, surface runoff generated by other precipitation is almost negligible, and the water level  
126 gradient, rather than the terrain slope, becomes the key factor controlling groundwater-lake water  
127 exchange.

128 In 2022, a severe drought event in Yangtze River caused the water level of HW lake to decline in  
129 parallel with that of the Yangtze River, ultimately leading to complete disconnection from the river.  
130 Consequently, the lake entered the dry season approximately two months earlier than that in a typical  
131 hydrological year. During the groundwater discharge period, the water level of HW lake was minimally  
132 affected by hydrological fluctuations in the Yangtze River and was instead primarily controlled by  
133 meteorological factors such as precipitation and evaporation.



134  
 135 **Figure 1.** Overview of the study area. (a) Geographical location of the HW lake (study area marked by a red box,  
 136 from Esri). (b) satellite image of HW lake (from Google Maps). (c) Hydrogeological cross-section along profile  
 137 A–A' (profile location indicated by line A–A' in b). **The cross-section shows only the upper part of the confined**  
 138 **aquifer that interacts with the lake, and does not display the full thickness of the aquifer.** (d1–d6) Zones of  
 139 concentrated LGD phenomena in HW lake, primarily manifested as spring outlets. (e1–e7) Spatial distribution of  
 140 field sampling sites collected between June and July 2022 (from Landsat 8). The numbers in the figure indicate the  
 141 sampling periods. For example, 202208 represents August 2022. **In June 2022 and July 2023, HW lake was**  
 142 **connected to the Yangtze River through a channel, receiving inflow from the river, which led to higher water levels**  
 143 **and an expanded inundated area. In the other months, HW lake was isolated from the Yangtze River, representing**  
 144 **the period when LGD occurred.**

146 **2.2 Field work and laboratory analysis**

147 Systematic field sampling was conducted from June 2022 to July 2023, with a sampling frequency

148 of once every two months. Except for the July 2023 sampling conducted in early July, all other  
 149 campaigns were performed at the end of the even-numbered months. Except for the campaigns in June  
 150 2022, December 2022, and July 2023, all sessions followed a standardized protocol involving the  
 151 collection of 16 lake water samples, 8 well water samples, and 8 pore water samples. A total of 241  
 152 samples were collected (Table 1, Figure 1 e1–e7), comprising 135 lake water samples, 56 well water  
 153 samples, and 50 pore water samples. In addition to sample collection, each campaign included  
 154 simultaneous measurements of environmental parameters, such as groundwater level, lake water level,  
 155 lake depth, and wind speed. Due to seasonal fluctuations in lake water levels, the shoreline position  
 156 shifted over time, resulting in slight spatial variations in the lakeshore sampling points across  
 157 campaigns. The wind speed was continuously recorded at 15-min intervals using a weather station  
 158 installed on the lakeshore, while the precipitation and evaporation data were sourced from the ECMWF  
 159 reanalysis datasets.

160 **Table 1.** Number of samples per sampling campaign.

Time	Number of lake water samples	Number of well water samples	Number of pore water samples	Total
June 23–29, 2022	16	8	2	26
August 26–30, 2022	16	8	8	32
October 26–31, 2022	16	8	8	32
December 17–30, 2022	35	8	10	53
February 20–25, 2023	16	8	8	32
April 23–27, 2023	16	8	8	32
July 6–14, 2023	20	8	6	34
Total	135	56	50	241

161  
 162 Samples from the lake center were collected by boat at fixed preselected locations. Surface water  
 163 was sampled approximately 0.5 m below the surface using a specialized surface water sampler. The  
 164 groundwater samples were categorized into two types: (1) well water collected from monitoring wells  
 165 located 0.5 to 2.5 km from the lakeshore, with the depths between 15 and 30 m, and (2) spring water  
 166 collected from natural springs situated within 15 m of the shoreline and in shallow lake zones, serving  
 167 as the representative lakeshore pore water. In addition, three nearshore lakebed sediment samples were  
 168 collected for incubation experiments to determine the  $^{222}\text{Rn}$  concentrations. The wind speed was  
 169 continuously recorded at 15-min intervals using a weather station installed on the lakeshore, while the  
 170 precipitation and evaporation data were sourced from the ECMWF reanalysis datasets.

171 For  $^{222}\text{Rn}$  analysis, the water samples were collected in 250 mL or 2.5 L glass bottles using an  
 172 overflow method to eliminate residual air.  $^{222}\text{Rn}$  concentrations were quantified using a RAD7 system

173 (Durrige Company, Inc.) equipped with RAD7-H<sub>2</sub>O and RAD7 Big Bottle accessories. To reduce the  
174 measurement uncertainty in lake water samples, the counting time was extended to 60 min per sample.  
175 All <sup>222</sup>Rn analyses were completed within 24 h of sampling, using the RAD7 aqueous system.

176 
$$A_0 = A \times e^{\lambda t} \quad (1)$$

177 where A<sub>0</sub> represents the <sup>222</sup>Rn concentration (Bq·m<sup>-3</sup>) at the sampling time; A represents the <sup>222</sup>Rn  
178 concentration (Bq·m<sup>-3</sup>) at the measurement time; λ represents the decay coefficient of <sup>222</sup>Rn, 0.181 d<sup>-1</sup>;  
179 and t represents the time interval (d) between sampling and measurement. The uncertainty in <sup>222</sup>Rn  
180 testing for lake water was approximately 35%, and the uncertainty in <sup>222</sup>Rn testing for groundwater is  
181 approximately 25%.

182 The samples for total nitrogen (TN) and total phosphorus (TP) analysis were field-filtered using a  
183 0.45 μm membrane filter and then stored in 30 mL polyethylene bottles and 20 mL brown screw-cap  
184 glass bottles, respectively. The TP samples were acidified with concentrated HNO<sub>3</sub> to a pH below 2,  
185 sealed, and subsequently analyzed using an inductively coupled plasma optical emission spectrometer  
186 (ICP-OES, iCAP 6000 series, Thermo Fisher Scientific, USA) at the School of Environmental Studies,  
187 China University of Geosciences (Wuhan), with a detection limit of 0.001 mg·L<sup>-1</sup>. TN samples were  
188 analyzed using a total organic carbon/nitrogen analyzer at the Wuhan Botanical Garden, Chinese  
189 Academy of Sciences, with a detection limit of 0.01 mg·L<sup>-1</sup>. Chl *a* concentrations were determined  
190 immediately after sampling using an AquaFluor fluorometer, with a detection limit of 0.5 μg·L<sup>-1</sup>.  
191 Water quality parameters, including pH, temperature, DO, ORP, and EC, were measured in situ using a  
192 HACH-HQ40D multi-parameter probe.

193

### 194 2.3. <sup>222</sup>Rn mass balance model

195 In recent years, <sup>222</sup>Rn has been widely used in studies of LGD. Derived from its parent isotope  
196 <sup>226</sup>Ra, <sup>222</sup>Rn has a half-life of 3.82 d. By treating lake water as a closed system, a mass balance model  
197 was developed based on both the sources and sinks of <sup>222</sup>Rn. The <sup>222</sup>Rn flux from the groundwater  
198 discharge was inferred as an unknown term. The <sup>222</sup>Rn mass balance model is expressed as follows, and  
199 a conceptual diagram of its budget terms is shown in Figure S1:

200 
$$\frac{\partial I^{222}Rn}{\partial t} = F_g + F_d + I^{226}Ra \times \lambda^{226}Ra - F_a - I^{222}Rn \times \lambda^{222}Rn \quad (2)$$

201 where  $F_g$ ,  $F_d$ , and  $F_a$  are the  $^{222}\text{Rn}$  fluxes ( $\text{Bq}\cdot\text{m}^{-2}\cdot\text{d}^{-1}$ ) of groundwater discharge, sediment diffusion, and  
 202 atmospheric escape, respectively;  $I^{226}\text{Ra}$  and  $I^{222}\text{Rn}$  are the pools of  $^{226}\text{Ra}$  and  $^{222}\text{Rn}$  in the lake water  
 203 ( $\text{Bq}\cdot\text{m}^{-2}$ ), which are equal to the concentrations of  $^{226}\text{Ra}$  and  $^{222}\text{Rn}$  multiplied by the water depth; and  
 204 the decay coefficients of  $^{222}\text{Rn}$  ( $\lambda^{222}\text{Rn}$ ) and  $^{226}\text{Ra}$  ( $\lambda^{226}\text{Ra}$ ) were  $0.181\text{ d}^{-1}$  and  $1.31 \times 10^{-11}\text{ d}^{-1}$ ,  
 205 respectively. The measurement times before and after sampling were the first day and the last day of  
 206 the sampling period, respectively. Since there is almost no difference in the lake water level and  $^{222}\text{Rn}$   
 207 concentration at the same location before and after sampling, the change in the  $^{222}\text{Rn}$  in lake water on  
 208 the left side of the equation can be approximated as 0 (Kluge et al., 2007).

209 The  $^{222}\text{Rn}$  flux diffused from the sediment to the lake is one source of the  $^{222}\text{Rn}$  mass balance  
 210 model, and it is calculated by the following formula:

$$211 \quad F_d = \sqrt{(\lambda^{222}\text{Rn} \times n D_m)} (C_p - C_w) \quad (3)$$

212 where  $C_p$  ( $\text{Bq}\cdot\text{m}^{-3}$ ) and  $C_w$  ( $\text{Bq}\cdot\text{m}^{-3}$ ) are  $^{222}\text{Rn}$  concentrations of pore water in sediments and overlying  
 213 lake water, respectively;  $D_m$  ( $\text{cm}^2\cdot\text{s}^{-1}$ ) is the  $^{222}\text{Rn}$  molecular diffusion coefficient in wet bulk sediment;  
 214  $n$  is the porosity of the sediment.

215 To determine the  $^{222}\text{Rn}$  concentrations in sediment pore water, a sediment equilibrium incubation  
 216 experiment was carried out following the procedure proposed by Corbett et al. (1998). The  $D_m$  is  
 217 expressed as:

$$218 \quad -\log D_m = \left( \frac{980}{T_w + 273} \right) + 1.59 \quad (4)$$

219 where  $T_w$  is water temperature ( $^{\circ}\text{C}$ ). An equilibrium incubation experiment with lakebed sediments was  
 220 carried out to obtain the  $^{222}\text{Rn}$  concentration in sediment pore water (Corbett et al., 1998).

221 The atmospheric loss of  $^{222}\text{Rn}$  is estimated based on the following empirical equation, which is  
 222 related to temperature and wind speed (MacIntyre et al., 1995):

$$223 \quad F_a = K (C_w - \alpha C_a) \quad (5)$$

224 where  $K$  values are in  $\text{cm h}^{-1}$  but have been scaled to  $\text{m}\cdot\text{d}^{-1}$  for input into Eq. (5);  $C_w$  is the  
 225 concentration of lake water  $^{222}\text{Rn}$  ( $\text{Bq}\cdot\text{m}^{-3}$ );  $C_a$  is the concentration of  $^{222}\text{Rn}$  in the air ( $\text{Bq}\cdot\text{m}^{-3}$ );  $\alpha$  is the  
 226 gas distribution coefficient (dimensionless) and is a temperature dependent function;  $^{222}\text{Rn}$  from the  
 227 decay of dissolved  $^{226}\text{Ra}$  can be generally omitted for  $^{222}\text{Rn}$  mass balance model.

228 The LGD rate can be obtained by dividing the  $^{222}\text{Rn}$  flux from groundwater discharge by the  $^{222}\text{Rn}$

229 concentration of surrounding groundwater. The formula is as follows (Luo et al., 2018; Wang et al.,  
230 2019):

$$231 \quad V = \frac{F_g}{C_g} \times 1000 \quad (6)$$

232 where V is the average LGD rate (mm·d<sup>-1</sup>); F<sub>g</sub> is the <sup>222</sup>Rn flux of groundwater discharge  
233 (Bq·m<sup>-2</sup>·d<sup>-1</sup>); C<sub>g</sub> is the <sup>222</sup>Rn concentrations in groundwater end member (Bq·m<sup>-3</sup>). A more detailed  
234 description can be found in Section 1 of the SI.

235

### 236 **3 Results and Discussion**

#### 237 **3.1 Seasonal variations in LGD rates**

##### 238 **3.1.1 Identification of seasonal variations in LGD**

###### 239 1. Lake water and groundwater level

240 The fluctuations in lake water levels and groundwater levels reflect a pronounced interaction  
241 between groundwater and the lake. During the field monitoring period, both exhibited significant  
242 changes in water level.

243 From June 2022 to July 2023, the lake water level fluctuated significantly between 22.30 and  
244 32.90 m, with a peak in late June 2022 and a low in late February 2023, demonstrating a variation of  
245 10.6 m (Figure 2a). Initially, the HW lake's high water level was sustained by its connection to the  
246 Yangtze River, which was at a high level in June 2022. However, a severe drought from August to  
247 September 2022 caused a rapid decline in the Yangtze River level, resulting in a corresponding  
248 decrease in the HW lake level. From late October 2022 to early May 2023, as the Yangtze River fell  
249 below 26.5 m, HW lake became hydrologically isolated, with no surface water inflow or outflow. The  
250 rate of water level decline slowed from October 2022 to February 2023 and began to rise in March  
251 2023 because of increased precipitation. In May 2023, when the Yangtze River exceeded 26.5 m again,  
252 the HW lake level rose sharply due to the inflow.

253 The groundwater levels from the eight surrounding wells exhibited similar fluctuations, ranging  
254 from 26.6 to 30.9 m (Figure 2a). From June 2022 to April 2023, groundwater levels consistently  
255 declined, with a slight increase observed from April to July 2023. Generally, the groundwater level  
256 trends mirrored those of the lake, except for the period from February to April 2023, when the  
257 groundwater levels decreased while the lake levels increased. The difference between the average  
258 groundwater and lake levels ranged from -1.98 to 4.45 m, affecting the LGD rates over time. In June

259 2022, all eight wells had groundwater levels lower than those of the lake, indicating complete lake  
260 recharge. By July 2023, four wells presented lower groundwater levels, whereas four exhibited higher  
261 levels, suggesting simultaneous recharge and discharge. From August 2022 to April 2023, groundwater  
262 consistently discharged into the lake, with the largest groundwater-lake level difference occurring in  
263 December 2022, indicating the highest LGD rate during that period (Figure 2b).

264

265 2. Lake water and groundwater  $^{222}\text{Rn}$  concentration

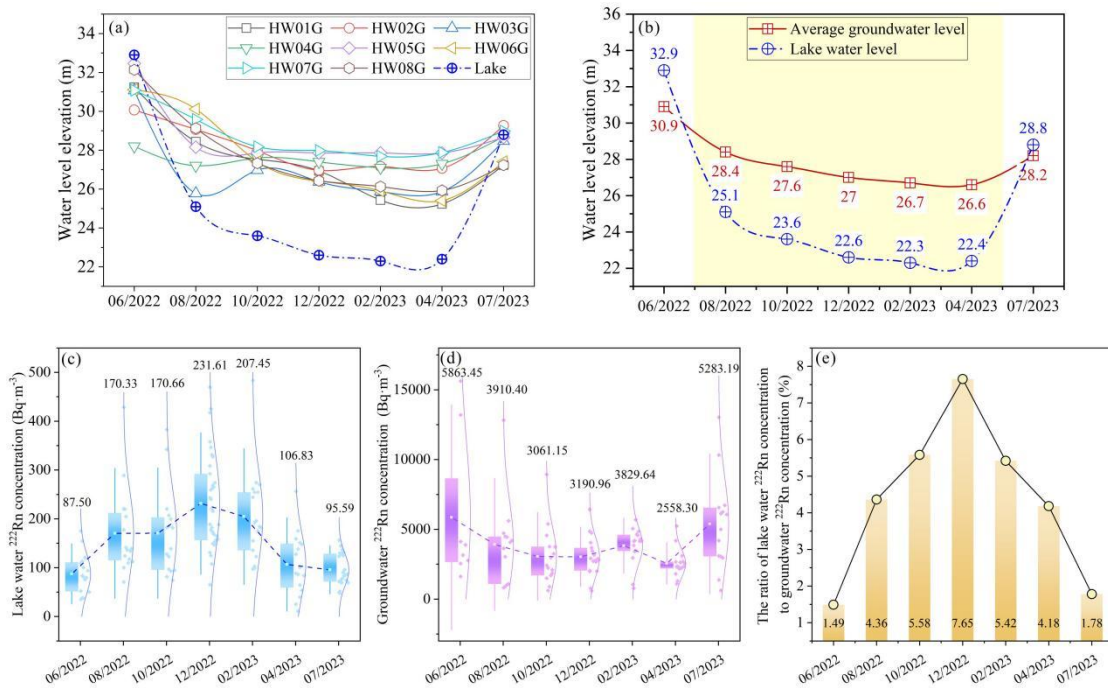
266 As a limited tracer for LGD, the concentration of  $^{222}\text{Rn}$  can indicate changes in LGD. During the  
267 field monitoring period, both lake water and groundwater concentrations underwent some variations,  
268 with differing magnitudes.

269 The fluctuation in the lake water  $^{222}\text{Rn}$  concentration from June 2022 to July 2023 (Figure 2c)  
270 exhibited an initial increase followed by a decrease. The lowest concentration occurred in June 2022,  
271 whereas the highest was recorded in December 2022, with values of 87.50 and 231.61  $\text{Bq}\cdot\text{m}^{-3}$ ,  
272 respectively. Similarly, groundwater also demonstrated a trend of initial decrease followed by an  
273 increase (Figure 2d), with the highest values in June 2022 and the lowest values in April 2023, at  
274 5863.45 and 2558.30  $\text{Bq}\cdot\text{m}^{-3}$  respectively.

275 During the period of groundwater discharge to the lake, when groundwater levels were  
276 consistently above lake levels (August 2022 to April 2023), the lake water  $^{222}\text{Rn}$  concentration ranged  
277 from 106.83 to 231.61  $\text{Bq}\cdot\text{m}^{-3}$ . While, the groundwater  $^{222}\text{Rn}$  concentration ranged from 2558.30 to  
278 3910.40  $\text{Bq}\cdot\text{m}^{-3}$ . Based on the coefficient of variation (CV, used to measure the relative dispersion of  
279 data, defined as the ratio of the standard deviation to the arithmetic mean), the  $^{222}\text{Rn}$  concentration in  
280 groundwater (CV  $\approx$  17.60%) is more stable than that in lake water (CV  $\approx$  26.62%), indicating a  
281 relatively consistent  $^{222}\text{Rn}$  endmember value for groundwater. Because groundwater was the primary  
282 source of  $^{222}\text{Rn}$  in the lake, variations in lake water  $^{222}\text{Rn}$  concentrations reflected changes in the LGD  
283 rate. As shown in Figure 2e, the ratio of lake water to groundwater  $^{222}\text{Rn}$  concentration initially  
284 increased and then decreased with a peak in December 2022, which suggested that the LGD rate was  
285 the highest during that period. This pattern is consistent with the relationship between groundwater and  
286 lake water levels.

287 Notably, as shown in Figure. 2a and c, during the two periods when the groundwater level was  
288 lower than the lake water level (June 2022 and July 2023), lake-water  $^{222}\text{Rn}$  concentrations were 87.50

289 and  $95.59 \text{ Bq m}^{-3}$ , respectively, which are lower than those in the other periods, indicating that  
 290 groundwater discharge was generally limited. Nevertheless, measurable  $^{222}\text{Rn}$  was still detected in the  
 291 lake water, suggesting additional inputs from other water sources. Concurrently,  $^{222}\text{Rn}$  concentrations in  
 292 the Yangtze River were  $90.50$  and  $50.47 \text{ Bq m}^{-3}$ , respectively. In June 2022, the lake-water  $^{222}\text{Rn}$   
 293 concentration was very similar to that of the Yangtze River, indicating that the lake was mainly  
 294 controlled by Yangtze River inputs during this period. In contrast, in July 2023 the lake-water  $^{222}\text{Rn}$   
 295 concentration was slightly higher than that of the Yangtze River; together with the observation that  
 296 groundwater levels in some monitoring wells were still slightly higher than the lake level, this suggests  
 297 the presence of weak, localized groundwater discharge during this period. In addition, intense summer  
 298 precipitation led to increased catchment runoff into the lake, which generally exhibited high  $^{222}\text{Rn}$   
 299 concentrations, primarily derived from  $^{222}\text{Rn}$  released through interactions between rainfall and surface  
 300 soils rather than from aquifer groundwater. Overall, the higher lake-water  $^{222}\text{Rn}$  concentration in July  
 301 2023 compared with June 2022 reflects the combined effects of Yangtze River input, localized weak  
 302 groundwater discharge, and precipitation-driven runoff.



303 **Figure 2.** (a) Variations in lake water level and groundwater levels across all groundwater monitoring points. (b)  
 304 Variations in lake water levels compared with the average groundwater level, the yellow area indicates the period  
 305 of LGD. (c) Variations in the lake water concentrations of  $^{222}\text{Rn}$ . (d) Variations in the groundwater concentrations  
 306 of  $^{222}\text{Rn}$ . In (c) and (d), the top and bottom of each box represent the 75th and 25th percentiles of the data,  
 307 respectively; the ends of the whiskers indicate the maximum and minimum values within 1.5 times the  
 308 interquartile range; the small white square inside the box denotes the mean; the side curves illustrate the data's  
 309 dispersion and distribution pattern; the dark blue dashed line connects the mean values across different periods. (e)  
 310 Variations in the ratio of  $^{222}\text{Rn}$  concentrations in lake water to groundwater.  
 311  
 312

### 3.1.2 Quantification of LGD rates for each period

314 Based on the lake level and groundwater monitoring data, the period from late August 2022 to late  
 315 April 2023 was characterized by consistently higher groundwater levels than the lake stage, during  
 316 which the lake was hydrologically isolated from external surface water inputs. This interval was  
 317 conservatively defined as the active phase of LGD (Figure 2b). Using the  $^{222}\text{Rn}$  mass balance model,  
 318 LGD rates were computed for each period of groundwater discharge during groundwater excretion.

319 Table 2 summarizes the detailed parameters and results for the source and sink terms across each  
 320 sampling period. **A comparison of the  $^{222}\text{Rn}$  fluxes associated with the source and sink terms in the**  
 321  **$^{222}\text{Rn}$  mass balance model for each period is shown in Figure S2.** In terms of source, the  
 322 groundwater-derived  $^{222}\text{Rn}$  input flux ranged from  $90.48 \pm 36.32$  to  $165.02 \pm 63.09$   $\text{Bq}\cdot\text{m}^{-2}\cdot\text{d}^{-1}$ , while  
 323 the diffusive flux of  $^{222}\text{Rn}$  from sediment pore water varied between  $18.82 \pm 2.88$  and  $21.51 \pm 3.29$   
 324  $\text{Bq}\cdot\text{m}^{-2}\cdot\text{d}^{-1}$ . In terms of sink, the radioactive decay flux of  $^{222}\text{Rn}$  in lake water ranged from  $73.47 \pm$   
 325  $26.74$  to  $163.49 \pm 59.51$   $\text{Bq}\cdot\text{m}^{-2}\cdot\text{d}^{-1}$ , and atmospheric evasion accounted for a flux of  $20.34 \pm 6.46$  to  
 326  $38.51 \pm 12.87$   $\text{Bq}\cdot\text{m}^{-2}\cdot\text{d}^{-1}$ .

327 The LGD rates for each period were calculated by dividing the flux of  $^{222}\text{Rn}$  input from LGD by  
 328 the corresponding groundwater  $^{222}\text{Rn}$  concentration (Figure 3a). From August 2022 to April 2023, the  
 329 LGD rates ranged from  $35.36 \pm 16.39$  to  $51.71 \pm 23.23$   $\text{mm}\cdot\text{d}^{-1}$ , with the maximum value being 1.46  
 330 times the minimum value. Notably, an increasing trend in groundwater excretion rates was observed  
 331 from August 2022 to December 2022, which was followed by a decreasing trend from December 2022  
 332 to April 2023. The highest rate of groundwater excretion occurred in December 2022, whereas the  
 333 lowest LGD rate was recorded in April 2023.

334 **Table 2.** The calculation parameters and results of the  $^{222}\text{Rn}$  mass balance model for each period.

	08/2022	10/2022	12/2022	02/2023	04/2023
Lake water $^{222}\text{Rn}$ concentration ( $\text{Bq}\cdot\text{m}^{-3}$ )	$170.33 \pm 88.71$	$170.66 \pm 89.10$	$231.61 \pm 96.53$	$207.45 \pm 98.16$	$106.83 \pm 63.85$
Air $^{222}\text{Rn}$ concentration ( $\text{Bq}\cdot\text{m}^{-3}$ )	16.20	16.20	16.20	16.20	16.20
Wind speed above 2 m lake surface ( $\text{m}\cdot\text{s}^{-1}$ )	$1.48 \pm 0.82$	$1.37 \pm 0.83$	$1.08 \pm 1.11$	$1.65 \pm 1.44$	$2.05 \pm 1.57$
K ( $\text{m}\cdot\text{d}^{-1}$ )	$0.30 \pm 0.02$	$0.17 \pm 0.01$	$0.08 \pm 0.01$	$0.16 \pm 0.01$	$0.34 \pm 0.01$
Sc	554.34	937.71	1572.2	1492.73	871.45
Lake water temperature ( $^{\circ}\text{C}$ )	29.19	18.43	8.86	9.78	19.89
Lake water depth (m)	4.9	4.2	3.9	3.8	3.8
Groundwater $^{222}\text{Rn}$ concentration ( $\text{Bq}\cdot\text{m}^{-3}$ )	$3910.40 \pm 3511.38$	$3061.15 \pm 2099.00$	$3028.17 \pm 1412.72$	$3829.64 \pm 1316.67$	$2588.30 \pm 998.11$
The $^{222}\text{Rn}$ flux of atmospheric escape	$49.80 \pm 18.20$	$30.96 \pm 10.27$	$20.34 \pm 6.46$	$37.12 \pm 11.56$	$38.51 \pm 12.87$

	(Bq·m <sup>-2</sup> ·d <sup>-1</sup> )				
The <sup>222</sup> Rn flux of pore in sediment diffusion (Bq·m <sup>-2</sup> ·d <sup>-1</sup> )	24.70±3.78	21.10 ± 3.23	18.82±2.88	18.74±2.87	21.51±3.29
The <sup>222</sup> Rn flux of decay (Bq·m <sup>-2</sup> ·d <sup>-1</sup> )	151.07± 54.99	129.74±47.22	163.49±59.51	142.68±51.93	73.47±26.74
The <sup>222</sup> Rn flux of groundwater discharge (Bq·m <sup>-2</sup> ·d <sup>-1</sup> )	176.16± 58.05	139.59±48.44	165.02± 63.09	161.06±60.63	90.48±36.32
LGDrate (mm·d <sup>-1</sup> )	45.05± 18.39	45.60± 34.20	51.71±23.23	42.06±18.60	35.36±16.39

335

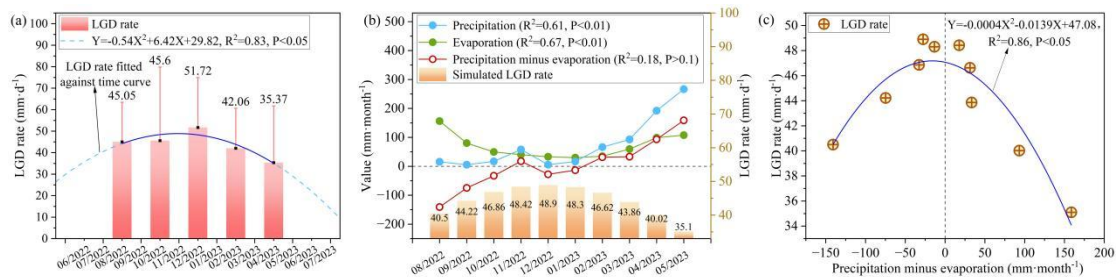
### 336 3.2 Climatic controls on seasonal variations in LGD rates

#### 337 3.2.1 Relationship between precipitation and evaporation and LGD rates

338 To obtain a higher temporal resolution within the hydrological year, fitting equations were  
 339 established to simulate the monthly LGD rates during groundwater discharge periods. The quadratic  
 340 equation representing LGD rates based on the <sup>222</sup>Rn mass balance model and time was  $Y = -0.54X^2 +$   
 341  $6.42X + 29.82$ ,  $R^2 = 0.83$ ,  $P < 0.05$  (where  $X$  represents the time and  $Y$  represents the LGD rate)  
 342 (Figure 3a). **Due to the small sample size, the robustness of the model's fit may be limited. Nonetheless,**  
 343 **under the field observed conditions, the monthly fluctuations of lake and groundwater levels were**  
 344 **relatively small and exhibited regular patterns, so the actual monthly variation of LGD showed a**  
 345 **predictable “increase followed by decrease” pattern. Therefore, the predictions of this model are**  
 346 **considered reliable.** Using this equation, the LGD rates were estimated for the period from August 2022  
 347 to May 2023. Although the water level at the Jianli station of the Yangtze River exceeded 26.5 m in  
 348 May 2023 for just 10 d, the inflow from the Yangtze River to the **HW lake** was minimal and  
 349 insufficient to raise the lake level above the groundwater level. Therefore, groundwater discharge into  
 350 the lake continued in May 2023. The simulated LGD rates ranged from 35.10 to 48.90 mm·d<sup>-1</sup> between  
 351 August 2022 and May 2023 (Figure 3b), with an approximately 8% error compared to the rates derived  
 352 from the <sup>222</sup>Rn mass balance model. These rates exhibited an increasing trend followed by a decreasing  
 353 trend, peaking in the middle of the groundwater discharge period, and tapering at both the beginning  
 354 and end stages.

355 During the groundwater discharge period, the changes in precipitation and evaporation were  
 356 inversely related to the changes in LGD, demonstrating a clear negative correlation with LGD rates,  
 357 with correlation coefficients of  $R^2=0.61$ ,  $P < 0.01$  and  $R^2=0.67$ ,  $P < 0.01$ , respectively (Figure 3b,  
 358 Figure S3 a and b). This suggested that lower monthly precipitation and evaporation contributed to the  
 359 larger LGD. This aligns with Jiang et al.'s (2024) results showing an inverse relationship between LGD

360 rates and precipitation/evaporation. Additionally, the value of precipitation minus evaporation (PME)  
 361 played a significant role in the hydrological cycle. By analyzing the relationship between the PME and  
 362 LGD rates, a non-linear relationship was identified, with the fitted equation being  
 363  $Y=-0.0004X^2-0.0139X+47.08$ ,  $R^2=0.86$ ,  $P<0.05$  (Figure 3c). When PME was greater than 0, a larger  
 364 PME resulted in a smaller LGD. When PME was less than 0, a smaller PME led to a smaller LGD.  
 365 When both precipitation and evaporation were low and the PME approached 0, LGD reached its  
 366 maximum value.



367  
 368 **Figure 3.** (a) The LGD rate of different periods and their fitted relationship between LGD rate and time. (b)  
 369 Relationship between simulated LGD rate and precipitation and evaporation, where  $R^2$  represents the linear  
 370 correlation coefficient between this indicator and the LGD rate. (c) Relationship between PME and LGD rate.

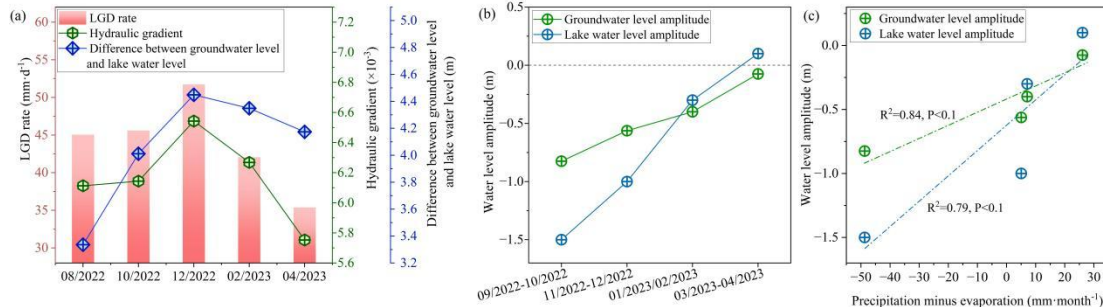
371

### 372 3.2.2 Dominance of precipitation and evaporation on seasonal variations in LGD rates

373 At different periods, a strong correlation was observed between the LGD rate and hydraulic  
 374 gradient ( $R^2=0.77$ ,  $P<0.05$ ) (Figure 4a), with both showing a consistent trend of change, in accordance  
 375 with Darcy's law (Tecklenburg and Blume; 2017) (Figure 4a). The hydraulic gradient is controlled by  
 376 the water level difference between groundwater and lake water—when this difference increases, the  
 377 LGD rate rises; when it decreases, the LGD rate declines. From August to December 2022, the water  
 378 level difference increased, leading to a rising LGD rate; from December 2022 to April 2023, the  
 379 difference decreased, resulting in a lower LGD rate (Figure 4a). This variation is determined by the  
 380 differences in groundwater level and lake water level. If the lake level drops more than the groundwater  
 381 level, the gradient increases and LGD intensifies; if the groundwater level drops more, the gradient and  
 382 LGD both decline (Figure 4b).

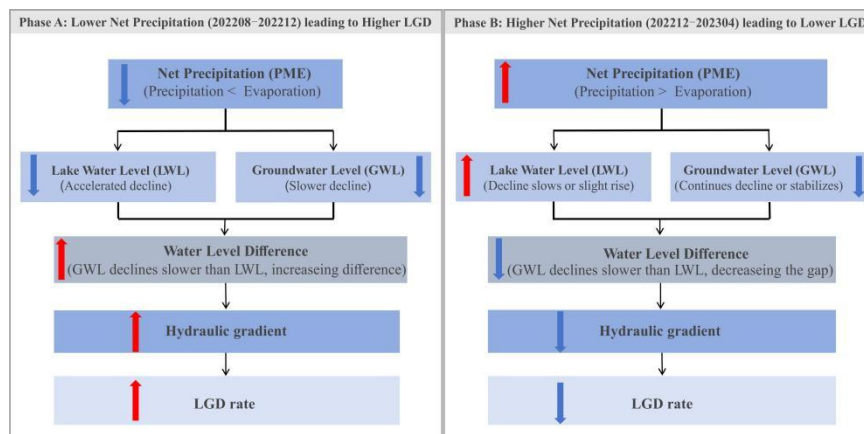
383 Monitoring data show that lake levels consistently declined during the study period, though the  
 384 rate of decline slowed, primarily due to low precipitation, reduced evaporation, cooler temperatures,  
 385 and decreased water consumption by agriculture and aquatic organisms. After February 2023,  
 386 increasing precipitation slightly raised lake levels. Groundwater levels also declined steadily, mainly  
 387 due to continuous LGD to lake, with minimal influence from human extraction or evaporation.

388 Between August and December 2022, groundwater levels fell more slowly than lake levels, increasing  
 389 the hydraulic gradient and LGD. From February 2023 onward, the groundwater level declined more  
 390 than the lake level, reducing the gradient and LGD rate. By April 2023, groundwater levels stabilized  
 391 while lake levels rose slightly, further decreasing the hydraulic gradient and LGD.



392 **Figure 4.** (a) Temporal characteristics of LGD rate, hydraulic gradient, and difference in groundwater and lake  
 393 water levels. (b) Temporal characteristics of groundwater and lake water level change amplitudes, negative values  
 394 indicate the amplitudes of water level decline, while positive values represent the amplitudes of water level rise. (c)  
 395 Relationship between groundwater level amplitude, lake water level amplitude, and PME.  
 396  
 397

398 Net precipitation (PME) directly affects lake level fluctuations and indirectly influences  
 399 groundwater levels by altering the water level difference (Figure 4c). A higher PME slows or reverses  
 400 lake level decline, reducing groundwater discharge and limiting groundwater level drop. A lower PME  
 401 accelerates lake level decline, enhancing groundwater discharge and deepening the groundwater level  
 402 drop. Therefore, PME modulated the lake level changes, which subsequently influenced the  
 403 groundwater levels, altered the lake-groundwater level difference, and regulated the hydraulic gradient,  
 404 ultimately controlling the seasonal variations in LGD rates. **A conceptual model of this mechanism is**  
 405 **shown in Figure 5.**



406 **Figure 5.** Conceptual model illustrating the mechanisms by which precipitation and evaporation influence LGD  
 407 rates. Red upward arrows and blue downward arrows indicate increasing and decreasing trends of the  
 408 corresponding parameters, respectively.  
 409  
 410

### 411 3.3 Impact of LGD-carried N and P on seasonal nutrient status of lakes

#### 412 3.3.1 Seasonal variations in TN and TP loads carried by LGD

413 Between August 2022 and April 2023, the concentration of TN in the groundwater exhibited  
414 notable temporal variability, ranging from 0.24 to 0.46 mmol·L<sup>-1</sup>. The lowest concentration was  
415 observed in August 2022, whereas the peak was observed in December 2022 (Table 3). In comparison,  
416 the range of temporal variations in TP concentration in groundwater was wider, spanning from 6.46 ×  
417 10<sup>-3</sup> to 2.19 × 10<sup>-2</sup> mmol·L<sup>-1</sup> (Table 3). The lowest TP concentration was recorded in December 2022,  
418 whereas the highest occurred in February 2023. Based on Eh, DO, and EC results (Table S3), TP  
419 concentration fluctuations likely result from the combined effects of internal adsorption and fixation,  
420 external input pulses, and geochemical competition under the background of an oxidation-enhanced  
421 environment driven by declining water levels (Eh and DO continuously increasing). For a detailed  
422 analysis, see the SI.

423 **Table 3.** Concentrations of TN and TP in groundwater at various periods and loads of TN and TP carried by LGD.

	TN concentration in groundwater (mmol·L <sup>-1</sup> )	TP concentration in groundwater (mmol·L <sup>-1</sup> )	LGD-TN (mmol·m <sup>-2</sup> ·d <sup>-1</sup> )	LGD-TP (mmol·m <sup>-2</sup> ·d <sup>-1</sup> )
08/2022	0.24	1.69×10 <sup>-2</sup>	10.76	0.76
10/2022	0.33	2.00×10 <sup>-2</sup>	15.22	0.91
12/2022	0.46	6.46×10 <sup>-3</sup>	24.00	0.33
02/2023	0.34	2.19×10 <sup>-2</sup>	14.25	0.92
04/2023	0.39	8.24×10 <sup>-3</sup>	13.93	0.29

424  
425 Multiplying the LGD rate by the concentrations of TN and TP in the groundwater demonstrated  
426 the nutrient load carried by LGD. The variations in nutrient loads from August 2022 to April 2023 are  
427 shown in Figures 6d and 6e. The loads of TN carried by LGD (LGD-TN) ranged from 10.76 to 24.00  
428 mmol·m<sup>-2</sup>·d<sup>-1</sup>, with the lowest load occurring in August 2022 and the highest in December 2022.  
429 Similarly, the loads of TP carried by LGD (LGD-TP) ranged from 0.29 to 0.92 mmol·m<sup>-2</sup>·d<sup>-1</sup>, with the  
430 highest load observed in December 2022 and the lowest in April 2023. The temporal variation in  
431 LGD-TN loads followed the trend of the LGD rate and was primarily influenced by the LGD rate. In  
432 contrast, the variation in LGD-TP loads over time did not align with the LGD rate and was mainly  
433 controlled by the concentration of TP in the groundwater.

434

#### 435 3.3.2 Seasonal relationship between LGD-carried TN/TP loads and nutrient status of lakes

436 The nutrient status of lakes is determined by various parameters, and the concentration of Chl *a* in  
437 lake water is considered particularly important. In this study, no significant correlation was observed

438 between Chl *a* concentration and the concentrations of TN and TP (Figures 6a and 6b). However, a  
439 correlation was identified with the TN to TP molar ratio (TN/TP) (Figure 6c). As the TN/TP ratio  
440 approached the Redfield ratio of 16 (Figure 6c), the Chl *a* concentration increased (AC, 1960), whereas  
441 a deviation from this ratio was associated with a decrease in Chl *a* concentration, which was consistent  
442 with the findings reported in other studies (Qin et al., 2020).

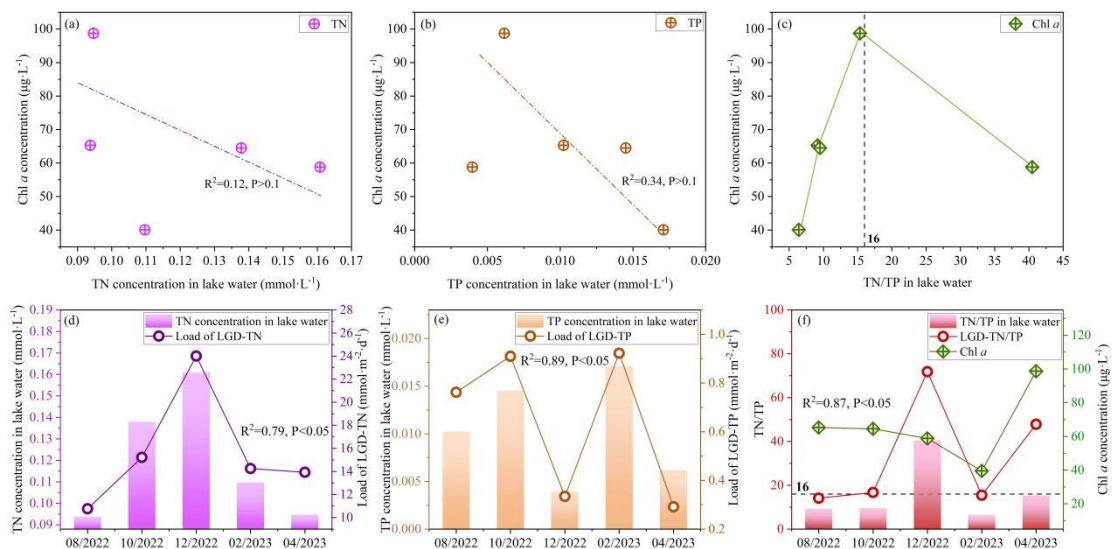
443 In the absence of surface water inflow and pollutant discharge from prohibited human activities, N  
444 and P enter the lake primarily through groundwater discharge, atmospheric deposition, and sediment  
445 release. To quantify TN and TP inputs from atmospheric deposition and sediment release,  
446 comprehensive field monitoring and experimental investigations were conducted during **HW lake**  
447 periods (Tables S1 and S2). The results indicated that the TN and TP loads from atmospheric deposition  
448 were 0.09 and  $9.68 \times 10^{-4}$  mmol·m<sup>-2</sup>·d<sup>-1</sup>, respectively. Simultaneously, the diffusive TN and TP loads  
449 from lakebed sediment were 0.08 and  $1.29 \times 10^{-4}$  mmol·m<sup>-2</sup>·d<sup>-1</sup>, respectively. In contrast, the LGD-TN  
450 and LGD-TP loads were 1–2 orders of magnitude higher than those from atmospheric deposition and  
451 sediment release, indicating that LGD-TN and LGD-TP constituted the primary sources of TN and TP  
452 in the lake. A strong correlation was observed between LGD-TN loads and TN concentrations in lake  
453 water ( $R^2=0.79$ ,  $P<0.05$ ) (Figure 6d), whereas the LGD-TP loads exhibited entirely consistent  
454 variations with TP concentrations in lake water ( $R^2=0.89$ ,  $P<0.05$ ) (Figure 6e), highlighting their  
455 significant influence.

456 The molar ratio of TN to TP in LGD (LGD-TN/TP) exhibited variations that were entirely  
457 consistent with the TN/TP ratio in the lake water, indicating a strong correlation ( $R^2=0.82$ ,  $P<0.05$ )  
458 (Figure 6f), thereby indicating its regulatory effect on the lake water TN/TP ratio. Based on the  
459 observed changes in LGD-TN/TP and their reflection in lake water TN/TP, when the lake water TN/TP  
460 reached 15.35 in April 2023, the Chl *a* concentration was elevated. These results demonstrated that the  
461 LGD-derived TN and TP substantially influenced the lake's nutrient status by modulating the TN/TP  
462 ratio, thereby affecting the Chl *a* concentration and the overall nutrient balance.

463 In addition, light availability and water temperature can also affect Chl *a* growth. Although higher  
464 temperatures and favorable light conditions in summer and autumn are theoretically conducive to  
465 phytoplankton proliferation, Figure. 6f shows that Chl *a* concentrations decreased continuously from  
466 August 2022 to February 2023, but increased rapidly and reached a peak in April 2023. This increase  
467 may be partly related to improvements in light and temperature conditions. However, during the

468 summer wet season, when temperature and light are most favorable and LGD is absent, the mean Chl *a*  
 469 concentration was only ~60 µg/L, which is much lower than the 98.69 µg/L observed in April 2023.  
 470 These results suggest that light and temperature are not the dominant drivers of Chl *a* variability;  
 471 instead, nitrogen and phosphorus concentrations and their TN/TP are the key controlling factors.

472 In lake eutrophication management, Chl *a* accumulation is governed not only by the absolute  
 473 concentrations of nitrogen and phosphorus but also by the TN/TP, which determines the dominant type  
 474 of nutrient limitation. Low TN/TP indicate nitrogen limitation, whereas high TN/TP indicate  
 475 phosphorus limitation. Given that present study has shown that groundwater can significantly alter lake  
 476 TN/TP, its influence should be explicitly considered when identifying nutrient limitation mechanisms  
 477 and developing targeted nitrogen and phosphorus reduction strategies.



478 **Figure 6.** Relationship between LGD-TN and TP loads and lake water TN and TP concentrations, as well as the  
 479 lake water Chl *a* concentration. (a) Relationship between Chl *a* concentration and lake water TN concentration. (b)  
 480 Relationship between Chl *a* concentration and lake water TP concentration. (c) Relationship between Chl *a*  
 481 concentration and lake water TN/TP. (d) Relationship between lake water TN concentration and LGD-TN load. (e)  
 482 Relationship between lake water TP concentration and LGD-TP load. (f) Relationship between lake water TN/TP  
 483 and LGD-TN/TP load.  
 484  
 485

### 486 3.4 Implications

#### 487 3.4.1 Advantages of high-frequency observations in LGD research

488 Traditional low-frequency sampling (e.g., single or quarterly measurements) has difficulty  
 489 capturing short-term variations in LGD and its coupling with short-term meteorological conditions and  
 490 water-level fluctuations. High-frequency observations can capture LGD responses to environmental  
 491 factors such as meteorology and water temperature over relatively short timescales, and provide  
 492 higher-resolution hydrological time series, thereby supplying reliable data for accurate estimation of  
 493 LGD rates. In addition, high-frequency observations can record the short-term contributions of LGD to

494 lake N and P inputs, enabling more precise flux assessments, avoiding biases associated with  
495 low-frequency sampling, and revealing the response relationships between LGD and Chl *a*  
496 concentrations or algal bloom events. Based on high-frequency data, LGD predictive models that  
497 account for the coupling of meteorological and hydrological factors can also be developed. Therefore,  
498 in LGD research, high-frequency observations are of significant value for accurately quantifying the  
499 impacts of groundwater on lake aquatic ecosystems and should be given priority.

500

#### 501 **3.4.2 Dominant factors controlling the seasonal variability of LGD**

502 The seasonal variability of LGD is primarily governed by three categories of  
503 factors—meteorological, hydrological, and anthropogenic—with the dominant controls differing across  
504 lake types.

505 (1) In open lakes (e.g., Poyang Lake and Tonle Sap Lake), hydrological processes dominate LGD  
506 dynamics. Their hydraulic connection with external rivers (such as the Yangtze and Mekong Rivers)  
507 causes rapid lake-level rises during the wet season and pronounced declines during the dry season,  
508 thereby modifying the hydraulic gradients between the littoral zone and aquifer and directly regulating  
509 groundwater discharge rates (Burnett et al. 2017; Li et al. 2020). (2) In closed lakes, where external  
510 fluvial influence is minimal, LGD dynamics are primarily controlled by the balance between  
511 precipitation and evaporation. Previous studies (Shi et al., 2022; Gan et al., 2024) have shown that  
512 under relatively stable groundwater levels, strong evaporation lowers lake levels, steepens the  
513 hydraulic gradient, and enhances LGD, whereas precipitation-dominated periods raise lake levels and  
514 suppress LGD. Hence, the seasonal oscillation of LGD in closed-basin systems essentially reflects a  
515 periodic adjustment of lake–groundwater hydraulic connectivity driven by meteorological forcing. (3)  
516 Anthropogenic activities, such as artificial water regulation and groundwater abstraction, can further  
517 modify LGD by altering regional hydrological circulation and the hydraulic relationships between  
518 lakes and aquifers (Xiong et al., 2023).

519

#### 520 **3.4.3 Regulatory role of LGD in lake nutrient status**

521 In closed lakes lacking significant surface inflow, LGD not only constitutes a key component of  
522 the water balance but also plays a pivotal role in nutrient transport and water quality evolution. The  
523 nitrogen and phosphorus fluxes carried by LGD directly influence the spatial and temporal patterns of

524 nutrient concentrations and Chl *a* within the lake. Several studies support this mechanism. For instance,  
525 Xu et al. (2024) showed that in small semi-arid lakes, LGD inputs largely determine the spatial  
526 distribution of Chl *a*, while Shi et al. (2022) reported a delayed response of lake trophic conditions to  
527 LGD-derived nutrient inputs. Such lakes are typically characterized by strong enclosure and long water  
528 residence times, conditions that amplify the influence of groundwater in biogeochemical cycling and  
529 ecological processes.

530         Synthesizing current knowledge, the linkage from seasonal forcing to ecological response in  
531 closed lakes can be conceptualized as follows: meteorological conditions (precipitation–evaporation  
532 balance) serve as the initial driver, regulating lake–groundwater hydraulic gradients and thereby  
533 controlling the seasonal magnitude of LGD. LGD, in turn, acts as a key conduit for nutrient fluxes,  
534 with its temporal variability shaping nutrient and Chl *a* distributions and ultimately modulating nutrient  
535 dynamics.

536         Previous studies have long approached the issue of internal nutrient loading in lakes primarily  
537 from the perspective of nitrogen and phosphorus release from bottom sediments, emphasizing the role  
538 of sediment-derived nutrients in shaping lake trophic status (Donohue and Garcia Molinos, 2009). For  
539 lakes where LGD-derived nutrient loads are relatively small during specific seasons, sedimentary  
540 nutrient release can indeed constitute a major source to the overlying water column, consistent with  
541 traditional understanding (Xu et al., 2017). However, in systems where LGD inputs are substantial,  
542 focusing solely on sedimentary release may not fully capture the true structure of lake nutrient sources.  
543 Although systematic studies that simultaneously assess both sediment nutrient release and  
544 LGD-derived inputs remain limited, existing evidence indicates that LGD is likely a key mechanism  
545 sustaining nutrient cycling and ecological succession in closed-basin lakes, highlighting the need for  
546 more comprehensive and quantitative investigations.

547         HW lake and other study cases are all typical closed shallow lakes, with limited exchange between  
548 the lake water and surface inflows, making LGD the primary hydrological input. A large number of  
549 similar lakes exist worldwide (e.g., closed lakes in the North American Great Plains, inland lakes in  
550 Africa and Central Asia), which may also be significantly influenced by LGD. Moreover, the  
551 regulatory effect of LGD on the spatial distribution of nitrogen, phosphorus, Chl *a*, and algal growth is  
552 amplified in lakes with long water residence times and stable sediment–water exchange. Therefore, this  
553 mechanism can be extended to other closed shallow lakes, particularly those with pronounced seasonal

554 climates and a high proportion of shallow areas.

555

#### 556 **3.4.4 Future recommendations for the lake management of closed shallow lakes**

557 Based on the above understanding, future research should adopt a systems-based perspective,  
558 focusing on comparative analyses of LGD inputs versus internal sediment release and quantitatively  
559 identifying their relative importance across different lake types and temporal scales to clarify the  
560 primary nutrient sources. This approach can provide a scientific basis for designing targeted nitrogen  
561 and phosphorus reduction strategies, as well as effective lake management and ecological restoration  
562 measures, thereby improving water quality and ecological function in closed lakes.

563 Moreover, climate change is expected to modulate these processes. In the East Asian monsoon  
564 region, future precipitation is projected to become more concentrated in summer, while non-summer  
565 periods, particularly autumn and winter, may experience reduced rainfall and increased evaporation.  
566 Because LGD predominantly occurs during non-summer periods, increased net precipitation  
567 (precipitation minus evaporation) could enhance groundwater discharge, directly elevating associated  
568 TN and TP fluxes. These considerations further underscore the critical role of groundwater in  
569 regulating nutrient balances in closed lakes and highlight the need for future nutrient management  
570 strategies to account for both LGD variability and climate-driven hydrological changes.

571

## 572 **4 Conclusions**

573 High-frequency monitoring and hydrochemical methods were employed to investigate the  
574 dynamics of LGD and associated nutrient loads in Heiwawu Oxbow Lake from August 2022 to April  
575 2023, revealing the regulatory mechanisms of precipitation-evaporation processes. The LGD flux  
576 exhibited an increasing-then-decreasing trend during the study period, ranging from 35.36 to 51.71  
577  $\text{mm}\cdot\text{d}^{-1}$ , with a peak in December 2022 and a minimum in April 2023. The LGD rates were governed  
578 by the hydraulic gradient between groundwater and lake water, which responded nonlinearly to net  
579 monthly precipitation. Positive net precipitation induced rapid lake recharge, reduced the hydraulic  
580 gradient, and suppressed LGD rates. Conversely, negative net precipitation enhanced  
581 evaporation-driven lake discharge, steepened the hydraulic gradient, and increased LGD rates. When  
582 net precipitation approached zero, a critical hydraulic equilibrium was reached, yielding the highest  
583 LGD rates. The LGD was also a major nutrient pathway, with TN and TP loads significantly correlating

584 with lake concentrations. The TN/TP ratio of LGD aligned with lake water, influencing phytoplankton  
585 structure and Chl *a* levels. This study reveals the regulatory mechanisms by which LGD is controlled  
586 by the precipitation–evaporation balance at the monthly scale, quantitatively evaluates the impacts of  
587 groundwater-driven nutrient transport on lake eutrophication, and, by reference to related global case  
588 studies, provides environmental protection and management implications from a groundwater  
589 perspective. **These findings offer new insights into eutrophication control in closed, shallow lakes, and  
590 suggest that the advantages of high-frequency LGD observations should be incorporated into lake  
591 management frameworks, along with the development of targeted control strategies during critical  
592 periods of groundwater–lake interactions.**

593

#### 594 **Acknowledgments**

595 The research was funded by National Natural Science Foundation of China (Nos. 42422707,  
596 U21A2026), and Fundamental and Interdisciplinary Disciplines Breakthrough Plan of the Ministry of  
597 Education of China (No. JYB2025XDXM911). The authors would like to thank all the reviewers who  
598 participated in the review, as well as MJEditor ([www.mjeditor.com](http://www.mjeditor.com)) for providing English editing  
599 services during the preparation of this manuscript.

600

#### 601 **Declaration of interests**

602 The authors declare that they have no known competing financial interests or personal  
603 relationships.

604

#### 605 **Data Availability Statement**

606 Data will be made available on request.

607

#### 608 **Author contributions**

609 Writing-review & editing, Writing-original draft, Methodology, Investigation, Data curation: XS.  
610 Writing-review & editing, Resources, Methodology, Funding acquisition: YD. Investigation: HT, JX,  
611 HS, YL. Project administration: YD. Funding acquisition: YG. Resources, Project administration: YW.

612

#### 613 **References**

614 Redfield, A. C.: The biological control of chemical factors in the environment, *Sci. Prog.*, 11, 150–170,

615 1960.

616 Adyasari, D., Dimova, N. T., Dulai, H., Gilfedder, B. S., Cartwright, I., McKenzie, T., and Fuleky, P.:  
617 Radon-222 as a groundwater discharge tracer to surface waters, *Earth-Sci. Rev.*, 238, 104321,  
618 <https://doi.org/10.1016/j.earscirev.2023.104321>, 2023.

619 Arnoux, M., Gibert-Brunet, E., Barbecot, F., Guillon, S., Gibson, J., and Noret, A.: Interactions  
620 between groundwater and seasonally ice-covered lakes: using water stable isotopes and radon-222  
621 multilayer mass balance models, *Hydrol. Process.*, 31, 2566–2581,  
622 <https://doi.org/10.1002/hyp.11206>, 2017.

623 Burnett, W. C., Wattayakorn, G., Supcharoen, R., Sioudom, K., Kum, V., Chanyotha, S., and  
624 Kritsananuwat, R.: Groundwater discharge and phosphorus dynamics in a flood-pulse system:  
625 Tonle Sap Lake, Cambodia, *J. Hydrol.*, 549, 79–91, <https://doi.org/10.1016/j.jhydrol.2017.03.049>,  
626 2017.

627 Corbett, D.R., Burnett, W., Cable, P., Clark, S.: A multiple approach to the determination of radon fluxes  
628 from sediments. *Journal of Radioanalytical and Nuclear Chemistry*, 236(1–2), 247–253,  
629 <https://doi.org/10.1007/bf02386351>, 1998.

630 Dabrowski, J. S., Charette, M. A., Mann, P. J., Ludwig, S. M., Natali, S. M., Holmes, R. M., and  
631 Henderson, P. B.: Using radon to quantify groundwater discharge and methane fluxes to a shallow,  
632 tundra lake on the Yukon–Kuskokwim Delta, Alaska, *Biogeochemistry*, 148, 69–89,  
633 <https://doi.org/10.1007/s10533-020-00647-w>, 2020.

634 Dimova, N. T. and Burnett, W. C.: Evaluation of groundwater discharge into small lakes based on the  
635 temporal distribution of radon-222, *Limnol. Oceanogr.*, 56, 486–494,  
636 <https://doi.org/10.4319/lo.2011.56.2.0486>, 2011.

637 Donohue, I., and Garcia Molinos, J.: Impacts of increased sediment loads on the ecology of lakes.  
638 *Biological reviews*, 84(4), 517–531, <https://doi.org/10.1111/j.1469-185X.2009.00081.x>, 2009.

639 Gan, Y., Sun, X., Wu, J., Du, Y., Deng, Y., Han, P., and Wang, Y.: Spatio-temporal variations of  
640 lacustrine groundwater discharge and related nutrient fluxes in a typical lake in front of hillocks, *J.*  
641 *Hydrol.*, 635, 131166, <https://doi.org/10.1016/j.jhydrol.2024.131166>, 2024.

642 Gao, X., Jia, T., Xu, Q., Wang, F., and Wang, A.: Records of lacustrine sedimentology and  
643 pollen–charcoal assemblages responding to climate change and human activities in Zhongzhouzi  
644 Oxbow Lake, Hubei Province for about 70 years, *Quaternary Sci.*, 36, 1445–1455,

645 <https://doi.org/10.11928/j.issn.1001-7410.2016.06.10>, 2016.

646 **Gonnea, M. E., Morris, P. J., Dulaiova, H., and Charette, M. A.:** **New perspectives on radium behavior**  
647 **within a subterranean estuary, *Mar. Chem.*, 109, 250–267,**  
648 **<https://doi.org/10.1016/j.marchem.2007.12.002>, 2008.**

649 Hu, M., Zhou, P., and Chen, C.: Study on coupling of typical elements in surface water and  
650 groundwater in the middle reaches of the Yangtze River, China, *J. Hydrol.*, 623, 130298,  
651 <https://doi.org/10.1016/j.jhydrol.2023.130298>, 2023.

652 Jia, T., Wang, F., and Yuan, S.: Sedimentation and its environmental significance of the Niuyu Lake  
653 along the middle reaches of the Yangtze River: a case study of Tian'e Island and Zhongzhouzi in  
654 the Jingjiang section of the Yangtze River, *Geogr. Res.*, 34, 861–871, 2015. (in Chinese)

655 Jiang, C., Jiang, C., Wang, Q., Liu, H., Li, D., Zhu, Q., and Liu, F.: Seasonal characteristics of  
656 groundwater discharge controlled by precipitation and its environmental effects in a coal mining  
657 subsidence lake, eastern China, *Sci. Total Environ.*, 915, 170067,  
658 <https://doi.org/10.1016/j.scitotenv.2024.170067>, 2024.

659 Jiang, X., Ma, R., Ma, T., and Sun, Z.: Modeling the effects of water diversion projects on surface  
660 water and groundwater interactions in the central Yangtze River Basin, *Sci. Total Environ.*, 830,  
661 154606, <https://doi.org/10.1016/j.scitotenv.2022.154606>, 2022.

662 Kazmierczak, J., Postma, D., Muller, S., Jessen, S., Nilsson, B., Czekaj, J., and Engesgaard, P.:  
663 Groundwater-controlled phosphorus release and transport from sandy aquifer into lake, *Limnol.*  
664 *Oceanogr.*, 65, 1447–1463, <https://doi.org/10.1002/lno.11447>, 2020.

665 Kluge, T., Ilmberger, J., von Rohden, C., and Aeschbach-Hertig, W.: Tracing and quantifying  
666 groundwater inflow into lakes using a simple method for radon-222 analysis, *Hydrol. Earth Syst.*  
667 *Sci.*, 11, 1621–1631, <https://doi.org/10.5194/hess-11-1621-2007>, 2007.

668 Lewandowski, J., Rosenberry, D. O., and Meinikmann, K.: Groundwater–lake interfaces, in:  
669 *Ecohydrological Interfaces*, edited by: Krause, S., Hannah, D. M., and Fleckenstein, J. H., Wiley,  
670 Hoboken, NJ, 103–122, <https://doi.org/10.1002/9781119639308.ch6>, 2024.

671 Luo, X., Kuang, X. X., Jiao, J. J., Liang, S. H., Mao, R., Zhang, X. L., and Li, H. L.: Evaluation of  
672 lacustrine groundwater discharge, hydrologic partitioning, and nutrient budgets in a proglacial  
673 lake in the Qinghai–Tibet Plateau using <sup>222</sup>Rn and stable isotopes, *Hydrol. Earth Syst. Sci.*, 22,  
674 5579–5598, <https://doi.org/10.5194/hess-22-5579-2018>, 2018.

675 MacIntyre, S., Wanninkof, R., & Chanton, J.P.: Trace gas exchange across the air-water interface in  
676 freshwater and coastal marine environments, in, edited by: Matson, PA and Harriss, RC, *Biogenic*  
677 *trace gases: Measuring emissions from soil and water*, *Methods Ecol. Evol.*, 52–97, 1995.

678 Meinikmann, K., Hupfer, M., and Lewandowski, J.: Phosphorus in groundwater discharge – a potential  
679 source for lake eutrophication, *J. Hydrol.*, 524, 214–226,  
680 <https://doi.org/10.1016/j.jhydrol.2015.02.031>, 2015.

681 Olid, C., Rodellas, V., Rocher-Ros, G., Garcia-Orellana, J., Diego-Feliu, M., Alorda-Kleinglass, A.,  
682 Bastviken, D., and Karlsson, J.: Groundwater discharge as a driver of methane emissions from  
683 Arctic lakes, *Nat. Commun.*, 13, 3667, <https://doi.org/10.1038/s41467-022-31219-1>, 2022.

684 Petermann, E., Gibson, J. J., Knöller, K., Pannier, T., Weiß, H., and Schubert, M.: Determination of  
685 groundwater discharge rates and water residence time of groundwater-fed lakes by stable isotopes  
686 of water ( $^{18}\text{O}$ ,  $^2\text{H}$ ) and radon ( $^{222}\text{Rn}$ ) mass balances, *Hydrol. Process.*, 32, 805–816,  
687 <https://doi.org/10.1002/hyp.11401>, 2018.

688 Qin, B., Zhou, J., Elser, J. J., Gardner, W. S., Deng, J., and Brookes, J. D.: Water depth underpins the  
689 relative roles and fates of nitrogen and phosphorus in lakes, *Environ. Sci. Technol.*, 54, 3191–3198,  
690 <https://doi.org/10.1021/acs.est.9b05835>, 2020.

691 Rosenberry, D. O., Lewandowski, J., Meinikmann, K., and Nützmann, G.: Groundwater – the  
692 disregarded component in lake water and nutrient budgets. Part 1: effects of groundwater on  
693 hydrology, *Hydrol. Process.*, 29, 2895–2921, <https://doi.org/10.1002/hyp.10403>, 2015.

694 Schmidt, A., Gibson, J. J., Santos, I. R., Schubert, M., Tattree, K., and Weiss, H.: The contribution of  
695 groundwater discharge to the overall water budget of two typical boreal lakes in Alberta, Canada  
696 estimated from a radon mass balance, *Hydrol. Earth Syst. Sci.*, 14, 79–89,  
697 <https://doi.org/10.5194/hess-14-79-2010>, 2010.

698 Shi, X., Luo, X., Jiao, J. J., and Zuo, J.: Dominance of evaporation on lacustrine groundwater discharge  
699 to regulate lake nutrient state and algal blooms, *Water Res.*, 219, 118620,  
700 <https://doi.org/10.1016/j.watres.2022.118620>, 2022.

701 Stets, E. G., Winter, T. C., Rosenberry, D. O., and Striegl, R. G.: Quantification of surface water and  
702 groundwater flows to open- and closed-basin lakes in a headwaters watershed using a descriptive  
703 oxygen stable isotope model, *Water Resour. Res.*, 46, W03504,  
704 <https://doi.org/10.1029/2009WR007793>, 2010.

705 Sun, X., Du, Y., Deng, Y., Fan, H., Tao, Y., and Ma, T.: Contribution of groundwater discharge and  
706 associated contaminants input to Dongting Lake, Central China, using multiple tracers ( $^{222}\text{Rn}$ ,  
707  $^{18}\text{O}$ ,  $\text{Cl}^-$ ), *Environ. Geochem. Health*, 43, 1239–1255,  
708 <https://doi.org/10.1007/s10653-020-00647-3>, 2021.

709 Sun, X., Du, Y., Wu, J., Xu, J., Tian, H., Deng, Y., and Wang, Y.: Two-decadal variability of lacustrine  
710 groundwater discharge: coupled controls from weather and hydrologic changes, *Water Resour.*  
711 *Res.*, 60, e2024WR037173, <https://doi.org/10.1029/2024WR037173>, 2024.

712 Sun, X., Du, Y., Wu, J., Xu, J., Tian, H., Han, P., and Wang, Y.: Spatial variability of lacustrine  
713 groundwater discharge at basin scale, *J. Hydrol.*, 629, 134404,  
714 <https://doi.org/10.1016/j.jhydrol.2024.134404>, 2025a.

715 Sun, X., Du, Y., Xu, J., Tian, H., Deng, Y., Gan, Y., and Wang, Y.: Control of groundwater–lake  
716 interaction zone structure on spatial variability of lacustrine groundwater discharge in oxbow lake,  
717 *Water Resour. Res.*, 61, e2024WR039334, <https://doi.org/10.1029/2024WR039334>, 2025b.

718 Tecklenburg, C. and Blume, T.: Identifying, characterizing and predicting spatial patterns of lacustrine  
719 groundwater discharge, *Hydrol. Earth Syst. Sci.*, 21, 5043–5063,  
720 <https://doi.org/10.5194/hess-21-5043-2017>, 2017.

721 Walvoord, M. A. and Striegl, R. G.: Increased groundwater to stream discharge from permafrost  
722 thawing in the Yukon River basin: potential impacts on lateral export of carbon and nitrogen,  
723 *Geophys. Res. Lett.*, 34, L12402, <https://doi.org/10.1029/2007GL030216>, 2007.

724 Wang, Q., Li, H., Zhang, Y., Wang, X., Zhang, C., Xiao, K., and Qu, W.: Evaluations of submarine  
725 groundwater discharge and associated heavy metal fluxes in Bohai Bay, China, *Sci. Total Environ.*,  
726 695, 133873, <https://doi.org/10.1016/j.scitotenv.2019.133873>, 2019.

727 Webster, I. T., Hancock, G. J., and Murray, A. S.: Modelling the effect of salinity on radium desorption  
728 from sediments, *Geochim. Cosmochim. Acta*, 59, 2469–2476,  
729 [https://doi.org/10.1016/0016-7037\(95\)00111-7](https://doi.org/10.1016/0016-7037(95)00111-7), 1995.

730 Xiong, L., Aldahan, A., Qian, R., Yi, P., Chen, X., Li, K., and He, P.: Spatio-temporal patterns and  
731 quantification of lake–groundwater interaction determined in a large water transfer lake, *Hydrol.*  
732 *Process.*, 37, e14867, <https://doi.org/10.1002/hyp.14867>, 2023.

733 Xu, R., Du, Y., Wang, Z., Sun, X., Yang, L., Liu, J., and Gan, Y.: Contrasting lacustrine groundwater  
734 discharge in two small perennial lakes around dried-up Chahannaer Lake, Northern China, *J.*

735 Hydrol. Reg. Stud., 58, 102280, <https://doi.org/10.1016/j.ejrh.2025.102280>, 2025.

736 Xu, Y., Wu, Y., Han, J., and Li, P.: The current status of heavy metal in lake sediments from China:  
737 Pollution and ecological risk assessment. *Ecology and evolution*, 7(14), 5454-5466.  
738 <https://doi.org/10.1002/ece3.3124>, 2017.

739 Xue, P., Wen, Z., Zhao, D., Jakada, H., and Liang, X.: Determination of hydraulic conductivity and its  
740 spatial variability in the Jiangnan Plain using a multi-format, multi-method approach, *J. Hydrol.*,  
741 594, 125917, <https://doi.org/10.1016/j.jhydrol.2021.125917>, 2021.

742 Zedler, J. B. and Kercher, S.: Wetland resources: status, trends, ecosystem services, and restorability,  
743 *Annu. Rev. Environ. Resour.*, 30, 39–74,  
744 <https://doi.org/10.1146/annurev.energy.30.050504.144248>, 2005.

745 Zheng, J., Chen, K., Wu, J., and Wu, J.: Lacustrine groundwater discharge as an important hidden  
746 source of nutrients to a large eutrophic lake: implications for eutrophication management, *Sci.*  
747 *Total Environ.*, 960, 178313, <https://doi.org/10.1016/j.scitotenv.2024.178313>, 2025.

748