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Diagnosing habitat degradation of submerged plants in urban lakes: An integrated analysis of underwater light limitation, lake optics, and DOM chemo-diversity

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ABSTRACT

Urban lakes are vital socio-economic and ecological assets, yet their aesthetically pleasant clear water states often degrade along with losses of submerged plants. Restoration failures are frequently due to ambiguous linkages between stressors and habitat deterioration. This study developed an integrated diagnostic framework, combining underwater light limitation modelling, lake optic analysis, and dissolved organic matter (DOM) chemo-diversity. The extent of habitat degradation is quantified by the probabilities of a random site that satisfies the minimal light requirement of submerged macrophytes in a lake. Meanwhile, causes of degradation are identified by lake optic analysis and highly correlated DOM-derived indexes that reflect footprints of primary pollution sources. Our findings reveal severe underwater light limitation in four investigated urban lakes in Jiujiang (China), which were driven mainly by elevated chlorophyll-a levels. Domestic sewage was identified as the predominant pollution source, with minimal agricultural and industrial impacts. By establishing a clear stressor-habitat linkage, the results emphasise reducing sewage inputs as a priority for effective lake restoration. This study offers a multi-disciplinary diagnostic framework for managing submerged plant habitat loss in urban lakes.

1. Introduction

Urban lakes are typically shallow water bodies within or adjacent to densely populated city areas and are invaluable socio-economic and ecological assets. They serve as recreational hubs that enhance community well-being (Walker et al., 2013) and provide essential ecological functions such as climate regulation, flood mitigation, and urban landscaping (Peng et al., 2004; Yu et al., 2021). Urban lakes, in general, are shallower and more eutrophic than natural lakes (Costadone and Sytsma, 2022). Rapid urbanisation and economic growth make urban lakes increasingly vulnerable to pollution from domestic wastewaters, industrial effluence, and non-point agricultural sources (Wang et al., 2023; Zheng et al., 2022) that degrade their ecosystem health (Costadone and Sytsma, 2022). Among the many signs of ecological decline in urban lakes, the widespread disappearance of submerged macrophytes is one of the most visible and ecologically consequential (Hilt et al., 2018; Howard-Williams et al., 2018). Submerged macrophyte beds are crucial to the structure and stability of shallow lake ecosystems (Coops et al., 2007; Grzybowski et al., 2023; Poikane et al., 2018; Schallenberg et al., 2018). Submerged plant beds in urban lakes reduce sediment resuspension, absorb nutrients, and create an aesthetically pleasant clear-water state (Liu et al., 2018; Liu et al., 2020a). Their losses mark a regime shift from clear to turbid states, with phytoplankton dominance and unfavourable poor water clarity (Abell et al.,

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2022; Scheffer et al., 1993).

Despite growing efforts in lake restoration, successful recoveries of stable submerged plant beds in urban lakes remain limited (Abell et al., 2022; Poikane et al., 2024). A major barrier is the lack of an integrative diagnostic framework that links environmental stressors to habitat degradation (Abell et al., 2022; Howard-Williams et al., 2018; Poikane et al., 2024; Sondergaard et al., 2007). Most existing studies assessed habitat suitability for submerged plants (Bakker et al., 2013; Valle et al., 2011; Van der Lee et al., 2006; Xu et al., 2020) or tracked lake pollution sources (Heredia et al., 2022; Romanelli et al., 2020; Xue et al., 2009; Zhou et al., 2022). However, they rarely establish a clear mechanistic linkage between a specific stressor and its impacts on habitat variables that sustain macrophyte communities. This challenges the identification of specific primary stressors and designing effective remediation strategies (Moss, 2007; Poikane et al., 2024; Velázquez-Ochoa and Enríquez, 2023). Diagnosing habitat degradation in freshwater systems increasingly requires integrating multiple lines of evidence across physical, chemical, and biological dimensions. Recent reviews have advocated for comprehensive, integrative diagnostic frameworks that combine emerging technologies and diverse knowledge streams to address multiple stressor impacts on freshwater fishes (Thomson-Laing et al., 2024). Cause-effect pathways have been established for freshwater fishes between critical habitat variables and hydrodynamic conditions (Shen et al., 2018; Yang et al., 2021). However, an equivalent comprehensive diagnostic framework for submerged macrophyte habitats in urban lakes remains lacking.

In this study, we are among the first to propose a framework that integrates the modelling of underwater light limitation, lake optical analysis, and high-resolution DOM chemo-diversity characterisation to trace the causal pathway from pollution source to habitat degradation (Fig. 1). Underwater irradiance is the single most important factor regulating the distribution and growth of submerged macrophytes in shallow lakes in temperate and subtropical regions (Bornette and Puijalon, 2011; Chambers and Kalff, 1985; Dong et al., 2021; Ersoy et al., 2020; Hilt et al., 2006; Hu et al., 2019; Liu et al., 2016; Middelboe and Markager, 1997; Xu et al., 2020), in addition to sediment properties that influence anchorage and respiration of rooting systems (Bornette and Puijalon, 2011; Liu et al., 2017). Numerous studies have employed underwater light limitation to estimate habitat suitability for and guide the transplantation of submerged plants in shallow lakes (Hilt et al., 2006; Hilt et al., 2018; Hu et al., 2019; Xu et al., 2020). Then, underwater light limitation is further investigated by lake optics that quantifies the relative contributions to underwater light attenuation of phytoplankton, tripton (non-phytoplankton particulate matter), and coloured dissolved inorganic matter (cDOM) (Gerbeaux and Ward, 1991; Pérez et al., 2011b; Zhang et al., 2007). Respectively, these three active optic substances imply that lakes are stressed by excessive nutrient loadings (Wu et al., 2017), strong wave actions (Gerbeaux and Ward, 1991; Zhang et al., 2007), and enhanced humic matter inputs (Choudhury et al., 2019; Cronin-Golomb et al., 2022). At last, DOM chemo-diversity is characterised to reveal specific pollution sources (Tanentzap and Fonvielle, 2024). For instance, DOM of sewage origins exhibit elevated S-containing compounds, particularly enriched O_3S and O_5S classes indicative of synthetic surfactant-like compounds (Hong et al., 2023; Ye et al., 2019); agricultural influences are marked by raised N-containing components and agricultural humic substances (Ge et al., 2022; Liu et al., 2020b); while industrial pollution is characterised by enrichment of O_4 class CHO compounds and presence of CHOS compounds containing two S atoms (He, 2018). Finally, stressor-impact linkages are established via collinearity analysis, among the extent of light limitation, the primary light attenuator, and derived indexes of DOM associated with various pollution sources.

Through this framework, we sequentially explored the degree of light limitation for submerged macrophytes, the primary light attenuator, and the pollution sources in urban lakes. Based on these findings, we could establish a causal pathway that links specific pollution sources to habitat degradation of submerged macrophytes in urban lakes, which provides a basis for designing effective restoration strategies.

2. Materials and methods

2.1. Study sites

We tested the diagnostic framework in urban lakes of Jiujiang city (113–116°E, 28–30°N). It is a large city adjacent to the middle section of the Yangtze River and an exemplary site for ecological protection within the Yangtze River Basin. The city has an area of 19,084 km^2, a population of 4.6 million, and an annual GDP of 403 billion RMB in 2022. The city has five urban lakes: Lake Gan-tang, Lake Nan-men, Lake Bai-shui, Lake Bali, and Lake Sai, referred to as Lake 1 to Lake 5 in the respective order (Fig. 2). Basic information on each lake is listed in Table 1. The five urban lakes all experience a typical subtropical monsoon climate, such that their water levels rise stochastically during the wet period (from late Spring to late Summer) and decrease through the following dry period (Autumn to early Spring). Lakes 1 and 2 are scenic water bodies downtown and connected via a 2 m wide channel. They share a catchment (primarily the urban area to the southeast of the lakes) where precipitations drain through an underground pipe system to them. Their waters are pumped out to the Yangtze River in wet seasons and supplied with municipal water in dry seasons to keep water depth between 2.0 and 2.5 m (personal communication with the local hydrological monitoring agency). Similarly, inflow via underground pipes to Lake 3 originates from precipitation in the lake's southeast region (forested hills and urban areas), and outflow to the Yangtze River is controllable via a sluice. Lake 3 has no additional water supply; thus, water received during the wet period is slowly lost (water level drops gradually) during the dry period. Lake 4 has three input tributaries (Sha River, Shili River, and Lianxi River in Fig. S1) that start from forested mountain regions in the Southern part of the city and flow partially through urban areas before reaching the lake. Lake 5 receives runoffs from the southern

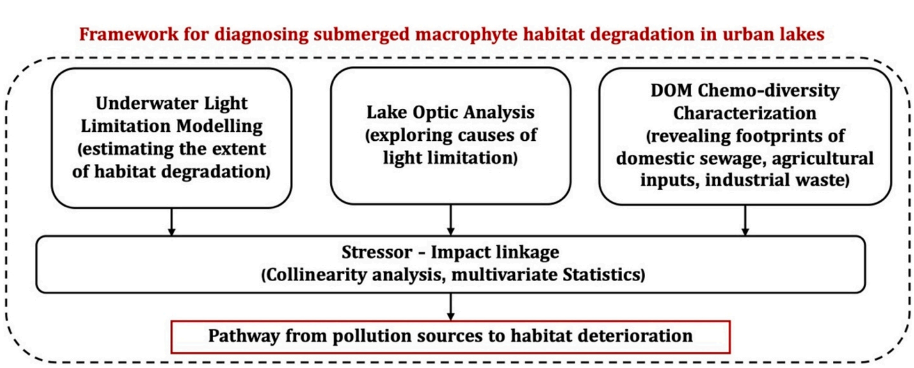

Fig. 1. A schematic flowchart of the diagnostic framework for submerged plant habitat degradation in urban lakes.

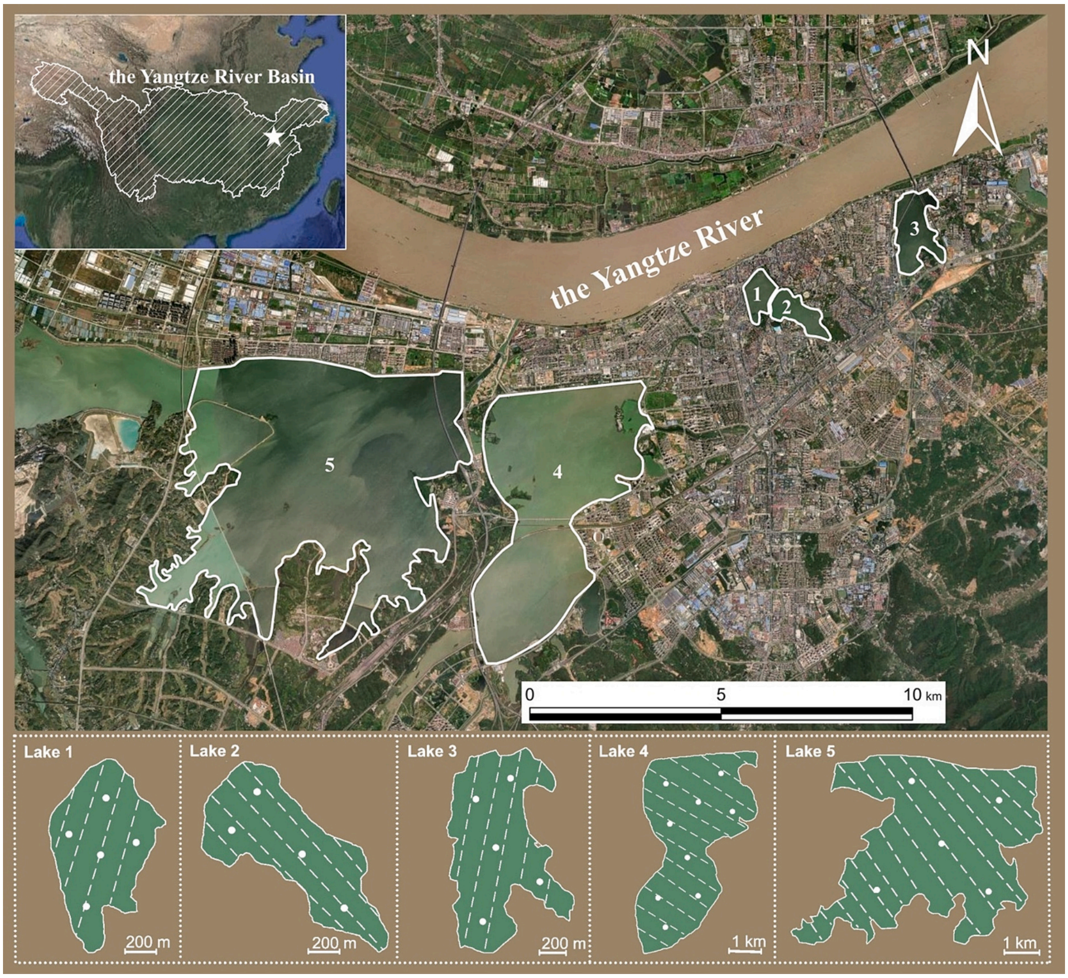

Fig. 2. (Upper Panel) A satellite image of Jiujiang city with five urban lakes (solid white outlines) along the Yangtze River (Jilin-1 Satellite Image captured on July 1st, 2022; a white star in an inserted map shows the location of the city in the Yangtze River Basin); (Bottom Panel) Sketches of the five urban lakes marked with sampling sites (dots) and parallel transects (dashed lines) for determining lake basin elevations.

Table 1
Basic lake information for the five urban lakes.

	Lake 1	Lake 2	Lake 3	Lake 4	Lake 5
Water depths (m)	1.98	1.60	2.34	3.05	2.71
Lake size (km^2) *	1.5	0.9	1.9	22.3	53.6
Catchment size (km^2) *	15.4	15.4	15.6	273	991
Catchment land use *	Urban area	Urban area	Urban area, Forests	Urban area, Forests	Forests, Farmland
Storage (million m^3)	2.79	1.44	4.45	68.02	145.26
Catchment to lake size ratio **	6.4	6.4	8.2	12.2	18.5

[*] Lake size, catchment size and land uses are from Dai (2010).
[**] Catchment to lake size ratio is calculated as the ratio of catchment size to lake size.

mountain regions and an inflow tributary (Long River in Fig. S1) that passes through forests and rural farmlands before arriving at the lake. Lakes 4 and 5 share an outflow channel to the Yangtze River. Due to their large sizes and catchment areas (Table 1), efforts for water level control are less effective. Lakes 4 and 5 show large seasonal water level fluctuations, with potential risks of flooding during wet seasons, while extremely low water levels occur during dry periods. Historically, submerged vegetation appeared in Lakes 4 and 5 at times (Liu et al., 2023; Luo et al., 2023), while it has been absent in Lakes 1, 2 and 3 for at least 5 years (personal communication with local environmental agencies). During this study, submerged vegetation only existed in shallow bay areas in Lake 5 (Liu et al., 2023).

2.2. Methods

We applied the diagnostic framework (Fig. 1) to the five urban lakes when submerged macrophytes are fully grown in the summer. Firstly, we used Monte Carlo simulations to estimate the probability distributions of water depth and light compensation depth (LCD) for submerged plants. LCD is defined as the depth at which daily gross photosynthetic production balances respiration, below which the minimal light requirement of submerged plants cannot be met (Cronin-Golomb et al., 2022; Hilt et al., 2006; Hu et al., 2019). We calculated the probability of a random site in a lake with LCD deeper than water depth (ensuring the minimal light requirement of submerged macrophytes is met) to indicate the extent of underwater light limitation. Secondly, we used stepwise regression analysis to estimate the relative contributions of chlorophyll-a, tripton, and cDOM to Secchi depth, a measure of underwater light attenuation. Thirdly, we used excitation-emission matrix (EEM) and Fourier transform ion cyclotron resonance mass spectrometry (FT-ICR MS) techniques to comprehensively characterise DOM chemodiversities, with derived indexes that inform footprints and influences of various pollution sources. Finally, we used multi-variable statistical techniques to explore collinearity among the extent of underwater light limitation, optically active substances, and the derived DOM indexes, establishing a linkage from the pollution source to habitat degradation.

2.3. Extent of underwater light limitations

To estimate the probability distribution of LCDs, we collected datasets of incident irradiance at the water surface, downwelling light attenuation coefficients, and compensation irradiances for submerged plants. We collected datasets of lake bottom elevations and water level fluctuations for the probability distribution of water depth. Incident irradiance is represented by daily averaged PAR (photosynthetically active radiation) recorded at Lushan Botanical Garden, Jiujiang, China, from 2017 to 2022 with a PAR Sensor TBQ-407 (Weiyi Tech. Ltd., China). A dataset of 720 daily averaged PAR values from June to September was collected. Underwater light attenuation coefficients (*Kd*) were determined in situ in June 2022 and retrieved from bi-monthly Secchi Depth (Z_{SD}) records between 2017 and 2020. Z_{SD} were measured by the local environmental protection agency in June and August and the local hydrological monitoring agency at multiple sampling sites in September, totalling 136 records. Compensation irradiances of leaves or phyto-elements of submerged macrophytes were from 5 to 52 μmol m^2 s^{-1} (the 5th to 95th percentiles), with a median of 16.3 μmol m^2 s^{-1} (Binzer et al., 2006). Considering the plasticity in photophysiology that low-light acclimated leaves have reduced compensation irradiance, the lower half of the reported range (5 to 16.3 μmol m^2 s^{-1}) was used to estimate LCDs. For each lake, basin elevations were determined at 50 m intervals along a transect and multiple parallel transects (Fig. 2) using an unmanned autopilot boat equipped with an echo sounder (APACH 3, CHCHAV®, China). Daily water level records from 2017 to 2021 for each lake were obtained from the local hydrological monitoring agency.

For the determination *of Kd* in situ, sampling sites of each lake were visited on two consecutive sunny and calm days after a week of no rain in June 2022. The number and locations of the sampling sites for each lake were set depending on the lake size and morphology (Fig. 2). At each site, downwelling light attenuation coefficients (*Kd*) were determined following the procedures outlined in Hu et al. (2019). Simultaneously, Z_{SD} was measured using a black and white quadrats Secchi disk of 30 cm in diameter. Pairs of *Kd* and Z_{SD} values were fitted to an empirical model: $Kd = a + b^*Z_{SD}^{-1}$ (a and b are constants) (Lee et al., 2015), using the least squares method (JMP 16, SAS®). The resulting model (Fig. S2) was then used to convert archived Z_{SD} for the summers (Jun. - Aug.) of 2017–2020 to *Kd*.

After data collection, various built-in distribution models in JMP 16 (SAS®, USA) were fitted to the datasets of incident irradiances, *Kd*, lake bottom elevations, and summer water level for each lake based on the lowest AICc (Akaike's Information Criterion with small-sample correction). A uniform distribution was assigned for datasets that cannot be adequately described (Anderson-Darling test, $p < 0.05$ or substantial deviations from the diagonal line in the QQ plot). Details of the fitted distributions are in Supplementary material 3. Next, the light compensation depth was computed following the Beer-Lambert model: $LCD = -\mathrm{Ln}\left(\frac{Ic}{Io}\right)/Kd$ (Hu et al., 2019; Kirk, 2011; Weiskerger et al., 2018) and water depths were computed by subtracting the lake bottom elevations from the water levels. Monte Carlo simulation was run 20,000 times to produce probability distributions of LCDs and water depths, using the Monte Carlo Simulator in JMP16 (SAS®, USA). To indicate the extent of light limitation, the chance of a random site with LCD deeper than the water depth was estimated for each lake based on the simulation outcome. All computations were performed in JMP 16 (SAS®, USA).

In addition to underwater light modelling, sediment organic content was determined to inform habitat suitability for submerged plants. During the field campaign, the top 20 cm of sediment at each sampling site (Fig. 2) was collected using a Peterson sediment grab. It was stored in a food-grade ziplock bag and transported back to a water chemistry laboratory at Lushan Botanical Garden (Jiujiang, China) within 6 h and kept at 4 °C. The sediment samples were dried at 80 °C until constant weight and analysed for LOI (loss on ignition) after 450 °C for 8 h to determine sediment organic content (Heiri et al., 2001).

2.4. Lake optic analysis

To investigate the relative contributions of chlorophyll-a, tripton, and cDOM to underwater light attenuation, the three optically active substances were measured in parallel with *Kd* and Z_{SD} determinations. A 0.5 L water sample was collected at 0.5 m depth at each sampling site and stored in hydrochloric acid-cleaned Nalgene bottles. It was transferred back to the water chemistry lab and stored at 4 °C. Within 24 h, the sample was analysed for concentrations of chlorophyll-a (*Chl-a*), total suspended matters (TSM), and absorption coefficient of cDOM ($a440_{CDOM}$) following the methods described in Zhang et al. (2007). Tripton was further calculated as "Tripton = *TSM* – 0.07**Chl-a*" (Zhang et al., 2007). Then, correlation analysis between Z_{SD} and the reciprocal of each optic component (Chl-a, Tripton, $a440_{CDOM}$) was performed (Brezonik et al., 2019; Nishijima et al., 2016). Significantly correlated components were step-wisely fitted to a multiple linear model to detect the one with the most contribution to maximum adjusted R^2. To assess consistencies in optic properties among the years, relationships between Z_{SD} and Chlorophyll-a concentration were compared between our field-collected and archived data for summers of 2017–2020. Both datasets were fitted to a robust model ($Z_{SD} = Chl\text{-}a^{\,b}$, *b* is a constant) with non-linear least square regressions in JMP16 (SAS®, USA).

2.5. DOM chemo-diversity

A second field campaign was carried out a week after the first to characterise DOM chemo-diversity. Five 0.5 L water samples at 0.5 m deep were collected for each lake. The sampling sites were shown in Fig. 2, except that Lake 2 had an additional site while Lake 4 had three fewer. The samples were transported to the lab on ice within 8 h. Upon arrival, water samples were immediately filtered through 0.45 μm glass fibre (pre-combusted at 450 °C for 4.5 h, GF/F, Whatman), acidified to pH 2 with hydrochloric acid and stored at −20 °C in the dark. The concentration of dissolved organic carbon (DOC) and the fluorescence excitation-emission matrices (EEMs) were measured as reported (Xu et al., 2023). An integrated water sample per lake with equal volume collected at each site was extracted for DOM via solid phase extraction. The DOM sample was then analysed in negative ion mode through a 15 T Bruker Solari X FT-ICR MS (Brucker Daltonics Inc., USA) equipped with an electrospray ionisation source (Apollo II) to acquire ultrahigh-resolution mass spectra. More details for the procedures can be found in Xu et al. (2023).

In EEM results, five groups of fluorescent substances were categorised based on their defined specific regions (Table S1). Region I (tyrosine-like protein), II (tryptophan-like protein), and IV (microbial-like) were classified as protein-like materials and likely associated with microbial transformed products (i.e. wastewater and autochthonous production); Region III (fulvic acid-like) and V (humic-like) were considered as humic-like materials and may relate to terrestrial inputs (Dong et al., 2020). Then the relative intensity of each group was calculated as the integrated fluorescent intensity of the corresponding region, divided by the sum intensity of all regions (Chen et al., 2003; Song et al., 2019). Further, fluorescence index (FI), biological index (BIX), and humification index (HIX) were calculated to inform DOM sources, the contribution of autotrophic productivity, and the degree of humification, respectively (Fellman et al., 2010). At last, difference among the five lakes in DOC concentration, relative intensity of each fluorescence group, and the fluorescence indexes were analysed with one-way ANOVA and Tukey HSD tests (JMP 16, SAS®, USA), with significant difference level set at 0.05.

For acquired mass spectra through FT-ICR MS, peak identifications were performed with Bruker Data Analysis software, and molecular formulas were assigned to each mass spectra by calibration against a reference list of natural DOM molecules, following stringent criteria as listed in our previous study (Xu et al., 2024). Normalised to the sum of the peak intensity of assigned molecular formulas in each sample, relative peak intensity was used to estimate a range of intensity-weighted parameters that semi-quantitatively characterise DOM (Koch and Dittmar, 2006; Wang et al., 2019a). Calculated intensity-weighted parameters include molecule number for each element (C, H, O, N, S), molecular weight (m/z), formula groups (CHO, CHOS, CHON, CHONS), atomic ratios (H/C, O/C), and modified aromatic index (AI_{mod}). Further, the relative intensities of the CHOS group by different molecule numbers of oxygen (from O_2S to $O_{15}S$) were calculated and plotted.

Assigned formulas were classified into various compounds as follows: polycyclic aromatics ($AI_{mod} > 0.66$), polyphenolics ($0.66 \geq AI_{mod} > 0.50$), highly unsaturated and phenolic compounds ($AI_{mod} \leq 0.50$, $H/C < 1.5$), aliphatic compounds including unsaturated aliphatics ($2.0 > H/C \geq 1.5$, $N = 0$) and peptide-like ($2.0 > H/C \geq 1.5$, $N > 0$), saturated compounds ($H/C \geq 2.0$, or $O/C \geq 0.9$) (Kellerman et al., 2018; Seidel et al., 2014). In addition, carboxylic-rich alicyclic molecules (CRAMs; DBE/C = 0.30–0.68, DBE/H = 0.20–0.95, DBE/O = 0.77–1.75) (Hertkorn et al., 2006), the island of stability index (IOS%) (Lechtenfeld et al., 2014), and molecular lability index (MLB_L%) (D'Andrilli et al., 2015) were also identified.

2.6. Collinearity analysis

Finally, principal component analysis (PCA) based on a correlation matrix was performed (JMP 16, SAS®, USA) to explore the multi-collinearities among the extent of light limitation, the three lake optic variables, and various derived DOM indexes that reflect the influence or footprints of different pollution sources.

3. Results

3.1. Extent of underwater light limitation

The extent of underwater light limitation was significantly greater in Lakes 1, 2, 3, and 4 than in Lake 5. Compatibly, LCDs and Z_{SD} were lower in the first four lakes than in Lake 5. The probabilities of a random site with LCD deeper than water depth in summer were 1.3 %, 6.7 %, 2.3 %, and 6.8 % for Lake 1, Lake 2, Lake 3, and Lake 4, respectively, while it was 19.4 % for Lake 5 (Fig. 3). Compatibly, the estimated LCDs in Lake 1, Lake 2, Lake 3, and Lake 4 (with 25th to 75th being 0.85–1.25 m, 0.83–1.12 m, 0.87–1.19 m, and 0.95–1.35 m in the respective order) were all shallower than in Lake 5 (1.41–1.92 m) (Fig. 3). The summer Z_{SD} (2017–2022) were different among the five lakes (Kruskal-Wallis Tests, $p < 0.001$), significantly less in Lakes 1, 2, 3 and 4 than in Lake 5 (Dunn test, $p < 0.001$), but not different among the first four lakes (Dunn test, $p > 0.05$) (Fig. 4). In contrast, the estimated water depths demonstrated an incompatible pattern with the difference in the extent of light limitation among the five lakes. Water depths were greater in Lakes 4 and 5 (with 25th - 75th being 2.2–3.9 m and 1.9–3.4 m, respectively) than in Lakes 1, 2, and 3 (1.8–2.2 m, 1.4–1.8 m, and 2.1–2.6 m, respectively) (Fig. 3). In addition, sediment LOI (loss on ignition) for Lake 1, Lake 2, Lake 3, Lake 4, and Lake 5 were 15.1 ± 1.5 %, 16.9 ± 0.5 %, 12.1 ± 4.1 %, 10.2 ± 1.6 %, and 12.9 ± 1.0 %, respectively.

3.2. Lake optics

The variation in Z_{SD} was primarily explained by chlorophyll-a and, to a lesser extent, by tripton for the summer of 2022 (field campaign). And lake optic properties were similar between the summer of 2022 (field campaign data) and those of 2017–2020 (archived data). For the summer of 2022, both *Chl-a* and tripton were significantly related to Z_{SD} ($p < 0.001$) (Fig. 5). A general linear model with both variables achieved the highest explanatory power at 79 % (Table 2, Model 3). Excluding *Chl-a* as a model variable reduces the explanatory power by 42 % (adjusted R^2 from 0.79 in Model 3 to 0.37 in Model 2) (Table 2), whereas excluding tripton drops it by only 2 % (adjusted R^2 from 0.79 in Model 3 to 0.77 in Model 1) (Table 2). The comparison of lake optic characteristics between the summer of 2022 and those of 2017–2020 revealed strong similarities. Both datasets fit well to an empirical Z_{SD} versus *Chl-a* model with little coefficient difference (Fig. S3). Respectively, they are:

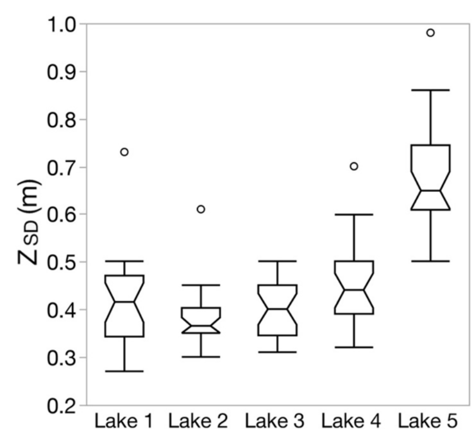

Fig. 4. Boxplot of Secchi depths collected over the summers for each lake.

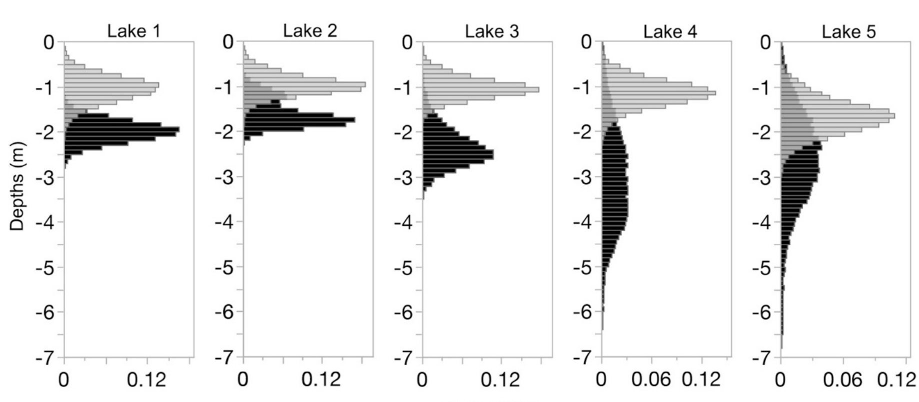

Fig. 3. Probability histogram of estimated light compensation depths (grey) against that of water depths (black) in the summer for each urban lake.

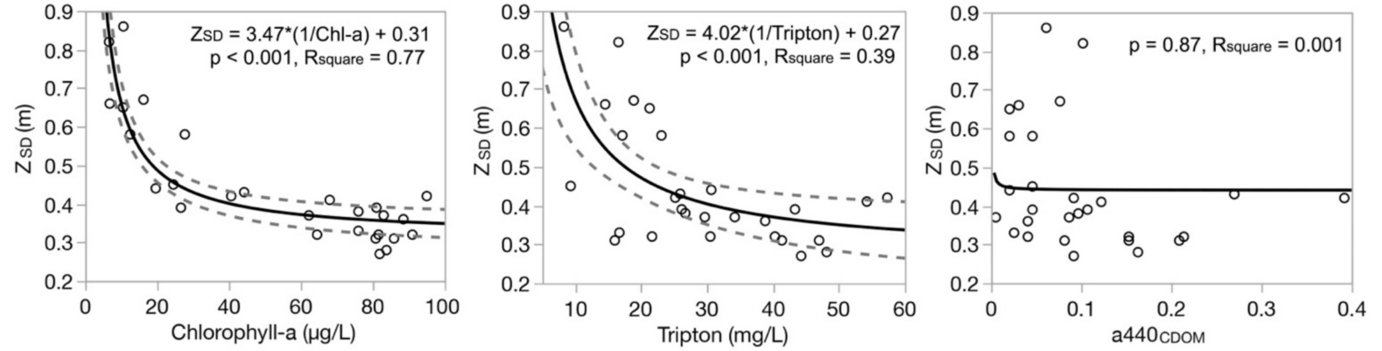

Fig. 5. Linear regression analysis between Z_{SD} and the reciprocal of each of the optic active substances (chlorophyll-a, tripton, and cDOM) (dotted line indicates the 95 % confident intervals of the fitted lines).

Table 2
Stepwise model constructions that use chlorophyll-a, tripton, or both as variables to predict Secchi depth, with reported adjusted R^2 and *p*-values.

Model	Stepwise Model Construction	Adjusted R^2	p-value
Model 1	$Z_{SD} = 3.47*Chl\text{-}a^{-1} + 0.31$	0.77	<0.001
Model 2	$Z_{SD} = 4.02*Tripton^{-1} + 0.27$	0.37	<0.001
Model 3	$Z_{SD} = 3.03*Chl\text{-}a^{-1} + 1.28*Tripton^{-1} + 0.28$	0.79	<0.001

$Z_{SD} = Chl\text{-}a^{-0.226}$ (RMSE = 0.085, d.f. = 27) for the campaign dataset, and $Z_{SD} = Chl\text{-}a^{0.248}$ (RMSE = 0.103, d.f. = 78) for the archived dataset.

3.3. DOM chemo-diversities

Averaged DOC ranged from 4.0 to 8.8 mg/L and was the lowest in Lake 5 among the five lakes (Fig. 6). DOM fluorescence intensity was, on average, the weakest in Lake 5, with Lake 5 being statistically weaker than Lake 1 and 2 (Fig. 6; Fig. S4). Among the fluorescing groups, the most and the second most abundant compounds were fulvic acid-like (34.5–36.3 %) and tryptophan-like materials (26.9–28.5 %) (Fig. 6; Fig. S4). Meanwhile, the proportions of humic-like substances (including agriculture humic-like or reprocessed terrestrial humic materials) were small, ranging from 9.4 to 10.1 % (Fig. 6). Further, derived FI values that inform DOM sources (< 1.2: terrestrially derived DOM from higher plants or soil organic matters; 1.2–1.8: a mixture of both terrestrial and microbial-derived; > 1.8: DOM derived from extracellular release and leachate from bacteria) were high (> 1.8) for Lake 1, 2, 3, and 4 (Fig. 6), whereas for Lake 5 it was 1.7, a value within the normal range of natural waters (1.2–1.8). Two other derived indexes (HIX and BIX) are shown in Fig. S5.

On the molecular level, the number of molecular formulae identified for the five lakes ranged from 6855 to 7927 (Table S2). Among the identified formulae, CHO, CHON, CHOS, and CHONS compounds account for 57.7–68.4 %, 21.7–28.3 %, 6.0–12.6 % and 1.2–2.8 % of DOM, respectively (Table S2). The proportions of CHOS and CHONS compounds demonstrated the most salient variations among the five lakes. They were nearly twice as much in Lakes 1, 2 and 3 as in Lakes 4 and 5 (Table S2). Similarly, the molecular number of S (rather than N) in Lakes 1, 2, and 3 was nearly twice that in Lakes 4 and 5 (Table S2). Further expansion of the CHOS group revealed abnormally high intensities of O_5S formulas in Lakes 1, 2, 3 and 4 (Fig. 7) (As in natural waters, the relative intensities from O_2S to $O_{15}S$ classes follow an approximate normal Gaussian profile with a maximum in class O_7S). The O_5S compounds accounted for 13.3 %, 14.6 %, 12.2 %, 12.8 % and 9.5 % of the CHOS compounds for Lakes 1, 2, 3, 4, and 5, respectively. The majority of the CHOS compounds belonged to the highly unsaturated and phenolic group (mostly degraded terrestrial DOMs) and the aliphatic groups (derived from bacteria and algal metabolites) (Fig. S6). Agreeably, the two groups were respectively the most and the second most abundant groups for all DOM compounds of each lake (Table S2).

3.4. Collinearity analysis

In principal component analysis (PCA) for a constellation of the degree of underwater light limitation, the optic variables and the derived DOM indexes, the first two components captured 91 % of the total variation (Fig. 8). In the loading plot, the probability of a random site with LCD deeper than water depth correlated highly with Secchi depth and negatively with chlorophyll-a, which is compatible with previous findings. It also demonstrates strong negative correlations with tryptophan-like proteins, DOC, and proportions of O_5S in CHOS, all of which share high collinearities (Fig. 8).

4. Discussion

4.1. Diagnosed habitat degradation pathway for submerged macrophytes

Justified by the extent of underwater light limitations, submerged macrophyte habitat was significantly degraded in the five urban lakes. Lakes 1, 2, 3, and 4 experienced a higher degree of deterioration than Lake 5, as the probabilities of a random site that satisfies the minimal light requirement of submerged macrophytes (LCD greater than water depth) were much lower in Lakes 1–4 than in Lake 5 (Fig. 3). LCD and Z_{SD} showed a compatible pattern that was significantly lower in Lakes 1–4 than in Lake 5, in contrast to water depths with an incompatible pattern (Lake 4 and 5 greater than Lakes 1, 2, and 3). Therefore, the underwater light limitation was attributed to constrained LCD and Z_{SD} rather than water depths. Subsequently, lake optic analysis identified chlorophyll-a as the primary cause of light attenuation in urban lakes during the summer (Fig. 5), a common symptom of eutrophication (Zou et al., 2020). Therefore, high nutrient loadings were considered to cause submerged plant habitat degradation in urban lakes. In addition, the sediment organic content of the five urban lakes was below the critical threshold at 17 %, above which reducing the environment proves unsuitable for the growth of submerged plants (T., Cao, personal communication, October, 2020). Thus, the lake sediment was still deemed suitable for the growth of submerged plants.

Nutrient enrichments remain the dominating stressor for freshwater lakes worldwide (Moss, 2007; Poikane et al., 2024). The excessive nutrients in urban lakes are potentially from non-point agricultural sources, domestic wastewater, and/or industrial effluent (Wang et al., 2023; Zheng et al., 2022). To diagnose which of them was the primary source in our case, we interpreted the results of the DOM chemo-diversity. DOM characteristics demonstrate footprints of both wastewater origins and domestic sewage for Lakes 1, 2, 3, and 4 in reference to Lake 5, but limited agricultural and industrial pollution. In EEM analysis, the most and second most dominating fluorescing components were fulvic acid-like materials and tryptophan-like materials (Fig. 6). Fulvic acid-like

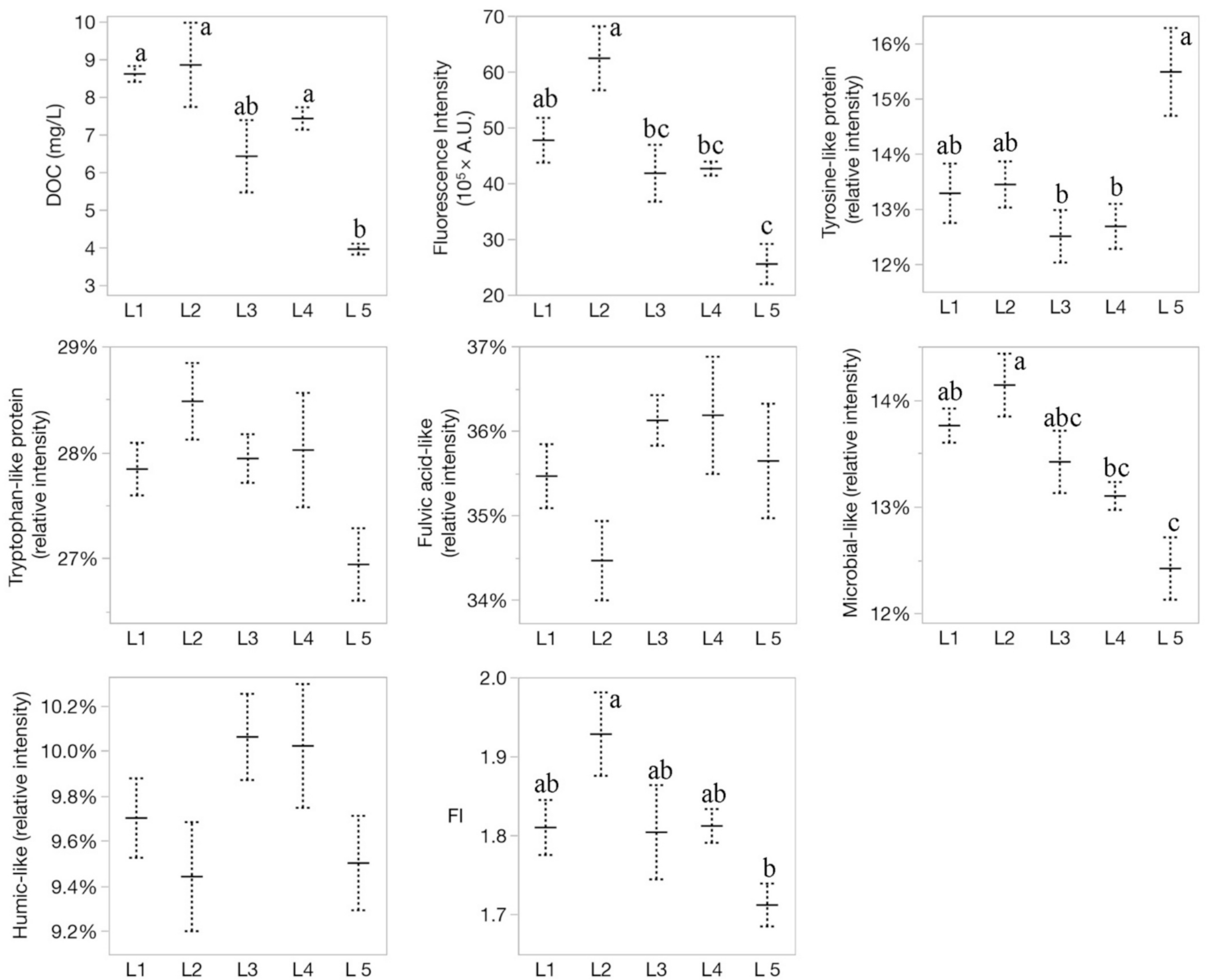

Fig. 6. DOC concentrations and EEM analysis derived indexes for the five urban lakes (L1 to L5 represent Lake 1 to Lake 5 in the respective order; solid bar represents the mean and dashed error bar represents one standard error with $n = 5$; different letters indicate significant differences among the lakes with $p < 0.05$).

substances are widespread in wetlands and forested environments, and tryptophan-like materials are widely present in wastewater (Dong et al., 2020). Their presence suggested a mixture of wastewater and terrestrial DOM origins (Dong et al., 2020). Meanwhile, the low proportions of humic-like substances imply limited agricultural impacts (Ge et al., 2022; Liu et al., 2020b). Compatibly, high FI values (>1.8) in Lakes 1, 2, 3, and 4 were most likely associated with high wastewater input, in contrast to Lake 5 with a value of 1.7 that is within the normal range (1.2 to 1.8) of natural waters (Fellman et al., 2010). In FT ICR MS analysis, the elevated heterogenous formulas (CHOS and CHONS groups) and molecular number of S (Table S2) indicate greater anthropogenic impacts in Lakes 1, 2 and 3 than in Lakes 4 and 5 (Xu et al., 2024). In particular, the enhanced proportions of O_5S in the CHOS group for Lakes 1, 2, 3, and 4 (Fig. 7) were likely associated with degraded surfactants popular in domestic sewage (Wang et al., 2019b; Ye et al., 2019), in reference to Lake 5. However, the low molecular number of S (less than 2) (Table S2) in all lakes suggested limited industrial pollution (He, 2018). As an integration, PCA analysis reveals the probability of a random site that satisfies the minimal light requirement of submerged plants was positively related to Z_{SD} and negatively related to chlorophyll-a content and the derived DOM indexes (tryptophan-like materials and proportions of O_5S in CHOS) that reflect anthropogenic influences and domestic sewages (Fig. 8). Meanwhile, chlorophyll-a and the DOM indexes were highly correlated.

Based on the analysis above, a degradation pathway is implied that domestic sewage discharge introduced excessive nutrients to the urban lakes that fuelled algal overgrowth, which enhanced underwater light attenuation and induced light limitation for submerged plants. The implied degradation pathway agrees with reality. Catchments of Lakes 1, 2, 3 and 4 all possess densely populated urban areas, while that of Lake 5 mainly consists of forested mountains and farmlands (Table 1) (Dai, 2010). Lake 1–4 may have received nutrients from domestic sewage through urban runoffs due to inappropriate rainwater and sewage diversions (Hongwei et al., 2008; Peng et al., 2004). Then, sewage inputs may have caused algal bloom and losses of submerged plants, as evidenced by the fact that Lakes 1, 2, 3, and 4 were devoid of submerged vegetation, while Lake 5 was not.

Further, the difference in the extent of habitat degradation, lake optics, and DOM-derived indexes among the five urban lakes may be explained by lake hydrology. Because the five lakes are in the same region, it can be assumed that precipitation, lake depths, groundwater movements (baseflow pattern), and slope of the drainage area (runoff pattern) are similar among them (Fu et al., 2024). Then, the catchment-to-lake size ratio could indicate the amount of water flushing lakes of a

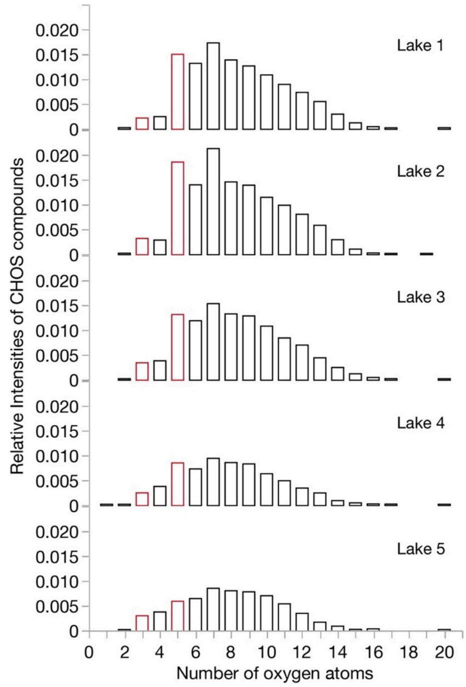

Fig. 7. Relative intensities of the CHOS formula compounds by the number of oxygen atoms (red bars represent the intensities of the O_3S and O_5S compounds that are indicative of synthetic surfactant-like compounds in domestic sewages). (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

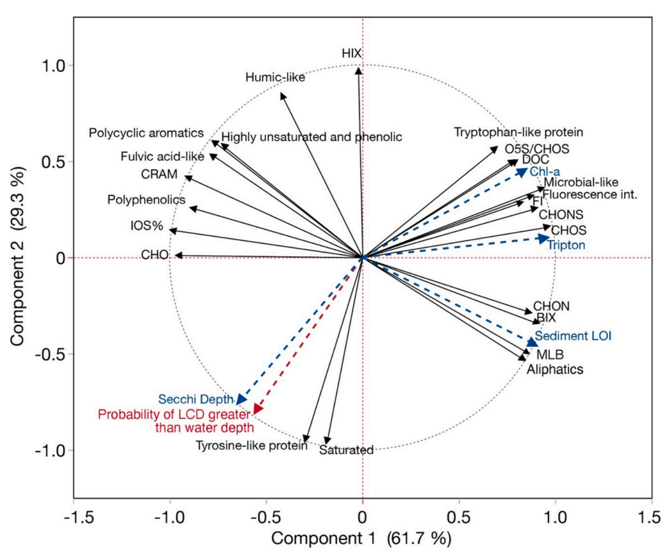

Fig. 8. Loading plot of the principal component analysis for the extent of light limitation (dashed red vector), habitat variables (dashed blue vector) and DOM-derived indexes (black vector) (correlation between any two variables are manifested by angles between the two corresponding vectors, with correlation coefficient estimated as the cosine of the corresponding angle). (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

standardised size in relative terms. Lake 5 has a much larger catchment-to-lake size ratio than Lake 1–4 (Table 1), implying water exchange is much more rapid in Lake 5 than in Lake 1–4. The rapid water exchange in Lake 5 could constrain phytoplankton growth (Odebrecht et al., 2015; Zhu et al., 2019) and dilute DOM content, favouring underwater light penetration and submerged plant colonisation.

4.2. Implication and advantages of the proposed diagnostic framework

Lake restoration is a complex task that requires collaborations among multiple distinctively different knowledge domains, including ecological theories (e.g. regime shift), environment science, policy development, financial investment, and engineering technologies (Howard-Williams et al., 2018; Ostrom, 2009). The nature of the task contrasts the mechanistic and reductionistic worldview of science with highly diversified disciplines, among which effective communication is often lacking (Ostrom, 2009). The isolated knowledge domains in lake restoration underpin the lack of linkage between stressors and submerged plant habitat degradations (Abell et al., 2022; Poikane et al., 2024; Sondergaard et al., 2007). In such a context, our diagnostic framework that integrates ecological (the driving role of underwater light in submerged plant dynamism), physical (partitioning of underwater light attenuation), and chemical knowledge (DOM chemo-diversity) may have provided insight for urban lake management. Recently, the integration of multiple knowledge domains in habitat assessments for aquatic animals has been reported (Sedighkia and Datta, 2023; Thomson-Laing et al., 2024; Yang et al., 2021); our study could serve as an early example for that of submerged plants in shallow lakes.

The framework involves two technical improvements. One is the modelling of underwater light limitations (Van der Lee et al., 2006). Underwater-light climate always demonstrates spatiotemporal variations in lakes (Gerbeaux and Ward, 1991; Pérez et al., 2011a; Zhang et al., 2007; Zhang et al., 2020). In comparison to previous studies using discrete data (Hu et al., 2019; Xu et al., 2020), our modelling approach has accounted for natural variations and uncertainties in lake bathymetry, sunlight, water level fluctuations, and photo-physiology of submerged plants (Supplement File 1) to assess habitat suitability for submerged plants. The second is the use of DOM chemo-diversity to trace pollution sources. Often isotopic element (e.g., nitrogen and/or carbon) compositions have been widely used to trace contamination in aquatic ecosystems (Romanelli et al., 2020; Zhou et al., 2022). However, the presence of multiple sources and biogeochemical transformations could easily blur the distinctive ranges of isotopes that define a specific pollution source (Nestler et al., 2011; Xue et al., 2009). In contrast, DOM-derived footprints of various pollution sources (domestic sewage, industrial effluent, and agricultural runoffs) are characteristically distinctive, even in a mixture with multiple pollution sources (Tanentzap and Fonvielle, 2024).

4.3. Limitations of the study

The diagnostic framework provided a process-based causal pathway that sewage pollution had deteriorated the underwater light climate for submerged macrophytes in Lakes 1, 2, 3, and 4, in reference to Lake 5. Therefore, reducing domestic sewage input (such as controlling urban runoff and upgrading stormwater infrastructure) should be prioritised as an effective restoration measure. However, reducing sewage input may not be enough. A significant concern is that previously discharged sewage may have precipitated and accumulated in sediments of the lakes, particularly during the dry period when outflow is low, forming internal nutrient loadings. These internal loadings could fuel algal blooms and prevent the establishment of submerged plants after external loading reductions (Abell et al., 2022; Liu et al., 2022; Wu et al., 2017). Although the sediment characteristic for root anchorage and growth was considered in our study, its potential as an internal nutrient loading has been overlooked. The formation of internal nutrient loadings can signal an advanced level of habitat deterioration compared to an initial level when the amount of precipitated sewage in sediment is insufficient to trigger an algal bloom. Internal loading requires sediment dredge or capping to remove or immobilise sediment nutrients as lake restoration measures (Abell et al., 2022; Hilt et al., 2018). In future studies, we will add another module in the framework that evaluates lake internal loadings. The new framework can diagnose not only the presence of stressors but also the level of habitat degradation and the trajectory of ecological collapse, as recommended by Neeson et al. (2016).

Water level fluctuation in extreme weather is another potential factor that might interfere with the habitat suitability of submerged macrophytes, but was not accounted for in the framework. As introduced earlier (Section 2.1), there are risks of extremely high-water levels during the wet season or extremely low levels during the dry season for Lakes 4 and 5. They could expose submerged plants to low-light stresses and desiccation, respectively. In addition, fish communities in these urban lakes are still unknown; neither benthivores nor zooplanktivorous fish are favoured for the growth of submerged plants (Abell et al., 2022; Sondergaard et al., 2007). A more comprehensive habitat diagnosis needs to involve those factors as well.

5. Conclusion

This study provides a diagnostic framework for the degradation of submerged macrophytes' habitat by integrating underwater light limitation modelling, lake optical analysis, and DOM chemo-diversity. Our findings revealed that underwater light limitation—primarily due to algal overgrowth fueled by domestic sewage—was the main barrier to plant recolonisation in urban lakes in Jiujiang City, China. The framework addresses a critical gap in restoration planning: the mechanistic linkage between stressors and habitat degradation. The framework clarifies cause-and-effect pathways and offers a practical tool for guiding restoration. Effective restoration efforts should prioritise sewage reduction, sediment management, and hydrological stabilisation. Future improvements should incorporate internal nutrient loading and hydrological stressors to better capture the trajectory of ecological decline. This study thus offers a multi-disciplinary diagnostic framework for managing aquatic plant habitat loss in urban lakes.

CRediT authorship contribution statement

Qian Hu: Writing – review & editing, Writing – original draft, Project administration, Methodology, Funding acquisition, Formal analysis, Data curation, Conceptualization. **Lei Xu:** Writing – original draft, Visualization, Methodology, Formal analysis, Data curation. **Songhe Jiang:** Validation, Resources, Data curation. **Manqi Chang:** Writing – review & editing, Validation, Data curation. **Aiwen Zhong:** Writing – review & editing, Supervision, Resources, Funding acquisition. **Wenkai Li:** Visualization, Validation, Data curation. **Hongmei Zhao:** Validation, Supervision, Resources.

Declaration of competing interest

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

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.ecoinf.2025.103307.

Data availability

Raw and processed datasets used in this study are uploaded and can be downloaded at:

https://figshare.com/articles/dataset/Data_and_Analysis_for_Diagnosing_habitat_degradation_of_submerged_plants_in_urban_lakes_an_integrated_analysis_of_underwater_light_limitation_lake_optics_and_DOM_chemo-diversity_/28904600

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