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Development of a phytoplankton-based index of biotic integrity for ecological health assessment in the Yangtze River

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Abstract

Background The application of index of biotic integrity (IBI) to evaluate river health can be an essential method for river ecosystem management. However, these types of methods were developed in small, low-order streams, and are therefore, infrequently applied to large rivers. To that end, phytoplankton communities and environmental variables were monitored in 30 sampling segments of the middle and lower reaches of the Yangtze River, China during the wet (July–August) and dry (November–December) seasons in 2017–2018. We developed a phytoplankton-based index of biotic integrity (P-IBI) and used the index to assess the ecological health of the Yangtze River. Relationships among P-IBI, its component metrics, and environmental factors were analyzed across different seasons.

Results Results obtained from the P-IBI indicated that the phytoplankton-based ecological health of the Yangtze River was rated as "good" during both seasons, with an overall better condition in the dry season. During the wet season, there were scattered river segments with P-IBI ratings of "fair" or below. Water quality and land use appeared to shape the patterns of P-IBI. In the wet season, P-IBI negatively correlated with total phosphorus, nitrate, total suspended solids, turbidity, conductivity, and dissolved oxygen. In the dry season, P-IBI positively correlated with total nitrogen, ammonium, and nitrite, and negatively correlated with water temperature.

Conclusions The ecological health of the Yangtze River as reflected by the P-IBI exhibited spatial and temporal variability, with the effect of water quality being greater than that of local land use. This study indicated the importance of considering seasonal effects in detecting large river ecological health. These findings enhanced our understanding of the ecological health and characterized potential benchmarks for management of the Yangtze River. These findings also may be applicable to other large rivers elsewhere.

Keywords Large river, Index of biotic integrity (IBI), Ecological health assessment, Water quality, Land use

Introduction

The origin and development of human civilizations is closely interrelated to large rivers (Best 2019). River ecosystems provide multiple ecological services such as water supply, recreation, transportation, and biodiversity as well as enriching landscape aesthetics (Kamp et al. 2007; Allan et al. 2021). Unfortunately, with expanded agriculture, urbanization, and industrialization needed by increasing human populations, overconsumption of natural resources has occurred on broad scales, resulting in a series of ecological and environmental issues (Chen et al. 2017a; Xia et al. 2020). A variety of river ecosystem problems resulting from multiple stressors include significant losses of biodiversity, increased water pollution, and degradation of aquatic habitats (Sala et al. 2000; Malmqvist and Rundle 2002; Reid et al. 2019). These

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factors in concert have shifted many large river ecosystems from “healthy and sustainable” to “unhealthy and unsustainable” in many areas of the world (Tickner et al. 2020).

The United States’ Clean Water Act of 1972 proposed criteria for river health as physical, chemical, and biological integrity, where “integrity” refers to conditions that maintain the natural structure and function of the ecosystem (Karr 1999). With the development and progress of ecological research, there has been an intensive exploration of the definition and methods that should be used to assess river health (Costanza and Mageau 1999; Norris and Thoms 1999; An et al. 2002). Although there are various approaches to assessing river health, predictive model methods and multi-variable assessment methods are the two main classes of approaches used (Wang et al. 2019; Lu and Chen 2020; Ruaro et al. 2020). The index of biotic integrity (IBI) contains numerous biological metrics that reflect community structure and diversity. Each metric is sensitive to one or more types of environmental disturbances, and can be used to describe interrelationships between biological properties and human disturbances (Karr and Chu 1997; Wang et al. 2019). Applications of IBI in aquatic ecosystem health assessments include many biological communities such as fishes (Karr 1981; Hughes et al. 2004; Detenbeck and Cincotta 2008; Casatti et al. 2009; Cooper et al. 2018; Yang et al. 2020), aquatic plants (Miller et al. 2006), zooplankton (Cai et al. 2020), macroinvertebrates (Chen et al. 2017b; Effert-Fanta et al. 2019), microbes (Li et al. 2018), periphyton (Wu et al. 2012a), and phytoplankton (Wu et al. 2012b; Feng et al. 2021; Hu et al. 2022).

Phytoplankton, as the preeminent primary producers in aquatic ecosystems, connects the physical (sunlight) and chemical (water quality) environment to secondary consumers, and thus, are highly essential components in aquatic food webs (Cardinale et al. 2002; Khan 2003). Compared with other aquatic organisms, phytoplankton are nearly microscopic with short life cycles, highly sensitive to environmental changes (Reynolds 2006), and reflect river ecosystem conditions (Abonyi et al. 2018; Wu et al. 2023). Therefore, the composition and diversity of phytoplankton communities are often used as biological indicators for river environmental conditions, ecosystem health, trophic status, and water quality (O’Farrell et al. 2002; Padisák et al. 2006; Katsiapi et al. 2011; Huang et al. 2019; Zhang et al. 2021). Studies using a phytoplankton-based index of biotic integrity (P-IBI) are being increasingly applied to freshwater ecosystems (Zhang et al. 2019, 2020; Lin et al. 2021; Feng et al. 2021), though most of these applications have been in lakes, reservoirs, and small- to medium-sized rivers. Furthermore, few P-IBI studies have considered temporal dynamics, which are commonly observed with phytoplankton (Zhu et al. 2021). Existing large-river P-IBI studies have focused on local river reaches (e.g., Tan et al. 2017; Liu et al. 2020), largely because of the logistical and cost challenges needed to assess an entire river over multiple seasons (Xiong et al. 2021, 2022). As a result, there is a scarcity of information on both the spatiotemporal dynamics of phytoplankton in large rivers and the use of P-IBI in assessing large-river ecological health (Wu et al. 2012b; Feng et al. 2021).

The Yangtze River in China is one of the largest rivers in the world. The Yangtze River basin accounts for 40% of China’s gross domestic product (GDP), and is occupied by one-third of China’s population (Chen et al. 2017a). Rapid economic development and urbanization that began in the 1950s coupled with many other stressors have caused severe water pollution, habitat losses, and structural and functional degradation of biological communities throughout the basin (Chen et al. 2020; Xiong et al. 2022, 2023). For instance, in their 2020 Living Yangtze Report, the World Wildlife Fund (WWF) assessed the health of the Yangtze River using various indices that encompassed hydrological processes (including floodplain connectivity), water quality, and aquatic biota and their habitats (WWF 2020). The assessment contained five ratings ranging from A (best) to E (worst). The middle Yangtze River main-stem rated as C whereas the upper and lower reaches of Yangtze River main-stem rated as B-. These ratings yielded an overall rating of B- for the entire Yangtze River main-stem (WWF 2020).

In this study, we established a P-IBI to assess the spatial and temporal ecological health of the middle and lower reaches in the Yangtze River. Our specific research objectives were two-fold: (1) To establish a P-IBI to assess river health and detect its spatial and temporal patterns in the middle and lower reaches of the Yangtze River; and (2) To assess relationships between the P-IBI and various environmental factors in same Yangtze River reaches. Results from the current study should provide better understanding of the overall ecological health of the Yangtze River and be applicable for assessing and managing other large rivers elsewhere.

Materials and methods

Study area and sampling sites

The Yangtze River is approximately 6300 km long; it is the longest in Asia and the third longest river in the world. It flows from the Qinghai-Tibet Plateau to the East China Sea, draining an area of 1.8×10^6 km^2 that accounts for 19% of the land mass in China (Chen et al. 2016a). The Yangtze River Basin crosses three economic zones in eastern (Shanghai), central (Wuhan), and western (Chongqing-Chengdu) China, and has experienced intensive levels of many different human-related stressors (Chen et al. 2017a). The current study focuses on the middle and lower reaches of the Yangtze River downstream of Yichang, Hubei Province, which has a total river length of 1893 km (Xiong et al. 2021). Within this reach of the Yangtze River, we sampled 30 and 25 river segments during the wet (July–August) and dry (November–December) seasons, respectively, during 2017 and 2018 (Additional file 1: Table S1), with the wet-season water flow being significantly greater than the dry season ($P<0.01$) (Additional file 1: Table S2). The annual runoff in the Yangtze River during 2017–2018 was not significantly different from past decades. For instance, the annual runoff at Yichang in 2017 and 2018 was 4403 and 4738 billion m^3, respectively. By comparison, the average runoff during a recent 10-year period (2008–2018) and a multi-decade period (1950–2015) were very similar to 2017–2018 values at 4214 and 4304 billion m^3, respectively (The Changjiang Water Resources Commission of the Ministry of Water Resources, http://www.cjh.com.cn/, http://www.cjw.gov.cn/). Given the above characteristics, the aforementioned sampling segments were considered representative of the middle and lower Yangtze River, with segments S1–14 established for the middle reach and segments S15–30 established for the lower reach (Fig. 1; Additional file 1: Table S1).

Phytoplankton and environmental factors analysis

Water quality and phytoplankton were sampled from each of the aforementioned river segments (i.e., S1–S30). Each river segment was about 2 km long with three sampling sites established within (Xiong et al. 2021). We used a plexiglass water sampler to collect 2-L water samples about 0.5 m from the water surface at each sampling site. Each 2-L water sample was placed into polyethylene bottles, stored in a mobile refrigerator, and sent to the laboratory for water quality analysis. Another 1-L water sample also was collected at the same location and placed into brown polyethylene bottles, with 15-mL of Lugol's solution added in situ for phytoplankton fixation. Those samples were sent to the laboratory to settle the phytoplankton cells for at least 48 h. The 1-L phytoplankton samples were concentrated to 30–50 mL using the siphon method for subsequent cell counting and species identification. Because the sedimentation method using Lugol's solution has been widely used in other Yangtze River studies, we use it as our standard method so better insure data consistency with previous studies (e.g.,

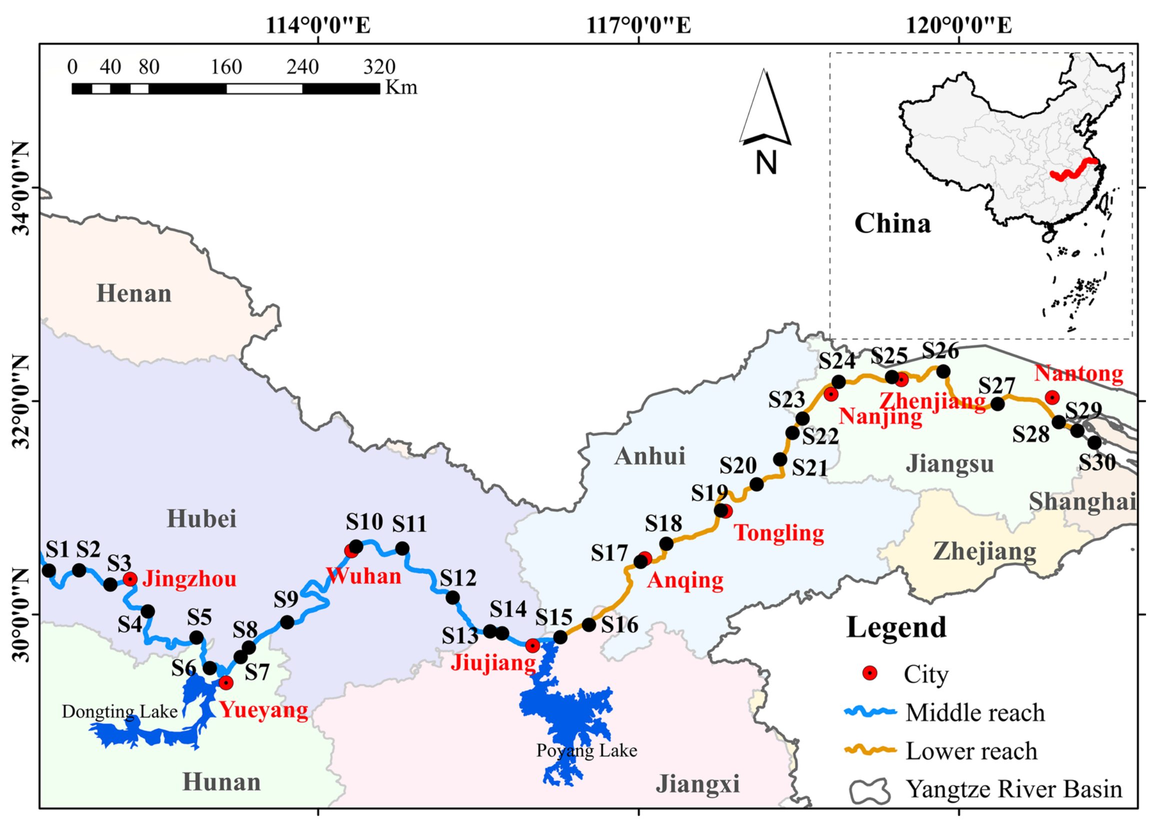

Fig. 1 Distribution of sampling segments in the middle and lower reaches of the Yangtze River, China during 2017–2018. Segments S1–14 are in the middle reach, and segments S15–30 are in the lower reach

Zhang et al. 2018; Huang et al. 2021). The specific concentrated volume of the sample was determined based on the turbidity and algal density of the sample at the time of phytoplankton sample identification and analysis. Phytoplankton samples were analyzed and identified under a microscope (BX48, Olympus Optical Co., Tokyo) at 400× magnification with a 0.1-mL plankton counting chamber (20 × 20 mm) (Ministry of Ecology and Environment of the People's Republic of China 2002; Huang et al. 2019; Tian et al. 2021). The phytoplankton collected were identified to the lowest possible taxon according to Hu and Wei (2006), with phytoplankton biomass calculated from the cell volume (Zhang et al. 2019). The cell density for each phytoplankton taxon was calculated following Reynolds (2006) as:

$$N = A \times n \times V_1/(A_S \times A_N \times V_0), \tag{1}$$

where N equals the phytoplankton cell density (cells/L), A equals the area of the phytoplankton counting chamber (mm^2), n equals the number of phytoplankton cells counted by microscopic observation (cells), V_1 equals the volume of the concentrated sample (mL), A_S equals the area of each view field of the microscope (mm^2), A_N is equals the number of view fields (in this study, we counted at least 100 view fields), and V_0 equals the volume of the phytoplankton counting chamber (mL).

Environmental factors measured or estimated included water quality, physical habitat, and land use. We measured a total of 13 physicochemical water quality parameters. Among them, water temperature (WT, °C), pH (std. units), conductivity (CON, μS/cm), and dissolved oxygen (DO, mg/L) were measured in the field using a portable multiparameter meter (YSI Professional Plus, USA). Turbidity (TUR, NTU) was measured with a portable turbidity meter (HACH 2100Q). The remaining eight parameters, including total nitrogen (TN, mg/L), ammonium (NH_4-N, mg/L), nitrate (NO_3-N, mg/L), nitrite (NO_2-N, mg/L), total phosphorus (TP, mg/L), orthophosphate (PO_4-P, mg/L), total suspended solids (TSS, mg/L), and chemical oxygen demand (COD, mg/L), were measured in the laboratory in accordance with standard methods (APHA 2005; Qu et al. 2020; Xiong et al. 2022).

The physical habitat assessment focused on three aspects. For the riparian zone, we assessed riparian zone width, land use type, vegetation conditions, modifications, and human disturbances. For the river channel, we determined water quantity and channel structure. For the river bed, we determined substrate composition, stability, and habitat complexity. The specific habitat survey methods and assessment results were obtained from a previous study (Lu 2020). We quantified riparian land use within a 5-km buffer zone surrounding the segment, which included 5-km upstream and 5-km downstream from the middle sampling site within each river segment, and 5-km landward from each side of the river bank (Xiong et al. 2021). Land use and cover (i.e., cropland, forest, wetland, and urban land) in the above 5-km riparian zone was classified using the free 10-m resolution images of global land cover from 2017 (FROM-GLC10-2017 V0.1.3) (Gong et al. 2019). These riparian zones were created using the buffering tool in ArcMap 10.7 software (ESRI, USA). More detailed information concerning the acquisition of the riparian land use data is available in Xiong et al. (2021).

P-IBI assessment system establishment

Site classification

Our sampling sites were divided into two groups—reference and impaired. This categorization is the foundation for conducting biological integrity assessments and will directly affect the IBI results (Detenbeck and Cincotta 2008; Zhu et al. 2021). In principle, the selection criterion for reference sites was to identify sampling segments that were minimally disturbed by humans and contained optimal physical, chemical, and biological conditions (Karr 1991; Ruaro and Gubiani 2013). There are many methods for the selection of reference sites in IBI studies for river health assessments (e.g., Stoddard et al. 2006; Liu et al. 2017). Collectively, these methods entail all or some combination of (1) river segments not affected by human disturbance, (2) comparison to other rivers classified as in "good" condition (Tan et al. 2015), (3) comparison to historical conditions of the river (Liu et al. 2017; Yang et al. 2020), and (4) the current optimal conditions that exist within the river (Tan et al. 2017; Zhang et al. 2020; Feng et al. 2021; Lin et al. 2021; Zheng et al. 2023). However, the middle and lower reaches of the Yangtze River basin are highly populated, intensively developed, and highly exploited, thus, it is not possible to find a natural river segment completely free from human influences to represent reference conditions. As the largest in China, the Yangtze River is regionally unique, which creates numerous challenges for identifying other rivers with similar conditions that could serve as valid references (Chen et al. 2017a). The upper reach of the Yangtze River suffers from cascading dams and other human disturbances, which is fundamentally different from the highly urbanized middle and lower reaches in terms of geographic conditions, hydrology, and water quality conditions. This variety of conditions and differing degrees of human influence creates many obstacles for providing appropriate reference sites (Gippel et al. 2017; Xiong et al. 2022). However, combining the above considerations with data accessibility, we chose to use sites reflecting the best current conditions of the river to serve as our reference sites. Under this approach, we felt it would be easier to avoid selecting severely impaired segments as reference sites, which would have resulted in highly biased assessments (Liu et al. 2017). Additionally, we felt this selection criteria would yield a reasonably unbiased river health assessment.

In selecting reference sites, we combined the physical habitat assessment, the comprehensive water quality index method (Liu et al. 2013; Zhang et al. 2019), and the phytoplankton diversity index for each site into a fourfold screening criteria matrix. First, there had to be no visible external pollution within a certain range of the segments, and the entirety of the segment had to be at least 10-km away from sewage outlets (Lin et al. 2021). Second, the river habitat evaluation score had to be greater than or equal to 120 (Lu 2020). Similarly, vegetation coverage of the riparian zone also had to greater than or equal to 70%, with the shoreline being little developed and/or utilized with no ports or docks (Lu 2020). Third, the water quality index score had to be less than 3.0 using the calculation method of the comprehensive water quality index (Zhang et al. 2019). Specifically, four water quality parameters, TP, NH_4-N, COD, and DO were selected to calculate the comprehensive water quality index, with the standard values used from the China Surface Water Environmental Quality Standards (GB 3838-2002) (Hu et al. 2022). With this particular index, lower comprehensive water quality index scores reflect better water quality. Fourth, the Shannon–Wiener diversity index for phytoplankton had to be greater than 2.0 for the diversity to be considered as "good" (Zhu et al. 2021).

Biological metrics screening

By referring to recent P-IBI studies, a total of 27 widely used metrics were selected as candidate biological metrics from four categories based on density, diversity index, biomass, and trophic status (Tan et al. 2017; Zhang et al. 2020; Feng et al. 2021; Zhu et al. 2021; Hu et al. 2022; Table 1). The 27 candidate metrics were subjected to discriminant ability test, redundancy test, and variability analysis, with the final remaining metrics used to establish the P-IBI assessment system (Wang et al. 2005; Wu et al. 2012a; Zhang et al. 2019; Hu et al. 2022).

The discriminant ability test was defined as the degree of overlap between the boxes (i.e., the 25th and 75th percentile) in box plots of metric values at reference and impaired sites (Wang et al. 2005; Wu et al. 2012a). Metrics with no overlap between the boxes of the box plots of the reference and impaired sites and those with overlap but with median values not in the other box were retained for the next analysis (Wu et al. 2012a). Metrics with overlap between the boxes of the box plots of reference and impaired sites and with median values in the other box were then excluded (Wu et al. 2012a). Redundancy analysis was performed to assess the redundancy between metrics using Pearson correlation analyses. All metrics with Pearson correlation coefficients $|r| \leq 0.70$ were automatically retained. However, when two metrics had $|r| > 0.70$ and contained similar ecological information, only one metric was retained for further analyses (Feng et al. 2021). Finally, metrics with coefficients of variation (CV) < 1.0 were retained because of the large deviations of metrics with CV > 1.0 (Wang et al. 2005; Wu et al. 2012a; Zhang et al. 2020).

Scoring and evaluation criteria for the P-IBI screened metrics

The ratio method was used to standardize the screened metrics (Feng et al. 2021; Zhu et al. 2021; Hu et al. 2022). For indicators negatively correlated with human disturbance (e.g., water pollution, habitat modifications, and other activities affecting river health), the best value was considered to be the 95th percentile of all sites, with the indicator score calculated as: indicator score = indicator value/best value. For indicators positively correlated with disturbance, the best value was considered to be the 5th percentile of all sites, with the indicator score calculated as: indicator score = (maximum value − indicator value)/(maximum value − best value) (Feng et al. 2021; Zhu et al. 2021). The indicator scores ranged from 0 to 1 and were scaled to 1 when they calculated to greater than 1 (Zhu et al. 2021; Hu et al. 2022). The sum of the screened indicator scores was equivalent to the P-IBI score for the site, with the P-IBI score for each river segment being the average P-IBI score across its three sampling sites.

To facilitate comparison of the P-IBI results between the two seasons, the P-IBI scores were processed according to Hu et al. (2022) to obtain the final adjusted P-IBI score (i.e., P-IBI final score = P-IBI raw score/95th percentile). The 95th percentile of the final score of the P-IBI for both quarters was used as a criterion for ecological health evaluation. This 95th percentile was divided into four equal parts, corresponding to the five evaluation levels of the P-IBI: results greater than the 95th percentile were rated as "excellent", with the other four levels scaled to rate as "good", "fair", "poor" and "extremely poor", respectively (Yang et al. 2020; Feng et al. 2021; Zhu et al. 2021; Hu et al. 2022).

Data analysis

Seasonal differences in P-IBI values were analyzed using one-way ANOVA. For mean separation purposes, the least significant difference test was performed when the variances were homogeneous whereas the non-parametric Games-Howell test was used when variances were not homogeneous (Qu et al. 2020). Data processing was completed using IBM SPSS Ver. 24 software. To quantify the combined and separate effects of land use and water

Table 1 Candidate metrics of phytoplankton index of biotic integrity (P-IBI) for the middle and lower reaches of the Yangtze River, China

Category	Candidate metric	Code	Expected response to disturbance	References
Density	Number of total species	M1	Decreased	Tan et al. (2017), Feng et al. (2021), Zhu et al. (2021), Hu et al. (2022)
	Number of Cyanophyta species	M2	Decreased	Feng et al. (2021), Zhu et al. (2021), Hu et al. (2022)
	Number of Chlorophyta species	M3	Decreased	Feng et al. (2021), Zhu et al. (2021), Hu et al. (2022)
	Number of Bacillariophyta species	M4	Decreased	Feng et al. (2021), Zhu et al. (2021), Hu et al. (2022)
	Cyanophyta species %	M5	Decreased	Feng et al. (2021), Zhu et al. (2021), Hu et al. (2022)
	Chlorophyta species %	M6	Decreased	Feng et al. (2021), Zhu et al. (2021), Hu et al. (2022)
	Bacillariophyta species %	M7	Decreased	Tan et al. (2017), Feng et al. (2021), Zhu et al. (2021), Hu et al. (2022)
	Total density	M8	Increased	Tan et al. (2017), Feng et al. (2021), Zhu et al. (2021), Hu et al. (2022)
	Cyanophyta density	M9	Decreased	Feng et al. (2021), Zhu et al. (2021), Hu et al. (2022)
	Chlorophyta density	M10	Decreased	Feng et al. (2021), Zhu et al. (2021), Hu et al. (2022)
	Bacillariophyta density	M11	Decreased	Feng et al. (2021), Zhu et al. (2021), Hu et al. (2022)
	Dominant species density %	M12	Increased	Zhu et al. (2021), Hu et al. (2022)
	Cyanophyta density %	M13	Increased	Feng et al. (2021), Zhu et al. (2021), Hu et al. (2022)
	Chlorophyta density %	M14	Increased	Feng et al. (2021), Zhu et al. (2021), Hu et al. (2022)
	Bacillariophyta density %	M15	Decreased	Tan et al. (2017), Feng et al. (2021), Zhu et al. (2021), Hu et al. (2022)
	Total density composed of Cyanophyta and Chlorophyta %	M16	Increased	Zhang et al. (2020)
	Total density composed of Bacillariophyta and Chlorophyta %	M17	Decreased	Tan et al. (2017), Zhang et al. (2020)
Diversity index	Shannon–Wiener index	M18	Decreased	Tan et al. (2017), Feng et al. (2021), Zhu et al. (2021), Hu et al. (2022)
	Margalef index	M19	Decreased	Tan et al. (2017), Feng et al. (2021), Zhu et al. (2021), Hu et al. (2022)
	Pielou index	M20	Decreased	Tan et al. (2017), Feng et al. (2021), Zhu et al. (2021), Hu et al. (2022)
	Simpson index	M21	Decreased	Zhu et al. (2021), Hu et al. (2022)
Biomass	Total biomass	M22	Increased	Tan et al. (2017), Zhu et al. (2021)
	Top three dominant species biomass	M23	Increased	Zhu et al. (2021)
	Cyanophyta biomass	M24	Increased	Zhu et al. (2021)
	Chlorophyta biomass	M25	Increased	Zhu et al. (2021)
	Bacillariophyta biomass	M26	Decreased	Zhu et al. (2021)
Trophic status	Diatom quotient	M27	Increased	Tan et al. (2017), Zhu et al. (2021)

quality on P-IBI, a Variance Partitioning Analysis (VPA) was performed within RDA using the "varpart" function in the "vegan" package. In this analysis, the variance was divided into multiple components based on adjusted R^2 values for land use, water quality, and residuals, with an ANOVA used to test the significance of each group of environmental variables. Pearson correlation analysis was used to determine the relationship between P-IBI and its component metrics with water quality and land use during different seasons. Results were presented using heat maps generated from the 'corrplot' and 'paletteer' packages in R (R Core Team 2020). Before the analysis, $\log(X+1)$ transformation was performed for all environmental parameters, with the significance level for all statistical analyses set at $P \leq 0.05$.

Results

Construction of the P-IBI

The reference sites were screened using the criteria outlined in "Site classification" section above during both seasons. This process yielded Yueyang (S6) and Tuanfeng (S11) during the wet season, and

Tuanfeng (S11) and Tongling (S19) during the dry season as appropriate references sites (Additional file 1: Table S3). Five metrics passed the discriminative ability test during the wet season (refer to Table 1), including number of Cyanophyta species (M2), Bacillariophyta density (M11), total biomass (M22), top three dominant species biomass (M23), and Bacillariophyta biomass (M26) (Additional file 1: Fig. S1). Fourteen metrics were passed during the dry season (refer to Table 1), including number of total species (M1), number of Cyanophyta species (M2), number of Chlorophyta species (M3), Cyanophyta species % (M5), Bacillariophyta species % (M7), total density (M8), Cyanophyta density (M9), Chlorophyta density (M10), Cyanophyta density % (M13), total density composed of Bacillariophyta and Chlorophyta % (M17), Margalef index (M19), total biomass (M22), Cyanophyta biomass (M24), and Chlorophyta biomass (M25) (Additional file 1: Fig. S2). The redundancy test deleted metrics with Pearson correlation coefficient $|r| > 0.70$ whereas two (M2 and M22) and six (M1, M5, M17, M22, M24, and M25) metrics with low correlation coefficients were retained from the wet and dry seasons, respectively (Additional file 1: Tables S4, S5). Because M24 and M25 contained CVs > 1 during the dry season, we also deleted these two indicators. In the end, the final P-IBI evaluation system included two (M2 and M22) and four (M1, M5, M17, and M22) metrics from the wet and dry seasons, respectively (Additional file 1: Table S6).

The score of each indicator of the P-IBI in the middle and lower reaches of the Yangtze River was calculated based on the response of the indicator to environmental disturbance (Additional file 1: Table S6). Total biomass (M22) was positively correlated with disturbance, and the 5th percentile of this indicator at all sites was used as the best value to calculate indicator scores. Number of total species (M1), number of Cyanophyta species (M2), Cyanophyta species % (M5), and total density composed of Bacillariophyta and Chlorophyta % (M17) were negatively correlated with disturbance, and the 95th percentile of these indicators at all sites were used as best values for calculation of indicator scores. In addition, the 95th percentile of the P-IBI final scores across the two seasons were used as the criteria for health evaluation. Using this approach, the assessment grades of the ecological health in the middle and lower reaches of the Yangtze River during wet and dry seasons were as follows: the P-IBI final score ≥ 1.00 as "excellent", 0.75 ≤ the P-IBI final score < 1 as "good", 0.50 ≤ the P-IBI final score < 0.75 as "fair", 0.25 ≤ the P-IBI final score < 0.50 as "poor", and the P-IBI final score < 0.25 as "extremely poor".

P-IBI health evaluation results

P-IBI values in the middle and lower reaches of the Yangtze River ranged from 0.27 to 1.08 during 2017–2018 (Additional file 1: Table S7). The mean P-IBI was 0.86, which indicated that the overall evaluation result was "good" during the study period. Temporally, mean P-IBI during wet and dry seasons was 0.82 and 0.91, respectively, with both values indicating "good" ecological health. One-way ANOVA results indicated that the overall P-IBI from the dry season was significantly greater than that from the wet season ($P < 0.05$) (Fig. 2).

The two seasons monitored exhibited different spatial patterns with P-IBI values (Fig. 3). During the wet season, two sampling segments (Huangshi [S12] and Pengze [S16]) rated as "excellent", nine segments (Zhijiang [S2], Shishou [S4], Luchengzhen [S7], Honghu [S9], Xinzhou [S14], Wuwei [S20], Wuhu [S21], Maanshan [S22], and Jingjiang [S27]) rated as "fair", the Zongyang (S18) segment rated as "poor", and the remaining 18 segments rated as "good" (Fig. 3a). Additionally, percentages of sampling segments rated as "excellent", "good", "fair" and "poor" ecological health were 7%, 60%, 30%, and 3%, respectively, during the wet season. During the dry season, two sampling segments (Tongling [S19] and Tongjingzhen [S24]) rated as "excellent", with the remaining 23 segments rating as "good" (Fig. 3b). Similarly, percentages of segments rated as "excellent" and "good" ecological health were 8% and 92%, respectively, during the dry season.

Relationships between P-IBI and environmental factors

The variance partitioning results indicated that water quality parameters in conjunction with land use played the greatest role in shaping spatiotemporal patterns of P-IBI. During the wet season, the pure water quality

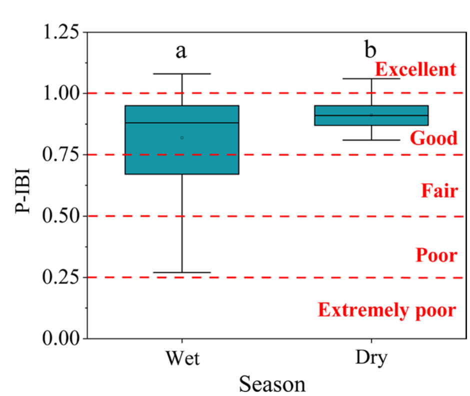

Fig. 2 Seasonal P-IBI patterns in the middle and lower reaches of the Yangtze River during 2017–2018. Box plots with different lowercase letters indicate a statistically significant difference ($P < 0.05$) between seasons

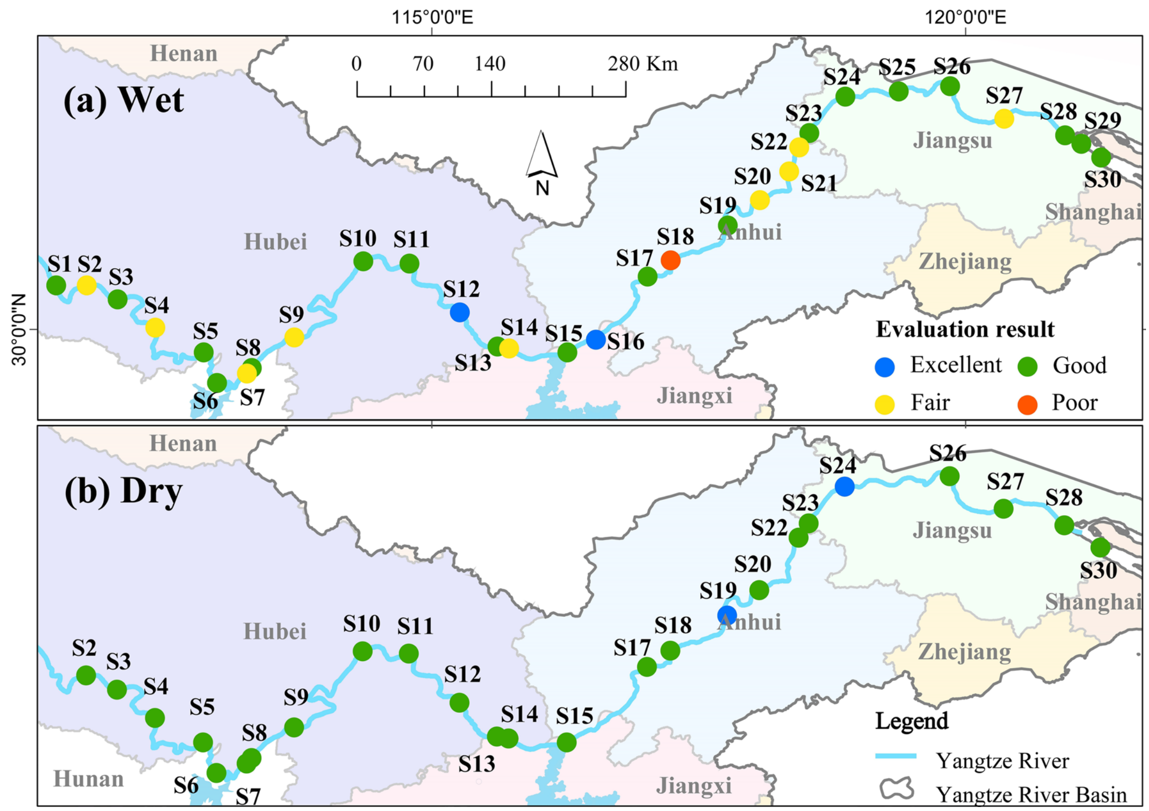

Fig. 3 Spatial P-IBI patterns of the **a** wet and **b** dry seasons in the middle and lower reaches of the Yangtze River during 2017–2018

effect (31%) contributed the most, followed by the combined effects of water quality and land use (16%), and then the pure effect of land use (3%) (Fig. 4a). During the dry season, the pure water quality effect (21%) was the largest contributor followed by the pure land use effect (6%) (Fig. 4b).

Pearson correlation analysis indicated that P-IBI and its component metrics were significantly correlated with both water quality and land use parameters ($P<0.05$). However, there were more environmental variables significantly correlated with P-IBI during the wet season (Fig. 5). Specifically, TP, NO_3-N, TSS, TUR, CON, and DO exhibited significant negative relationships with P-IBI ($P<0.05$) while forest and urban land use exhibited significant positive relationships with P-IBI ($P<0.05$) during the wet season (Fig. 5a). During the dry season, TN, NH_4-N, and NO_2-N exhibited significant positive relationships with P-IBI ($P<0.05$) while cropland and WT exhibited significant negative relationships with P-IBI ($P<0.05$) (Fig. 5b).

Discussion

Spatiotemporal variations of P-IBI

The assessment results of P-IBI suggested there were seasonal variations of the ecological health in the middle and lower reaches of the Yangtze River during 2017–2018, with the level of health generally being better during the dry season ($P<0.05$). Zhu et al. (2021) also reported that P-IBI from Ge Lake, Jiangsu Province was greater during the dry season, presumably due to the increase in cyanobacteria and decline in water quality induced by the greater summer temperatures. However, findings from other studies have suggested that ecological health tends to be poorer during dry seasons (Zhang et al. 2020; Lin et al. 2021; Hu et al. 2022). Lin et al. (2021) suggested that the better ecological health reflected by the P-IBI in wet seasons might be due to greater precipitation and river flows at those times of year. In another study, Wang et al. (2014) observed a positive relationship between river water quality and monthly precipitation. Purportedly, increased river flows associated with seasonal rainfall generally improved water quality by diluting pollutants, which resulted in better ecosystem health (Cheng et al. 2018). The Yangtze River basin has a subtropical monsoon climate with greater temperatures and rainfall typically occurring during summers (i.e., the wet season), with lower temperatures and less rainfall more common during winters (i.e., the dry season) (Zhang et al. 2007). In fact, this study did experience greater precipitation during the wet season (Additional file 1: Table S8).

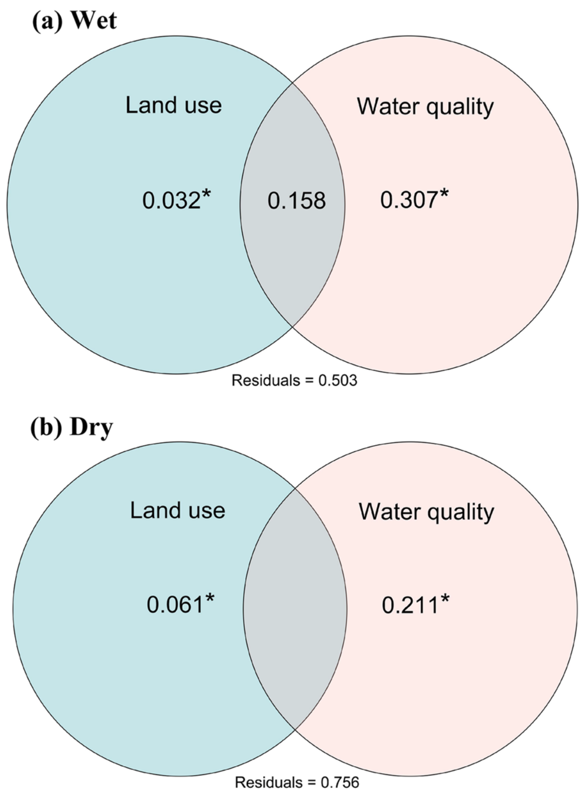

Fig. 4 Results of variance partitioning for different drivers (i.e., land use and water quality) of P-IBI in the middle and lower reaches of the Yangtze River in 2017–2018 for **a** the wet season and **b** the dry season. Values in the circles indicate the amount of variation in P-IBI explained independently or jointly by land use and water quality parameters. All scores (*$P < 0.05$) are based on adjusted R^2 values, with residuals shown below and negative values not shown

Furthermore, this pattern was considered typical, as annual precipitation totals and wet-season and dry-season precipitation totals were similar in 2017–2018 compared to a recent period spanning 2010–2016 (Additional file 1: Table S8). Thus, precipitation amounts during our study years appeared to be fairly typical for lower and middle portions of the Yangtze River basin.

Given that the region's hydrological processes are largely precipitation-driven, peak nutrient discharges in the Yangtze River often correspond with precipitation (Xiong et al. 2022). Rainfall runoff brings nutrients, organic matter, heavy metals, and other pollutants from the watersheds into the river proper (Ren et al. 2008; Kuang 2012; Tong et al. 2017; Chen et al. 2019). While increasing river flows, abundant precipitation during summers also increases diffusion of pollutants within the river, which can directly increase the TN and TP concentrations (Xu et al. 2019; He et al. 2021). While assessing the water quality of the middle and lower Yangtze River, Xiong et al. (2022) reported the overall water quality condition from WQI (i.e., water quality index) scores was significantly greater during dry seasons compared to wet seasons, which is consistent with the P-IBI results in the current study.

The assessment of P-IBI in the middle and lower reaches of the Yangtze River also exhibited spatial variation, which was similar to previous water quality assessments in the river (Xiong et al. 2022; Wu et al. 2023). A total of 10 river sections (S2, S4, S7, S9, S14, S18, S20, S21, S22, and S27) contained fair to poor P-IBI values during the wet season. The main driver for low wet-season P-IBI values in these segments was the low M2 score (i.e., number of Cyanophyta species). Water quality parameters and P-IBI composition indicators are usually highly correlated (Hu et al. 2022). In fact, we detected a negative correlation between M2 (number of Cyanophyta species) and several water quality parameters (TP, NO_3-N, TSS, TUR, CON and DO) during the wet season (Fig. 5a). In particular, the Luchengzhen site (S7) and six more downstream river segments (S9, S14, S18, S20, S21, and S22) all contained elevated TP, TUR, TSS, and NO_3-N levels (Xiong et al. 2022). These water quality characteristics combined resulted in lower M2 values in these river segments, which affected the P-IBI values. Water quality parameters such as nutrients and pH often have significant effects on cyanobacterial richness (Xie et al. 2012; Liao et al. 2016). Greater TUR and TSS levels might have also reduced sunlight penetration into the water column, which would have affected phytoplankton growth, and ultimately, phytoplankton diversity (Ding et al. 2021, 2022a).

Relationships between P-IBI and environmental factors

In the current study, both water quality parameters and land use appeared to play important roles in shaping spatial and temporal patterns of the P-IBI in the middle and lower reaches of the Yangtze River. Zhu et al. (2021) and Hu et al. (2022) also reported water quality parameters to be highly correlated with P-IBI and its constituent metrics. These studies also reported that P-IBI varied significantly between wet and dry seasons, likely due to different hydrological conditions.

The current study also found P-IBI to be negatively correlated with several water quality parameters (e.g., TP, NO_3-N, TSS, TUR, CON, DO) during the wet season. Turbidity is an important environmental factor affecting phytoplankton (Tian et al. 2021). For instance, elevated turbidity reduces light transmission in the water column, which in turn, modifies phytoplankton community structure (Ding et al. 2022b). Conductivity indicates inorganic enrichments in the environment, which can induce changes in phytoplankton species diversity (Flores and Barone 1998). During the dry season, P-IBI was positively

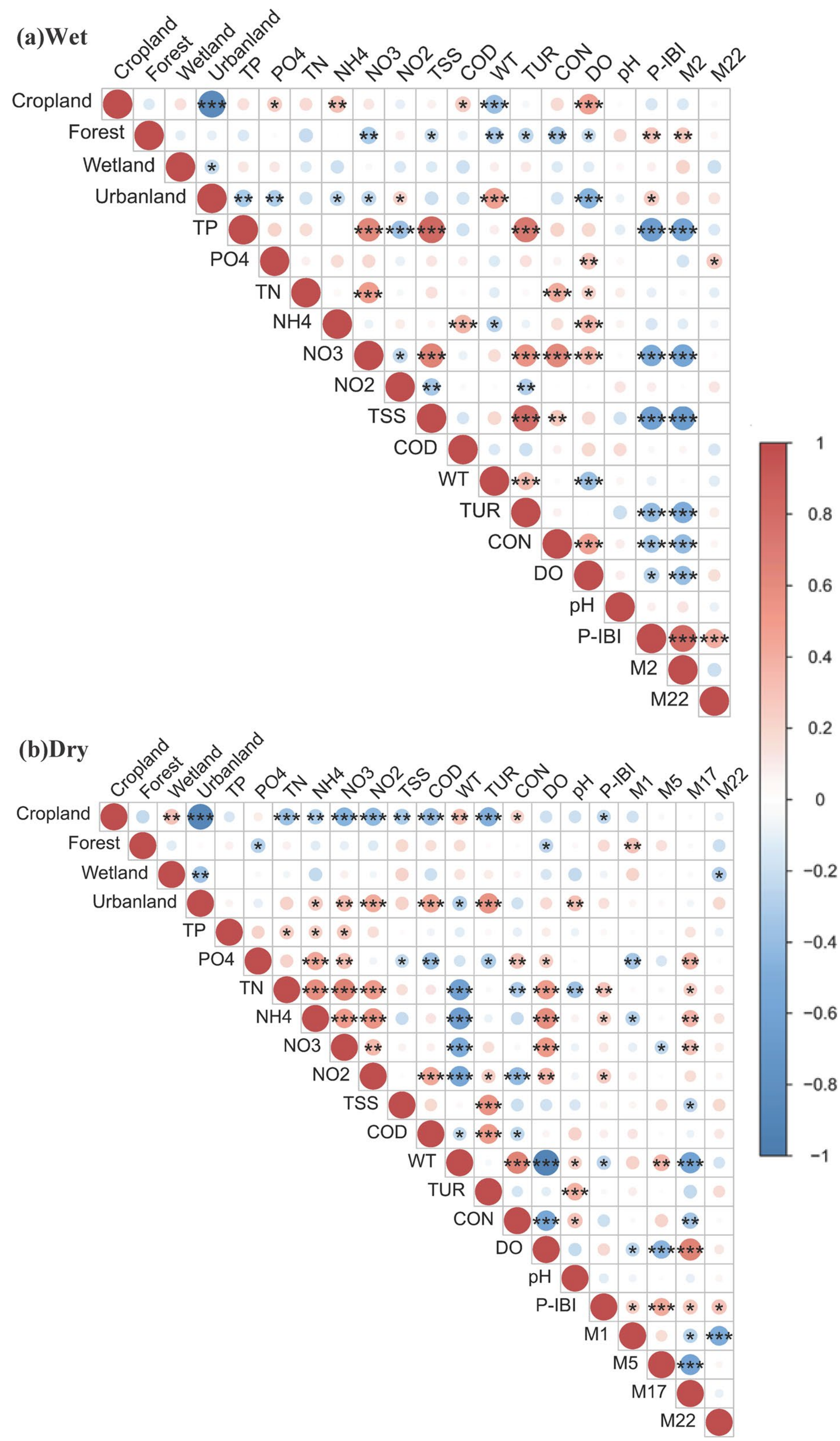

Fig. 5 Pearson correlations among P-IBI, water quality parameters, and land use of the **a** wet and **b** dry seasons in the middle and lower reaches of the Yangtze River during 2017–2018. *$P<0.05$, **$P<0.01$, ***$P<0.001$

correlated with nitrogenous nutrients. Although nitrogen is an important water nutrient and often a limiting factor for phytoplankton growth, most phytoplankton prioritize the uptake of NH_4-N (Dortch 1990). However, increases in TN also can promote the dominant phytoplankton species to some extent (Shetye et al. 2019), thus, altering the phytoplankton community structure (Liu et al. 2019; Li et al. 2022).

The intensity of land use also is an important driver of riverine ecosystem health (Cheng et al. 2018; Xiong et al. 2023). Land use changes occur at large spatial scales, with the resulting decline in water quality purported to be an important driver of phytoplankton growth (Katsiapi et al. 2012; Peng et al. 2021). In the current study, P-IBI exhibited a positive relationship with the forest and urban land use during the wet season. Forests tend to be catchment areas for potential pollutants in the water, and riparian vegetation buffers usually have filtration and barrier effects on pollutants (Chen et al. 2016b), all of which play an important role in improving water quality in the receiving stream (Mello et al. 2018). Wu et al. (2021) reported that the correlation between urban land use and pollutants (i.e., NH_4-N and COD) was stronger during dry seasons compared to wet seasons, which was consistent with the results of the current study. With the wet season being the flood season in the Yangtze River basin, increased river flows from seasonal rainfall, in effect, dilutes pollutants, which may have somewhat buffered the negative impacts of urban land use on the ecological health of the Yangtze River (Xiong et al. 2022). During the dry season, P-IBI was negatively related to croplands, which was attributed to the fact that croplands affect river health by increasing potential inputs of nonpoint source pollutants such as phosphorus and nitrogen (Allan 2004; Cheng et al. 2018).

Limitations and recommendations

There were some limitations in the current study that should be considered for future studies. For instance, different phytoplankton sampling methods can affect community composition and diversity (Wu et al. 2011; Jiang et al. 2020), which could then affect the eventual P-IBI assessment. To insure consistency with previous studies (e.g., Zhang et al. 2018), we used the classical sedimentation method for quantifying phytoplankton samples in the current study. However, future studies might consider the use of phytoplankton nets or similar gears for sample collections. Although sedimentation methods as used here are generally more effective at retaining smaller nanoplankton taxa than many nets (e.g., Kraatz 1940), they generally reflect qualitative community features similar to other gears (Zagorski and Walz 1973). However, some studies (e.g., Hallegraeff 1977) recommend that gear choice should be dictated by the measures of interest (e.g., total seston dry weight, total particle volume, or chlorophyll-a concentration).

Because various regional IBI systems have differing degrees of specificity and uniqueness, it may be necessary to consider the actual environmental conditions of the study area when attempting their use across large spatial scales (Gippel et al. 2017). Thus, reference sites should be selected carefully and objectively, with a comprehensive analysis of the actual conditions of the study area in order to limit the degree of subjectivity in reference site selection (Gippel et al. 2017; Liu et al. 2017).

To improve the accuracy of P-IBI health assessments, the river's ecological health should be comprehensively assessed across multiple seasons (Zhu et al. 2021). The current study used wet and dry seasons during 2017–2018. However, monitoring assessments based on short-term datasets could have limitations in reflecting the river's actual ecological health across different hydrological periods. Including a robust seasonal component to river health assessments should help remedy this shortcoming.

Finally, the ecological health of all rivers is impacted by many factors, including the number and magnitude of upstream dams, navigation practices (e.g., channelization, dredging, and bank revetments), sand mining, and various types of habitat modifications (Allan 2004; Li et al. 2013; Chen et al. 2020; Feng et al. 2021; Lin et al. 2021). It was not possible to incorporate all of these factors into our statistical analyses in the current study. Thus, longer-term monitoring and evaluation of rivers with multiple influencing factors should be considered in the future to build a more robust P-IBI assessment scheme for large rivers.

Conclusions

In the current study, we used the P-IBI to assess the ecological health of the middle and lower reaches of the Yangtze River during both wet and dry seasons. We reached the following conclusions.

(1) Seasonal and spatial variations in ecological health were observed in the middle and lower reaches of the Yangtze River, with overall better ecological condition reflected during the dry season.
(2) Both water quality and land use appeared to significantly influence the P-IBI. Overall, water quality played a larger role than land use in driving the P-IBI.
(3) It is recommended that any comprehensive evaluation of large-river ecological health based on P-IBI should incorporate seasonal effects to generate a

more robust assessment of the river's actual ecological health.

Abbreviations

IBI Index of biotic integrity
P-IBI Phytoplankton-based index of biotic integrity
GDP Gross domestic product
WWF World Wildlife Fund
WT Water temperature
CON Conductivity
DO Dissolved oxygen
TUR Turbidity
TN Total nitrogen
NH_4-N Ammonium
NO_3-N Nitrate
NO_2-N Nitrite
TP Total phosphorus
PO_4-P Orthophosphate
TSS Total suspended solids
COD Chemical oxygen demand
CV Coefficient of variation
WQI Water quality index

Supplementary Information

The online version contains supplementary material available at https://doi.org/10.1186/s13717-023-00456-7.

Additional file 1: Table S1. Site names, codes, and locations in the middle and lower Yangtze River. **Table S2.** Monthly average water flow (m^3/s) in wet and dry seasons of 2017–2018 and annual runoff (billions m^3) during 2017–2018 and historical periods in the middle and lower Yangtze River. **Table S3.** Biological and environmental conditions at reference sites in the middle and lower reaches of the Yangtze River during the wet and dry seasons of 2017–2018. **Table S4.** Pearson's correlation coefficients between candidate phytoplankton metrics in the middle and lower reaches of the Yangtze River in the wet season during 2017–2018. **Table S5.** Pearson's correlation coefficients between candidate phytoplankton metrics in the middle and lower reaches of the Yangtze River during the dry season during 2017–2018. **Table S6.** P-IBI component metrics and score calculation for the wet and dry seasons in the middle and lower reaches of the Yangtze River during 2017–2018. **Table S7.** P-IBI results in the wet and dry seasons of the middle and lower reaches of the Yangtze River during 2017–2018. **Table S8.** Precipitation (mm) in sites along the middle and lower reaches of the Yangtze River during 2010–2018. **Figure S1.** Box plots of 5 candidate metrics between the reference and impaired sites in the wet season. Note: (a) M2, number of Cyanophyta species; (b) M11, Bacillariophyta density; (c) M22, total biomass; (d) M23, top three dominant species biomass; (e) M26, Bacillariophyta biomass. **Figure S2.** Box plots of 14 candidate metrics between the reference and impaired sites in the dry season. Note: (a) M1, number of total species; (b) M2, number of Cyanophyta species; (c) M3, number of Chlorophyta species; (d) M5, % Cyanophyta species; (e) M7, % Bacillariophyta species; (f) M8, total density; (g) M9, Cyanophyta density; (h) M10, Chlorophyta density; (i) M13, % Cyanophyta density; (j) M17, % total density composed of Bacillariophyte and Chlorophyta; (k) M19, Margalef index; (l) M22, total biomass; (m) M24, Cyanophyta biomass; (n) M25, Chlorophyta biomass.

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Author contributions
YC designed the study. WG, FX, and YL collected the data. WG analyzed the data. WG, FX, YL, XQ, WX and YC wrote the manuscript. All authors contributed to revising, commenting, and finalizing the manuscript.

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Availability of data and materials
All data will be available in the data center of Institute of Hydrobiology, Chinese Academy of Sciences (https://www.ihb.ac.cn).

Declarations

Ethics approval and consent to participate
Not applicable.

Consent for publication
Not applicable.

Competing interests
The authors declare that they have no competing interest.

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