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Original Articles

Evaluating the ecological health of aquatic habitats in a megacity through a multimetric index model based on macroinvertebrates

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ABSTRACT

Globally, urban water bodies suffer from a variety of ecological pressures which profoundly change freshwater ecosystem services and pose a great threat to aquatic biodiversity. To identify the effects of these pressures on aquatic communities, we systematically investigated the macroinvertebrates in different water types (e.g., mountain rivers, plain rivers, lakes, and reservoirs) in all water systems in Beijing, a megacity in China, to reveal the key environmental factors and response mechanisms affecting the spatial distribution of macroinvertebrate communities. A total of 188 macroinvertebrate taxa were identified at 61 survey sections, and environmental factors such as flow velocity, water depth, water temperature, and total nitrogen content were found to substantially affect the structure and spatial distribution of the macroinvertebrate communities. A multimetric index (MMI) model based on macroinvertebrates was developed to assess the ecological quality of each water type, and the developed MMI was demonstrated to be widely applicable. In the MMI, each community metric was weighted based on the goodness of fit for each biological metric and environmental metric to obtain the observed MMI values of the measured sample sites, and further model training and prediction was performed based on all sample site data. The MMI results revealed that the overall ecological quality of mountain rivers with less anthropogenic interference was relatively good (MMI = 0.62 ± 0.28), the overall ecological quality of lakes experiencing ecological disturbance and undergoing ecological restoration practices was moderate (MMI = 0.43 ± 0.09), and the overall ecological quality of plain rivers and reservoirs with strong anthropogenic interference was relatively poor (MMI = 0.24 ± 0.12 and 0.32 ± 0.12, respectively). Specific recommendations for ecological protection of different water types were formulated, providing a scientific basis and decision-making support for urban ecological planning and sustainable development.

1. Introduction

Urbanization is one of the most important anthropogenic processes driving ecosystem and biodiversity changes (Grimm et al., 2008). Globally, the growing percentage of people living in cities, which currently exceeds 55%, is accompanied by large-scale land modifications that can lead to disruption of natural habitats and loss of biodiversity (McKinney, 2002). Urbanization often affects biodiversity in a complex and widespread manner, and its associated impacts are strongly dependent on taxa and spatiotemporal scale (Piano et al., 2020; Zhang et al., 2022). For instance, recent studies of terrestrial invertebrates, birds, and mammals demonstrated that biodiversity responds to urbanization interference to a certain extent following the intermediate disturbance hypothesis; that is, species richness can peak in areas of intermediate urbanization relative to that in areas of low and high urbanization (Batáry et al., 2018; Concepción et al., 2015; Parsons et al., 2018).

Aquatic habitat changes associated with urban development, including habitat reduction and fragmentation, hydrology modifications, reduced vegetation cover, channelization, pollutant enrichment, and other anthropogenic disturbances, can all alter aquatic community structure (Geist and Hawkins, 2016; Grimm et al., 2008). As key primary consumers in aquatic communities, macroinvertebrates play a fundamental role as regulators of material and energy flows and system stability in freshwater ecosystems, and they are used widely as biological indicators to assess water quality and ecosystem integrity (Yao et al.,

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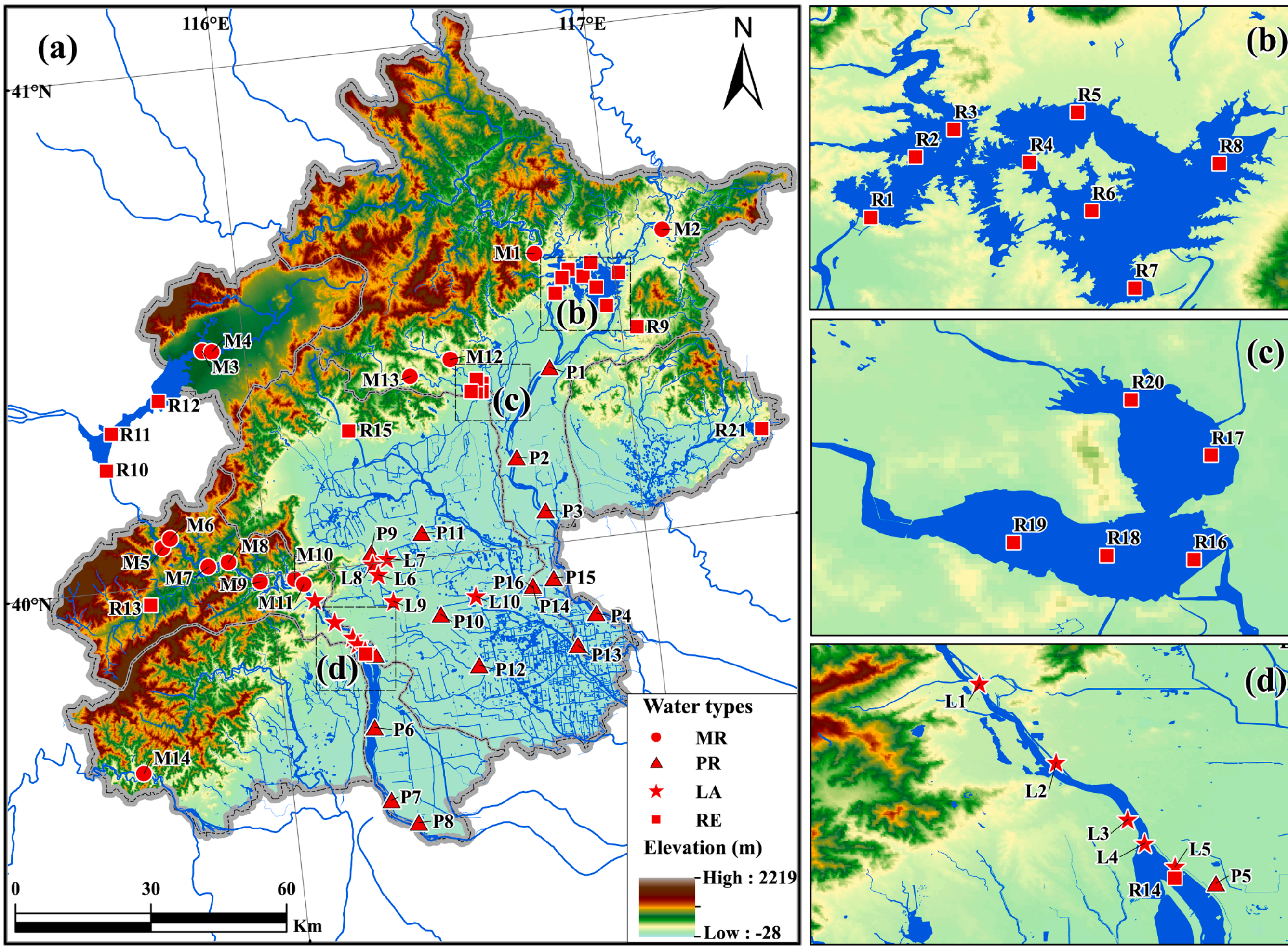

Fig. 1. (a) General distribution of mountain river (MR), plain river (PR), lake (LA), and reservoir (RE) sampling sites in Beijing. Detailed distribution of sampling sites of (b) the Miyun reservoir, (c) the Huairou reservoir, and (d) the midstream section of the Yongding River.

2022; Zhou et al., 2019). Studies have demonstrated that significant negative correlation exists between macroinvertebrate species richness and urbanization (Fitzgerald et al., 2012; Smucker and Detenbeck, 2014), and that increased human activity is gradually threatening macroinvertebrate diversity and integrity (Barnum et al., 2017; White and Walsh, 2020). Additionally, urbanization often prioritizes economic, recreational, ornamental, or water conservation purposes, resulting in reduced heterogeneity of aquatic habitats and indirectly accelerating homogenization of macroinvertebrate communities (Barnum et al., 2017; He et al., 2022; McGoff et al., 2013). In many metropolitan cities, such as Melbourne, Australia (Walsh et al., 2001), Boston and Salt Lake City, USA (Cuffney et al., 2010), and multiple cities in Denmark, Sweden, and Germany (McGoff et al., 2013), the macroinvertebrates of urban water bodies have undergone homogenization. Despite extensive ecological restoration of urban rivers in some countries and regions (Carlson et al., 2018; Feio et al., 2022), systematic and effective assessment of such restoration programs is lacking.

Previous studies have developed multiple methods to assess riverine ecological quality using macroinvertebrates as indicator taxa (Feio et al., 2021; Obubu et al., 2021), and such methods have been demonstrated to be useful and widely applicable in natural freshwater ecosystems in many regions such as Europe, Africa, Asia, and South America. However, the methods adopted in those earlier studies have often been overreliant on a single influencing factor (e.g., organic or chemical pollution), and thus they cannot be applied in assessing the integrated ecological quality of urban water bodies with multiple disturbances (Kumar et al., 2022, 2023). Recently, multimetric and multivariable approaches have been developed that successfully overcome the limitations of single influencing factors (Kaboré et al., 2022; Zhou et al., 2019), and have been applied to assess the ecological quality of natural riverine systems over different geographic spaces. The primary advantage of a multimetric index (MMI) model is the possibility to integrate different biometric measures (e.g., abundance, composition, tolerance, and trophic status) into a single value that is more sensitive to different types of disturbance and is more easily used (Hering et al., 2006; Zhou et al., 2022). Moreover, each component of an MMI has a predictable relationship with a specific environmental effect (Alemneh et al., 2019; Kaboré et al., 2022), and these components can reflect the cumulative impact of various disturbances and reveal the overall situation of the monitored river segment (Hering et al., 2006). Therefore, such methods are increasingly used for actions aimed at conservation and restoration, thereby providing technical support to managers and decision makers (Barbour et al., 1995; Hering et al., 2004; Karr and Chu, 1999).

The city of Beijing in China has a long history of urbanization. With a current population of over 20 million permanent residents, it is one of the largest megacities in the world. It has undergone rapid and extensive expansion and landscape modification over the past 70 years. The city now has an abundance of artificial, seminatural, and natural freshwater ecosystems. Documentation shows that deterioration or drying up of water bodies has led to great loss of biodiversity. For example, the midstream and downstream sections of the Yongding River have dried up for 25 years, and other rivers and lakes have experienced large-scale pollution over long periods of time (Dai et al., 2021; Zhang et al., 2008). However, in recent years, with the transfer of water across basins such as the South-to-North Water Transfer Project and following a series of ecological restoration projects of rivers and lakes (e.g., the middle route of the South-to-North Water Transfer Project, the Water Diversion Project from the Yellow River to Beijing, the Chaobai River Ecological Water Replenishment Project, the Yongding River Ecological Water Replenishment Project, the Yongding River Ecological Restoration Project, and the Chaobai River Ecological Restoration Project), Beijing's freshwater bodies have been improved from a state where all rivers were dry and all water polluted, and the water quality conditions have improved substantially (Du et al., 2021; Sheng and Webber, 2021). However, the full extent of water ecosystem function restoration has yet to be assessed comprehensively.

To this end, we systematically investigated and sampled representative sections of all types of aquatic habitat in Beijing, including suburban mountain rivers and reservoirs, as well as urban plain rivers and lakes. Furthermore, we analyzed the environmental characteristics of the various types of aquatic habitat as well as both the community composition and the spatial distribution of macroinvertebrates. We hypothesized that macroinvertebrate communities in suburban mountain rivers in Beijing would have the best health, whereas other suburban and urban water bodies would be subjected to different ecological pressures and have reduced macroinvertebrate community health. On this basis, we (a) explored the response relationships of macroinvertebrate abundance to environmental variables in different types of urban aquatic habitat, (b) developed an MMI model for macroinvertebrate communities by integrating biodiversity, structural, and functional measures, and (c) investigated the response pattern of macroinvertebrate communities to environmental factors by coupling the MMI and environmental factors to assess and predict the integrated ecological quality of each type of aquatic habitat. This study provides useful tools for assessing the ecological quality of urban freshwater ecosystems, as well as methodological guidance and scientific support for water restoration practices in megacities such as Beijing.

2. Materials and methods

2.1. Sample collection and environmental factor measurement

Field surveys and sampling were conducted in all five river systems (i.e., the Yongding River, Chaobai River, Beiyun River, Jiyun River, and Daqinghe River) in Beijing (39.4°–41.6°N, 115.7°–117.4°E) (Fig. 1) from May 2019 to November 2020. The sampling sites were divided into four types of aquatic habitat (Fig. S1): mountain rivers (14 sites), plain rivers (16 sites), lakes (10 sites), and reservoirs (21 sites), according to habitat features and the requirements of the "Technical regulations for ecological health on aquatic ecosystem assessment" in relation to Beijing (Beijing Municipal Administration for Market Regulation, 2020). Based on the environmental gradient of the different sites, the sampling frequency was set to vary from 3 to 12 times per site (Table S1). The sites on mountain rivers were generally at an elevation of > 100 m. Such rivers are mainly seminatural rivers with relatively little anthropogenic interference/modification, and have a substrate of mainly gravel and pebbles. Sites on plain rivers were generally at an elevation of < 100 m. Such rivers typically have strong anthropogenic interference/modification and are often subject to channelization, and the hard river surface cover substrates are mainly pebbles, coarse sand, and fine sand. Lakes in the region are mainly urban landscape water bodies with strong anthropogenic interference, and their substrate is mainly artificially modified gravel, pebbles, and deposited mud/fine sand. The regional reservoirs are subject to relatively little anthropogenic interference/modification, and are mostly drinking water reservoirs with a substrate comprising mainly mud (see Table S1).

During sampling, a kick net with an aperture of 420 μm (with a sampling area of 1 m^2) was used for sampling sites in flowing water bodies with a water depth of < 1 m. A Peterson grab (with a sampling area of 1/16 m^2) was used for sampling water bodies with a water depth of > 2 m. A D-shaped net (with a sampling area of 1/3 m^2) was used for sections with a water depth of 1–2 m that could not be waded, and each sampling was repeated at least 3–4 times at a single site. On the day of sampling, all samples were transferred to the laboratory, washed with a 420 μm pore size sieve, and stored in 50 mL centrifuge tubes with 95% ethanol. Each individual of the macroinvertebrates was identified under a biological microscope (Leica DM2500) to at least genus level (Morse et al., 1994; Epler, 2001; Wang and Wang, 2011). Then, the surface of each individual was sucked dry with filter paper, and the specimen was placed on a scale to weigh its biomass (0.001 g).

Ten environmental variables commonly used to interpret benthic macroinvertebrate community composition were monitored (Yao et al., 2022; Zhou et al., 2019). The mean flow velocity (v) was measured using a direct current velocity meter (Runsun LS300-A, Chengdu, China). Depth (h) was recorded using a steel ruler and a marine depth gauge. Transparency (SD) was measured using a Secchi disc (Wuhan STD DC20, Wuhan, China). Water temperature (WT), pH, dissolved oxygen (DO), and other variables were measured in the field using the YSI EXO water quality monitoring platform (Xylem EXO2s, Yellow Springs, USA). Chlorophyll *a* (Chl.a) was measured using a portable chlorophyll fluorometer (Sea bird FLSB-875, Bellevue, USA). From each sample layer, 500 mL of water was collected and stored in a portable refrigerator before the sample was returned to the laboratory for testing of total nitrogen (TN) and total phosphorus (TP). TN and TP were measured according to the "Water and Wastewater Detection and Analysis Method (4th edition)" (Ministry of Ecology and Environment of the People's Republic of China, 2002). Another 2 L of water was taken from each sample layer and stored at pH < 2.0 before the sample was returned to the laboratory for testing of the permanganate index (COD_{Mn}). COD_{Mn} was tested using a multifunctional water quality detector (HACH DRB200, Loveland, USA) within 24 h after sampling (Zhang et al., 2019).

2.2. Data analysis

2.2.1. Analysis of variance

The environmental variables and macroinvertebrate community indices were tested for normality and variance homogeneity using the Shapiro method and Bartlett method (R package "stats"; R Core Team, 2021) to describe their differences among the four types of aquatic habitat. Variables tested to be normal and variance-homogenous were further analyzed using one-way analysis of variance (one-way ANOVA) to detect overall significant differences among the different types of aquatic habitat, and post hoc analysis was performed using the Tukey HSD method to detect pairs of habitat types contributing to the overall difference. Variables tested to be nonnormal or heterogenous in variance were further analyzed using the Kruskal–Wallis method for overall difference and the Bonferroni-corrected method for post hoc analysis (R package "agricolae", function "Kruskal"; de Mendiburu, 2021). Significance levels were indicated by p-values, i.e., $p < 0.05$ and $p < 0.01$.

2.2.2. Indicator taxa

Indicator taxa in each type of aquatic habitat were identified quantitatively using the indval function (R package "labdsv"; Roberts, 2016), and taxa passing the significance test at the $p < 0.05$ level were used as the indicator taxa for that aquatic habitat type. If a taxon had significant values for multiple aquatic habitat types, the aquatic habitat with the higher indicator value was taken as the aquatic habitat type indicated by that taxon (Dufrêne and Legendre, 1997):

$$\mathrm{Indval} = f_{ji} S_{ji} \quad (1)$$

$$f_{ji} = \frac{\sum_{k=1}^{n_j} p_{ki}}{m_j} \quad (2)$$

$$S_{ji} = \frac{\left(\sum_{k=1}^{n_j} n_{ki}\right)/m_j}{\sum_{l=1}^{L}\left[\left(\sum_{m=1}^{n_l} n_{mi}\right)/m_l\right]} \tag{3}$$

where f_{ji} is the fidelity of taxon i in water type j, S_{ji} is the specificity of taxon i in water type j, p_{ki} is the presence/absence (1/0) of taxon i at sample site k, m_j is the number of sample sites in water type j, n_{ki} is the abundance of taxon i at sample site k, m_j is the number of samples in aquatic habitat type j, n_{mi} is the abundance of taxon i at sample site m, m_l is the number of sample sites of water type l, and L is the number of water types.

2.2.3. Ordination

The abundance of each taxon was pretreated using the $\log_{10}(x + 1)$ transformation, and a detrended correspondence analysis (DCA) was applied to the macroinvertebrate data to investigate the response of macroinvertebrates to environmental gradients (CANOCO 4.5, Wageningen University & Research, Wageningen, Netherlands). Canonical correspondence analysis (CCA, based on unimodal response models) was conducted if the longest gradient from the DCA exceeded a threshold value (Lth = 4.0, the index indicating heterogeneity and deviation of taxa); otherwise, a redundancy analysis (RDA, based on linear response models) was performed (Lepš and Šmilauer, 2003). Then, 1000 permutation tests were performed via Monte Carlo Permutation Test analysis to identify the environmental factors that affect the spatial distribution of the macroinvertebrate community (Dou et al., 2022).

2.3. MMI development

2.3.1. Metric screening

A total of 43 macroinvertebrate community metrics were selected to describe the macroinvertebrate community structure, including biodiversity, community morphological structure, functional structure, and tolerance (Table S2). These metrics were selected based on their predictable responses to changes in environmental variables (Barbour et al., 1996, 1999; Grafe, 2002). For example, the species richness of macroinvertebrates decreases monotonically with environmental degradation. To this end, 5 metrics with unclear response patterns were removed, and the remaining 38 community metrics were normalized to values in the range of 0–1. Additionally, correlations among the macroinvertebrate community metrics were further tested using the Spearman method (R package "PerformanceAnalytics") to reduce collinearity and ensure the independence of each metric (Nagendra, 2012). Then, those metrics with $R < |0.6|$ were screened as biological metrics of the MMI, while ensuring that the integrity of the community could be fully described. The environmental metrics followed the screening process described above.

2.3.2. MMI calculation

The MMI was used as the basic framework (Plafkin et al., 1989), and combined with the above steps for parameter filtering and model development:

$$MMI = \sum_{n} w_i MIi \tag{4}$$

where MMI is a dimensionless index value between 0 and 1, and larger values indicate higher ecological quality of the site; w_i is the weight of the index; and MI_i is the dimensionless macroinvertebrate community metric value between 0 and 1 that indicates the degree of superiority or inferiority of the community metric i, with 0 indicating "very poor" and 1 indicating "excellent".

2.3.3. Metric weighting

An integrated weighting method combining subjective and objective analyses was used to quantify the weight (w_i) of each MMI (Moya et al., 2011), and the goodness of fit of the screened community metrics (MI_i) with each environmental metric was used as the basis for the weighting. A generalized additive model (GAM) (R package "mgcv") was used to construct regression relationships between the community metrics and environmental metrics (Zhou et al., 2022), and the goodness of fit of the model was used as reference to assign the weights (w_i) for each macroinvertebrate community metric. In the model, because the environmental metrics used raw data and the community metrics were standardized, "logit" (link = log) was used to link the two sets of variables for fitting (Britton, 2012). The Bayesian information criterion (BIC) was used as the basis for weighting each community metric according to the GAM optimal model selection criteria:

$$w_i = \frac{C_i}{\sum C_i} \tag{5}$$

$$C_i = \frac{(BIC_{max} - BIC_{min}) + BIC_i}{\sum_i^n (BIC_{max} - BIC_{min}) + BIC_i} \tag{6}$$

where w_i is the weight value of community metric i, C_i is the adjusted value of each community metric; BIC is the goodness-of-fit value, BIC_i is the observed BIC value of model i, BIC_{max} represents the maximum BIC value among all models, and BIC_{min} represents the minimum BIC value.

2.3.4. MMI performance

The macroinvertebrate community metric values predicted by the GAM and the measured metric values were weighted separately to quantify the performance of the MMI model, resulting in predicted MMI values and observed MMI values. In accordance with Hering et al. (2006), MMI values can be divided into five classes: 0–0.2 representing "bad", 0.2–0.4 representing "poor", 0.4–0.6 representing "moderate",

Table 1
Values (mean ± standard deviation; unnormalized) of environmental variables and macroinvertebrate metrics for each of four types of aquatic habitat.

	MR	PR	LA	RE
h (m)	0.85 ± 0.47^{RE}	1.12 ± 0.92^{RE}	1.46 ± 1.03^{RE}	$21.22 \pm 13.52^{MR, PR, LA}$
SD (m)	0.61 ± 0.36^{RE}	0.56 ± 0.48^{RE}	0.72 ± 0.44^{RE}	$2.33 \pm 1.14^{MR, PR, LA}$
v (m/s)	$0.491 \pm 0.20^{PR, LA, RE}$	0.014 ± 0.002^{MR}	0.001 ± 0.00^{MR}	0.001 ± 0.00^{MR}
WT (℃)	21.12 ± 1.66	22.75 ± 1.01^{RE}	23.19 ± 1.25^{RE}	$19.56 \pm 4.06^{PR, LA}$
DO (mg/L)	9.57 ± 1.56	$9.63 \pm 1.46^{\mathbf{RE}}$	9.62 ± 1.3	$8.47 \pm 1.13^{\mathbf{PR}}$
pH	8.53 ± 0.23	8.62 ± 0.4	8.50 ± 0.45	8.38 ± 0.26
Chl.a (μg/L)	$5.34 \pm 4.31^{PR, LA}$	$10.49 \pm 8.10^{MR, RE}$	$9.92 \pm 5.43^{MR, RE}$	$4.40 \pm 1.36^{PR, LA}$
COD_{Mn} (mg/L)	3.75 ± 0.79^{LA}	4.38 ± 0.10^{RE}	$4.90 \pm 1.57^{MR, RE}$	$3.53 \pm 0.71^{PR, LA}$
TN (mg/L)	1.81 ± 1.02^{PR}	$4.31 \pm 2.63^{MR, LA, RE}$	1.63 ± 0.77^{PR}	1.33 ± 0.51^{PR}
TP (mg/L)	$0.028 \pm 0.013^{PR, LA}$	$0.094 \pm 0.071^{MR, RE}$	$0.101 \pm 0.142^{MR, RE}$	$0.028 \pm 0.010^{PR, LA}$
S	$37 \pm 20^{PR, LA, RE}$	$11 \pm 6^{MR, RE}$	17 ± 9^{MR}	13 ± 7^{MR}
D (ind./m^2)	284.1 ± 122.5	259.9 ± 289.7	175.8 ± 125.4	696.5 ± 1193.3
BMI (g/m^2)	79.6 ± 66.1	109.1 ± 14.7^{RE}	77.0 ± 63.6	32.0 ± 56.7^{PR}
H'	$2.59 \pm 0.61^{PR, LA, RE}$	1.82 ± 0.41^{MR}	2.03 ± 0.8^{MR}	1.66 ± 0.77^{MR}

Note: Main variations of the measured environmental variables and macroinvertebrate metrics in different aquatic habitat types are listed above with further statistical details provided in Tables S3 and S4. Bold superscripts represent significant Kruskal–Wallis and post hoc test results, and italic superscripts represent significant ($p < 0.05$) ANOVA and post hoc results. Abbreviations: MR - mountain river; PR - plain river, LA - lake; RE - reservoir; h - water depth; SD - transparency; v - velocity; WT - water temperature; DO - dissolved oxygen; Chl.a - Chlorophyll a; COD_{Mn} - permanganate index; TN - total nitrogen; TP - total phosphorus; S - species richness; D - abundance; BMI - biomass.

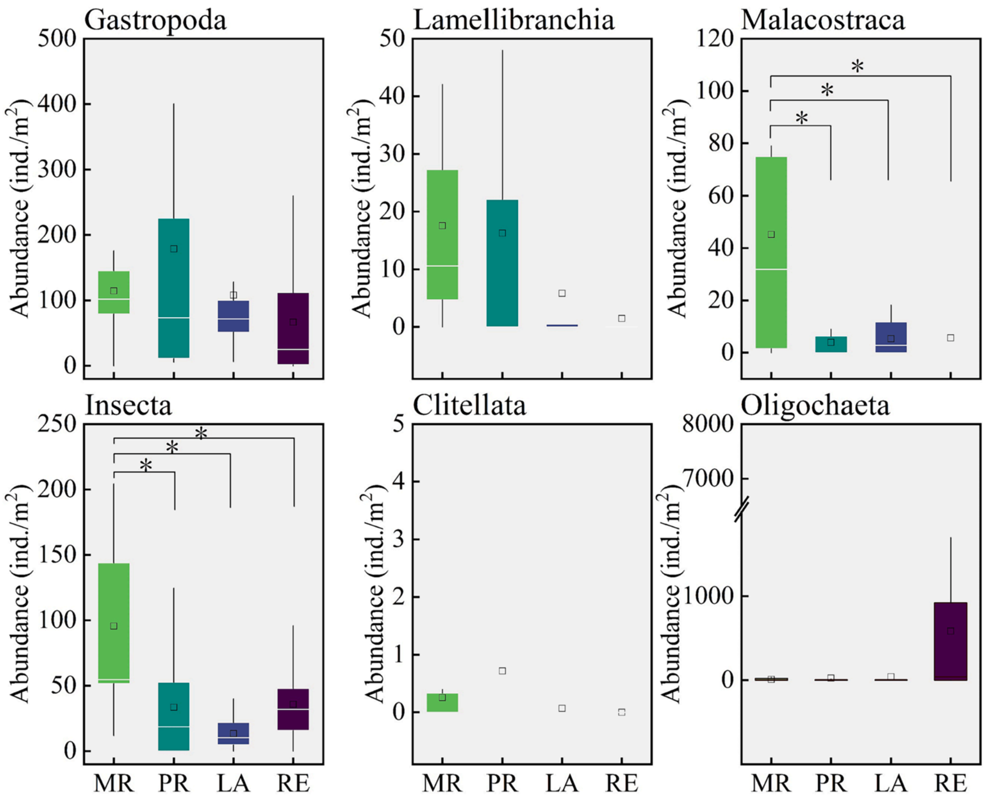

Fig. 2. Box plots comparing the abundance of macroinvertebrates among the four types of aquatic habitat. The box represents the interquartile range (IQR), white horizontal line represents the median, black square indicates the mean value, whiskers represent 1.5 × IQR, and the upper and lower box whiskers are maxima and minima. Abbreviations: MR - mountain river; PR - plain river; LA - lake; and RE - reservoir. *$p \leq 0.05$. Further statistical details are provided in Table S5.

0.6–0.8 representing "good", and 0.8–1.0 representing "high".

3. Results

3.1. Environmental variables and macroinvertebrate community structure

Variations of the measured environmental variables in different aquatic habitat types are listed in Table 1. Mountain rivers had the shallowest h (0.85 ± 0.47 m), the highest v (0.491 ± 0.20 m/s) that was markedly higher than that of plain rivers (0.014 ± 0.002 m/s), and the lowest TP content (0.028 ± 0.013 mg/L). Plain rivers had shallower h (1.12 ± 0.92 m), lower v (0.014 ± 0.002 m/s), the highest Chl.a content (10.49 ± 8.10 μg/L), and the highest TN content (4.31 ± 2.63 mg/L) that was substantially higher than that of the other aquatic habitat types (1.55 ± 0.79 mg/L). Lakes had shallower h (1.46 ± 1.03 m), the highest COD_{Mn} content (4.90 ± 1.57 mg/L), and the highest TP content (0.101 ± 0.142 mg/L). Reservoirs had the greatest h and SD (21.22 ± 13.52 and 2.33 ± 1.14 m, respectively) that were markedly higher than those of the other aquatic habitat types (1.32 ± 1.51 and 0.75 ± 0.49 m, respectively), as well as the lowest WT (19.56 ± 4.06 °C), lowest DO content (8.47 ± 1.13 mg/L), lowest Chl.a and COD_{Mn} content (4.40 ± 1.36 μg/L and 3.53 ± 0.71 mg/L, respectively), and lowest content of both TN and TP (1.33 ± 0.51 and 0.028 ± 0.010 mg/L, respectively). Overall, the DO content was relatively high in all four aquatic habitat types (9.22 ± 1.45 mg/L) and all aquatic habitat types were weakly alkaline (pH: 8.50 ± 0.35).

We collected and identified 15,826 macroinvertebrate individuals belonging to 3 phyla, 7 orders, 58 families, 124 genera, and 188 taxa. Among them, there were 128 insect taxa (68.1%), 26 gastropod taxa (13.8%), 11 malacostracan taxa (5.9%), 8 oligochaete taxa (4.3%), 8 lamellibranch taxa (4.3%), 6 clitellate taxa (3.2%), and 1 polychaete taxon (0.5%).

The species richness, abundance, biomass, and the Shannon–Wiener Index of macroinvertebrates varied greatly among the four types of aquatic habitat (Table 1; Figs. 2 and 3). The species richness of mountain river sites was 153, with that of individual sites varying from 5 to 67. Abundance varied from 112.0 to 449.7 ind./m^2, biomass varied from 0.33 to 221.9 g/m^2, and the Shannon-Wiener Index varied from 1.48 to 3.28. The species richness of the plain river sites was 64, with that of individual sites varying from 4 to 26. Abundance varied from 24.0 to 1074.1 ind./m^2, biomass varied from 0.2 to 387.3 g/m^2, and the Shannon–Wiener Index varied from 1.32 to 2.79. The species richness of the lake sites was 68, with that of individual sites varying from 4 to 34. Abundance varied from 79.3 to 466.0 ind./m^2, biomass varied from 8.17 to 198.13 g/m^2, and the Shannon–Wiener Index varied from 0.31 to 3.08. The species richness of the reservoir sites was 72, with that of individual sites varying from 2 to 27. Abundance varied from 32.0 to 5546.0 ind./m^2, biomass varied from 0.02 to 257.7 g/m^2, and the Shannon–Wiener Index varied from 0.056 to 2.88. Overall, the macroinvertebrate diversity (α diversity) of mountain rivers (37 ± 20) was substantially higher than that of the other types of aquatic habitat (13 ± 8, Table 1), with markedly higher abundances of malacostracans and insects (Fig. 2). Biomass was notably higher in plain rivers (109.1 ± 125.6 g/m^2), abundance was substantially lower in lakes (175.8 ± 125.4 ind./m^2), and reservoirs had the highest abundance (696.5 ± 1193.3 ind./m^2) and the lowest biomass (32.0 ± 56.7 g/m^2).

The indicator value results showed that the numbers and composition of indicator taxa were different in the four types of aquatic habitat (Table 2). The largest number of indicator taxa (17 taxa) in mountain rivers were typical widespread freshwater taxa that included *Radix swinhoei*, *Semisulcospira cancellata*, *Hippeutis umbilicalis*, *Corbicula fluminea*, and *Limnodrilus udekemianus*, which are eutrophic indicator taxa with high tolerance, and *Acanthacorydalis orientalis*, *Heptagenia* sp., and *Hydropsyche* sp., which are clean-stream indicator taxa with high

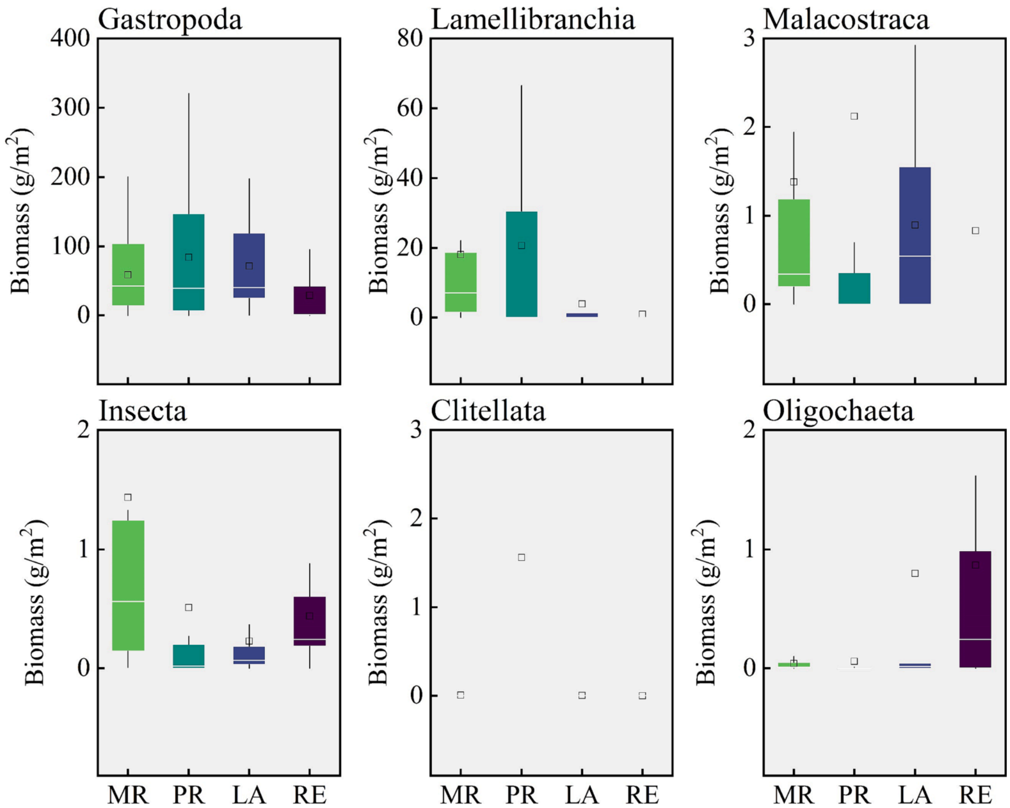

Fig. 3. Box plots comparing the biomass of macroinvertebrates among the four types of aquatic habitat. The box represents the interquartile range (IQR), white horizontal line represents the median, black square indicates the mean value, whiskers represent 1.5 × IQR, and the upper and lower box whiskers are maxima and minima. Abbreviations: MR - mountain river; PR - plain river; LA - Lake; RE - reservoir.

sensitivity. Fewer indicator taxa (only four widespread freshwater taxa) were identified in the plain rivers: *Bithynia fuchsiana*, *Chironomus riparius*, *Physa acuta*, and *Chironomus pallidivittatus*. The indicator taxa in the lakes were dominated by freshwater snails with extreme tolerance (for pollution or extreme environmental conditions), e.g., *Bithynia misella* and *Cipangopaludina chinensis*. Thirteen taxa in the reservoirs were identified as indicator taxa, including *Radix auricularia*, *Bellamya aeruginosa*, and *Semisulcospira amurensis*, which are widespread freshwater taxa with unclear indication, and *Propsilocerus akamusi*, *Einfeldia dissidens*, *Tubifex tubifex*, and *Limnodrilus hoffmeisteri*, which are typical endemic deepwater taxa.

3.2. Relationships between environmental variables and macroinvertebrate community structure

The results of DCA based on all macroinvertebrate data showed a value of Lth < 4; thus, the RDA model was used. The results of DCA based on the indicator taxon data showed a value of Lth > 4; thus, the CCA model was used. The results of RDA and CCA showed that the taxonomic composition and the distribution of macroinvertebrates were significantly affected by v, WT, TN, and h (Fig. 4; Tables S6 and S7). Most aquatic insects (e.g., EPT taxa, belonging to the Ephemeroptera, Plecoptera, or Trichoptera), crustaceans (e.g., *Gammaridea* sp.), bivalves (e.g., *C. fluminea*), and some freshwater snails (e.g., *Radix lagotis*) exhibited high v preference. Most chironomid taxa (e.g., *C. riparius*) and freshwater snails (e.g., *C. chinensis*) exhibited high WT and high TN content preference. Some of the oligochaete taxa (e.g., *T. tubifex*) and chironomid taxa (e.g., *P. akamusi*) showed a preference for high h, whereas most other taxa showed a negative preference for high h. Additionally, macroinvertebrate communities were weakly correlated with water quality metrics such as TP, COD_{Mn}, and DO (Table S7), indicating that macroinvertebrate commnuity structure and diversity were not fully determined by these metrics in this study.

3.3. Assessment of ecological quality

A total of 19 macroinvertebrate metrics were screened for MMI development, comprising 3 biodiversity metrics, 9 morphological structure metrics, 4 functional structure metrics, and 3 tolerance metrics (Fig. S2). Again, SD and WT were removed from the 12 environmental metrics, and the environmental metrics h, DO, pH, Chl.a, COD_{Mn}, TN, TP, v, longitude, and latitude were included in the GAM model (Fig. S3).

The values of macroinvertebrate metrics, GAM BIC, and suggested weights between different types of aquatic habitat are listed in Table 3. Combinations of different macroinvertebrate metrics showed important differences in suggested weights, and varied from 3.3% (crustacean–mollusk richness) to 9.8% (relative abundance of tolerant taxa). Metrics reflecting the taxonomic composition and diversity of Ephemeroptera taxa, Plecoptera taxa, and Trichoptera taxa (e.g., EPT richness), and metrics related to tolerance (e.g., the relative abundance of sensitive taxa, biological index, and relative abundance of tolerant taxa) showed predictable relationships between macroinvertebrates and environmental variables, and these metrics were weighted at 30.1% of the total. Although the relationships between the single metrics reflecting chironomids and mollusks (e.g., chironomid richness, relative abundance of Diptera, relative abundance of chironomids, crustacean–mollusk richness, and relative abundance of crustaceans and mollusks) and environmental metrics were not sensitive (generally < 5.0%), the weight of the response to environmental metrics increased significantly (19.0%) when they were combined. Four metrics reflecting the functional structure (i.e., the relative abundance of scrapers, relative abundance of shredders, relative abundance of clingers, and richness of gathering collectors + filtering collectors) contributed 19.3% of the weight to the MMI. Additionally, combination of the relative abundances of the five

Table 2
Indicator taxa in the four types of aquatic habitat.

Mountain River	Plain river	Lake	Reservoir
Thienemannimyia geijskesi (*TG*, 0.99)	*Physa acuta* (*PHY*, 0.95)	*Pila tischbeini* (*PI*, 1)	*Limnodrilus claparedeianus* (*LC*, 0.94)
Polypedilum albicorne (*PO*, 0.94)	*Bithynia fuchsiana* (*BF*, 0.89)	*Bithynia misella* (*BM*, 1)	*Tubifex tubifex* (*TUB*, 0.93)
Limnodrilus udekemianus (*LU*, 0.93)	*Chironomus pallidivittatus* (*CP*, 0.83)	*Viviparus chui* (*VC*, 0.86)	*Parafossarulus striatulus* (*PAR*, 0.75)
Parapenaeopsis hardwickii (*PA*, 0.93)	*Chironomus riparius* (*CR*, 0.78)	*Cipangopaludina chinensis* (*CI*, 0.79)	*Limnodrilus hoffmeisteri* (*LC*, 0.73)
Polypylis hemisphaerula (*PHE*, 0.92)			*Bellamya aeruginosa* (*BAL*, 0.71)
Paracercion sp. (*PS*, 0.88)			*Semisulcospira amurensis* (*SA*, 0.71)
Gammarus sp. (*GA*, 0.86)			*Bellamya limnophila* (*BL*, 0.63)
Polypedilum nubifer (*PN*, 0.86)			*Propsilocerus paradoxus* (*PP*, 0.62)
Hippeutis umbilicalis (*HU*, 0.86)			*Propsilocerus akamusi* (*PR*, 0.59)
Radix lagotis (*RL*, 0.84)			*Einfeldia dissidens* (*ED*, 0.57)
Semisulcospira cancellata (*SC*, 0.84)			*Monopylephorus limosus* (*ML*, 0.57)
Acanthacorydalis orientalis (*AO*, 0.83)			*Radix Auricularia* (*RA*, 0.54)
Corbicula nitens (*CN*, 0.79)			*Radi plicatula* (*RP*, 0.5)
Heptagenia sp. (*HEP*, 0.78)			
Corbicula fluminea (*CF*, 0.73)			
Hydropsyche sp. (*HS*, 0.71)			
Radix swinhoei (*RS*, 0.56)			

Note: Abbreviations and indicated values listed in parentheses.

most dominant macroinvertebrates showed a weight of 27.0% in response to environmental metrics, indicating that the structure of the dominant taxa (Table S8) was important for the MMI development.

Using the weights assigned in Table 3, the observed MMI (0.38 ± 0.22) and the predicted MMI (0.39 ± 0.22) were calculated by weighting the macroinvertebrate metrics predicted by the GAM and the measured metrics, respectively, with a value of R^2 of 0.63 (Fig. 5).

Among all sites (Fig. 6), those with high and good ecological quality were relatively few (4 and 5, respectively) and accounted for relatively low percentages (6.56% and 8.2%, respectively). Moderate ecological quality was found at 18 sites, with a percentage of 29.51%. Sites with poor or bad ecological quality were more numerous, with 23 and 11 sites or percentages of 37.7% and 18.03%, respectively. The percentage of mountain river sites with good to high ecological quality was notably high, reaching 64.29%. The percentage of plain river sites with poor ecological quality was notably high, reaching 87.5%. Among the lake sites, those with moderate and poor ecological quality numbered 6 (60.00%) and 4 (40%), respectively. For reservoir sites, the ecological quality of 8 sites was moderate (38.10%) and that of 13 sites was poor (61.90%). Overall, the ecological quality of the four types of aquatic habitat decreased in the following order: mountain rivers (0.62 ± 0.28) > lakes (0.43 ± 0.09) > reservoirs (0.32 ± 0.12) > plain rivers (0.24 ± 0.12).

4. Discussion

The MMI showed good detection of the ecological quality of the 61 sites (Figs. 5 and 6). Overall, 14.76% of sites with good to high ecological quality were located in mountain rivers (64.29% of mountain rivers), indicating that the ecological quality of Beijing's mountain rivers was generally good, mainly because of the protective policies and measures of the managers. For example, in M5–M11, this section was largely perennially dry and the ecological quality was poor in 2017 (Mu et al., 2018; Zhang et al., 2019); in March 2019, continued replenishment of water from the upstream section reservoir (i.e., R10–R12) to the downstream sections in batches led to improved river connectivity and better hydraulic condition (Li et al., 2022; Zhai et al., 2022). Furthermore, sufficient DO, low content of both nitrogen and phosphorus, and low organic pollution have led to recovery in the diversity of aquatic plants, phytoplankton, and reeds in the stream (unpublished). These factors, coupled with reduced landscape modification, together with increased habitat complexity and heterogeneity (Mo et al., 2021), can support a suitable habitat for macroinvertebrates. However, the ecological quality at some mountain river sites was still poor or bad (21.43% of mountain rivers), mainly related to water pollution. Sites M3 and M4, for example, have received domestic and industrial sewage from cities over a long period, causing severe excesses in nitrogen, phosphorus, and COD content, and the water body has undergone extreme deterioration (Lin et al., 2019); thus, the ecological quality was found to be poor. Since 2019, the ecological engineering of this river section was thoroughly modified by dredging, planting of aquatic plants, and introducing reclaimed water (Ni, 2021). Currently, the water quality in this river section is substantially improved (Shao et al., 2019), and it is believed that both the macroinvertebrate community structure and the ecological quality will be further improved by these measures.

The total percentage of sites with bad to poor ecological quality was as high as 55.73%, which was mainly related to strong anthropogenic interference. For instance, at sites L1–L5, the lakes were greatly modified by managers through river filling, application of antiseepage membranes, construction of new lakes, and substrate reshaping, leading to weakened water dynamics and reduced water connectivity, and the water bodies lacked efficient renewal, as shown by their high nitrogen content, high phosphorus content, high organic pollutant content, and high phytoplankton biomass (Pan et al., 2022). Moreover, biotic releases and hydrophyte removal are conducted periodically in some lakes (Fig. S4), which disrupt the nutrient balance of the lake ecosystem and homogenize the substrate. These actions have interrupted the metapopulation dynamics of aquatic biota (Zhang et al., 2022), leading to homogenization of macroinvertebrate communities, mainly characterized by a loss of sensitive taxa (e.g., EPT) and increase in widespread or tolerant taxa, which is a finding supported by the indicator taxon results (Table 2). Additionally, the ecological quality of the plain river sites was poor (Fig. 6), mainly because of unregulated ecological replenishment and river channelization. For example, at sites P1–P4 and P6–P8, the water bodies were supplied entirely from upstream sections (3–4 times per year), and because of prolonged river disconnection (~25 years), the groundwater level declined and channel desertification occurred, resulting in severe water infiltration (Fig. S4). Thus, the river was disconnected for a short period (~15 days) after replenishment ceased. However, restoration of macroinvertebrate communities generally requires a long period (around one month or longer; Xu et al., 2014, 2018), and the substrate and flow velocity in these reaches simply cannot support the restoration of macroinvertebrate communities (Duan et al., 2010).

All reservoir sites exhibited moderate, porr, or bad ecological quality, which might indicate the effect of water depth and substrate on macroinvertebrate taxa (Duan et al., 2010). For example, the substrate in the Miyun Reservoir, with an average water depth of ~40 m, was mainly mud and severely homogenized, with weak lighting, low DO, and oligotrophic conditions, which limited the ecological niches available

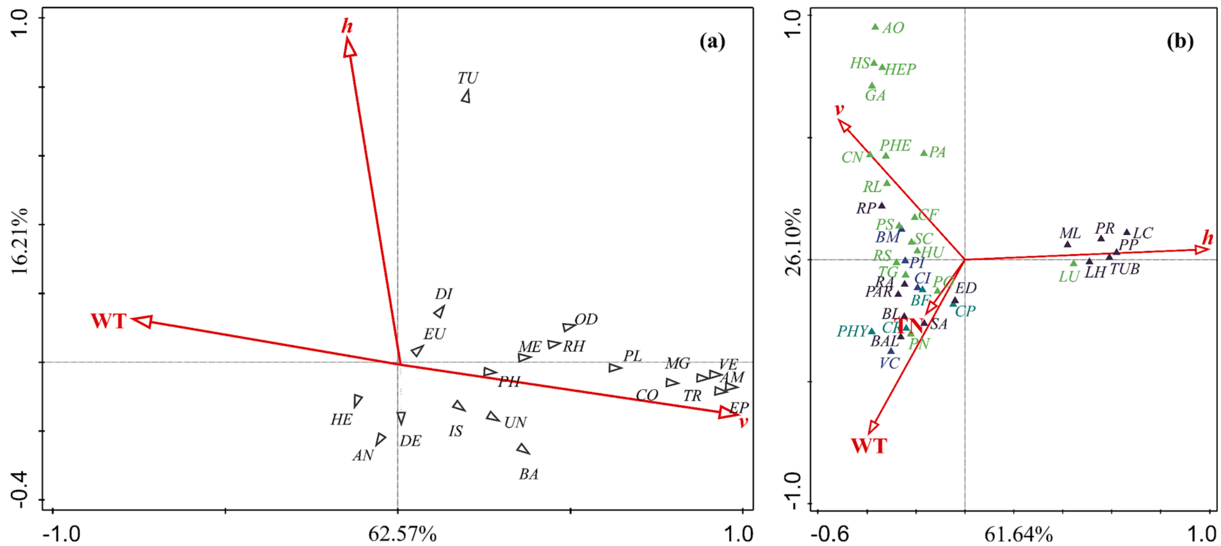

Fig. 4. (a) Results of RDA illustrating the relationships between macroinvertebrate abundances and the environmental variables in typical waters of Beijing. Macroinvertebrate taxa: ME - Mesogastropoda; BA - Basommatophora; AN - Anisomyaria; VE - Venerida; EU - Eulamellibranchia; UN - Unionoida; DE - Decapoda; AM - Amphipoda; IS - Isopoda; DI - Diptera; OD - Odonata; HE - Hemiptera; CO - Coleoptera; TR - Trichoptera; PL - Plecoptera; MGa - Megaloptera; EP - Ephemeroptera; PH - Pharyngobdellida; RH - Rhynchobdellida; GN - Gnathobdellida; TU - Tubificida; NE - Nereidida. (b) Results of CCA showing the relationships between indicator taxa and environmental variables in typical waters of Beijing. Abbreviations: h - depth; WT - water temperature; v - velocity; TN - total nitrogen. Abbreviations for indicator taxa are given in Table 2. Here, we show the environmental factors significantly associated with macroinvertebrate abundance; other detailed results are provided in Tables S6 and S7.

Table 3
Average values of macroinvertebrate metrics for each of the four types of aquatic habitat.

Variables	MR	PR	LA	RE	GAM's BIC	Weights
RA of sensitive taxa	0.12 ± 0.09	0.073 ± 0.11	0.16 ± 0.17^{RE}	0.04 ± 0.06^{LA}	−93.26	9.80%
RA of tolerant taxa	$0.88 \pm 0.086^{\mathbf{RE}}$	0.93 ± 0.11	0.84 ± 0.17	$0.96 \pm 0.07^{\mathbf{MR}}$	−93.26	9.80%
GC + FC Richness	$20.5 \pm 11.18^{PR,\ LA,\ RE}$	5.81 ± 5.39^{MR}	9.45 ± 6.88^{MR}	9.38 ± 4.81^{MR}	−65.28	7.50%
Biological Index	$6.27 \pm 0.87^{\mathbf{RE}}$	6.59 ± 1.13	$5.85 \pm 2.37^{\mathbf{RE}}$	$8.04 \pm 1.33^{\mathbf{MR,\ LA}}$	−48.97	6.60%
RA of 3rd dominant taxa	0.097 ± 0.03	0.14 ± 0.05	0.12 ± 0.05	0.12 ± 0.05	−27.12	5.70%
RA of 4th dominant taxa	0.079 ± 0.03	$0.11 \pm 0.04^{\mathbf{RE}}$	0.09 ± 0.05	$0.064 \pm 0.04^{\mathbf{PR}}$	−22.29	5.60%
RA of 5th dominant taxa	0.067 ± 0.03	0.065 ± 0.031	0.06 ± 0.04	0.043 ± 0.03	−20.04	5.50%
RA of 2nd dominant taxa	$0.12 \pm 0.05^{\mathbf{PR,\ RE}}$	$0.19 \pm 0.05^{\mathbf{MR}}$	0.16 ± 0.09	$0.21 \pm 0.1^{\mathbf{MR}}$	−7.78	5.10%
RA of 1st dominant taxa	0.26 ± 0.14	0.34 ± 0.11	0.33 ± 0.24	0.41 ± 0.18	−5.65	5.10%
Chironomidae Richness	4.57 ± 2.47	2.31 ± 2.25	3.27 ± 3.02	4.29 ± 3.03	20.23	4.40%
RA of SC	0.37 ± 0.19	$0.53 \pm 0.27^{\mathbf{RE}}$	$0.51 \pm 0.28^{\mathbf{RE}}$	$0.25 \pm 0.25^{\mathbf{PR,\ LA}}$	27.87	4.20%
RA of SH	0.13 ± 0.08	0.09 ± 0.09	0.05 ± 0.059	0.09 ± 0.07	42.29	3.90%
EPT Richness	$7.0 \pm 6.2^{PR,\ LA,\ RE}$	0.06 ± 0.2^{MR}	0.09 ± 0.3^{MR}	0.1 ± 0.4^{MR}	42.29	3.90%
RA of non-insect	0.67 ± 0.21	0.71 ± 0.34	0.88 ± 0.13	0.8 ± 0.24	44.95	3.90%
RA of Chironomidae	0.21 ± 0.24	0.27 ± 0.33	0.096 ± 0.11	0.2 ± 0.24	47.66	3.90%
RA of Diptera	0.23 ± 0.24	0.27 ± 0.34	0.1 ± 0.12	0.24 ± 0.27	50.26	3.80%
RA of cn	0.18 ± 0.08^{RE}	0.16 ± 0.16^{RE}	0.2 ± 0.28	$0.46 \pm 0.37^{MR,\ PR}$	56.82	3.70%
RA of Crustacea-Mollusca	0.41 ± 0.27	0.57 ± 0.37	0.62 ± 0.3	0.3 ± 0.35	59.77	3.70%
Crustacea-Mollusca Richness	$12.36 \pm 6.17^{\mathbf{PR,\ RE}}$	$5.5 \pm 3.5^{\mathbf{MR}}$	6.73 ± 4.45	$4.43 \pm 4.72^{\mathbf{MR}}$	92.61	3.30%

Note: Weighting of values used in the MMI synthesis were determined using goodness of fit (BIC) to the physical environmental variables in generalized additive models. Values of macroinvertebrate metrics, GAM's BIC, and the suggested weights among the different types of aquatic habitat are listed in Table 3 with further statistical details provided in Tables S9 and S10. Bold superscripts represent significant Kruskal–Wallis and post hoc test results, and italic superscripts represent significant ($p < 0.05$) ANOVA and post hoc results. MR - mountain river; PR - plain river; LA - lake; RE - reservoir. EPT: a collective group of insects belonging to Ephemeroptera, Plecoptera, and Trichoptera. RA: relative abundance, ranging from 0 to 1. Functional feeding groups: FC - filtering collectors; GC - gathering collectors; SC - scrapers; PR - predators; SH - shredders. Functional habit groups: cn - clingers.

for macroinvertebrate taxa (Hu et al., 2018; Li et al., 2018). Therefore only a few tolerant taxa (e.g., oligochaete taxa and chironomid larvae) were found distributed in this reservoir, as supported by our results (Figs. 2 and 3; Table 2). In comparison, the water depth in the Huairou Reservoir was generally shallow (~8 m), the substrate was mainly sand, the water renewal cycles were short (3–10 times per year), and the submerged plants at the bottom of the reservoir were abundant; thus, this reservoir could support the survival of certain taxa other than oligochaetes or chironomids (e.g., gastropods) (Table S11).

Overall, the effects of urban modification on macroinvertebrate diversity can be strongly specific to certain taxa and habitat types; furthermore, such effects might have impacts on seasonal changes in macroinvertebrate diversity (e.g., Fig. S5). Projects addressing ecological water replenishment/transfer and ecological restoration in Beijing have contributed to a considerable increase in groundwater recharge and initial improvements in water quality (Long et al., 2020). Although ecological quality remains poor, biodiversity in the freshwater ecosystem of the megacity of Beijing is gradually being improved by these restoration projects, and managers have changed their attitude from that of "no management" to that of "starting to manage." We anticipate that such "management" or "restoration" will have substantial implications for biodiversity restoration (Benayas et al., 2009;

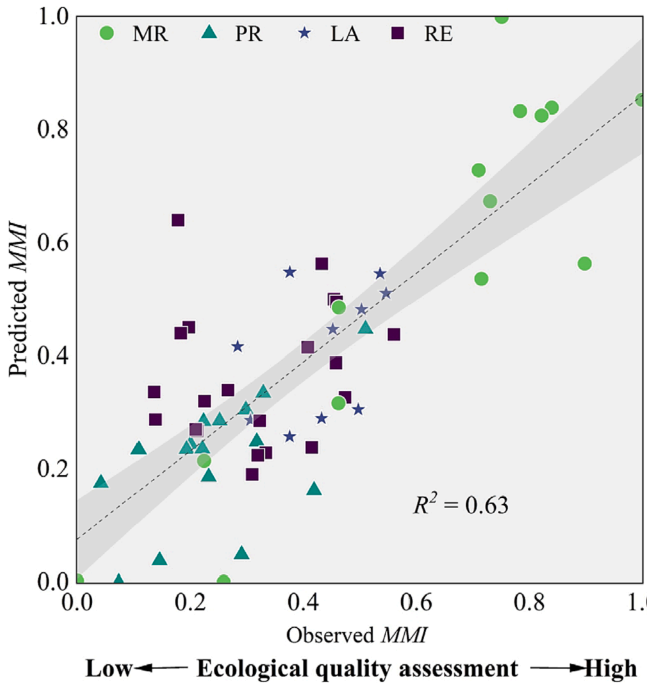

Fig. 5. Comparison of observed MMI values and predicted MMI values for 61 sampling sites. Abbreviations: MR - mountain river; PR - plain river; LA - lake; RE - reservoir. Areas in gray represent 95% confidence intervals.

Lepori et al., 2005; Palmer et al., 2014).

The world's great rivers have been endangered by unprecedented ecological pressures (Best, 2019), resulting in the decline/degradation of freshwater ecosystem services (Butchart et al., 2010). However, nearly all ecological restoration efforts focus on rehabilitating river/freshwater ecosystems for human services (e.g., recreation) in small river reaches, not restoring entire river/freshwater ecosystems to some pristine or natural state (Birk et al., 2012; Feio et al., 2021), which inevitably conflicts with other restorative strategies (e.g., biodiversity restoration) (Bullock et al., 2011), as also shown in our findings. Therefore, it is critical to develop ecological restoration measures aimed at restoring biodiversity and multiple service functions to improve river/watershed ecological quality. One key point is to determine the sources of river/watershed degradation and the scale of the ecological pressures that should be reduced for effective restoration of river/watershed function or structure, depending on the nature of the stressors for a particular river/watershed (Palmer et al., 2014). Given these concerns and the existing ecological pressures on urban waters in Beijing, we further suggest that discharge of urban and domestic sewage should be completely eliminated and instead continuously recycled as recycled water (i.e., point source pollution should be eliminated to optimize water quality); the substrate of urban rivers and lakes should be continuously improved with reference to the substrate type of mountain rivers (i.e., spatial heterogeneity of the substrate should be increased to improve the potential for biological recovery); and ecological water replenishment/transfer projects should not adopt a policy of stable total water allocation, but rather conisder a strategy of sustained ecological flow (i.e., increasing longitudinal connectivity of river segments to ensure that dispersal of biological propagules is not deterred); and the

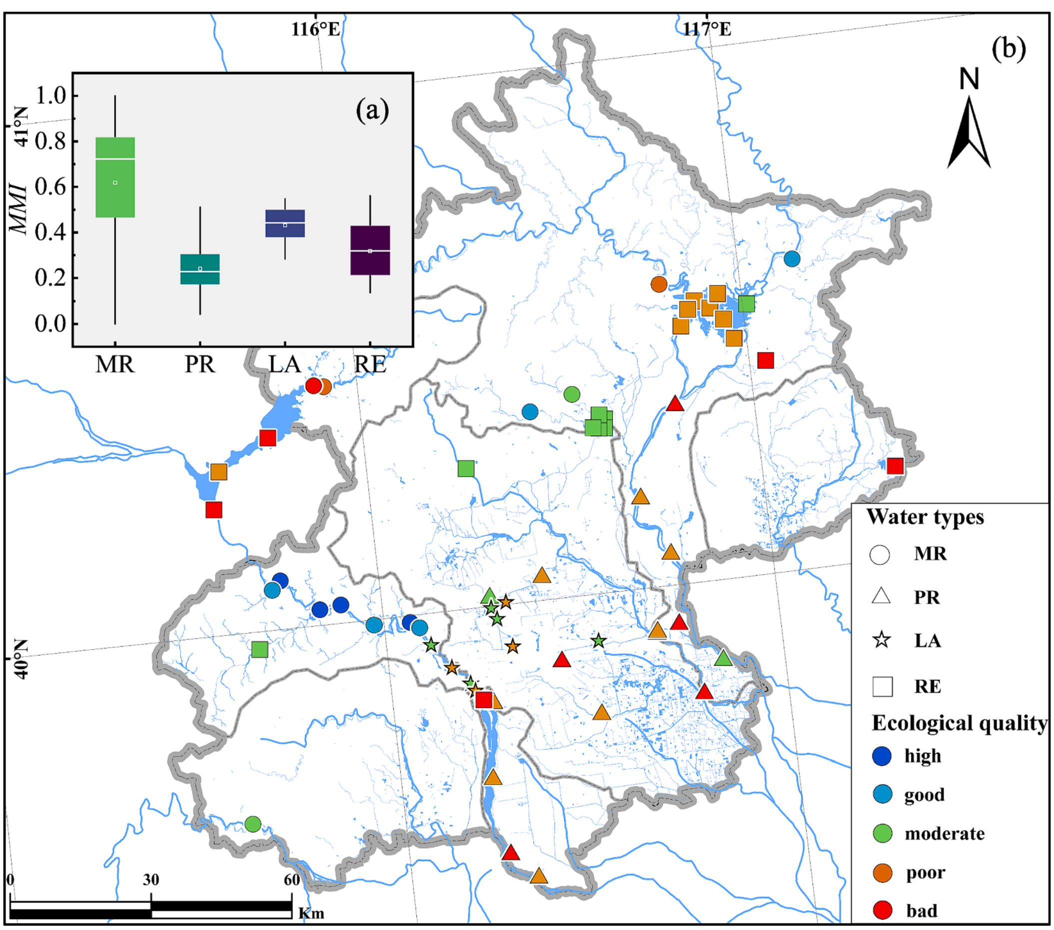

Fig. 6. (a) Differences in ecological classes of the four types of aquatic habitat. (b) Spatial ecological quality classes of the 61 sampling sites. Abbreviations: MR - mountain river; PR - plain river; LA - lake; RE - reservoir.

water quality and multiple taxa should be monitored continuously (focusing on the restoration of ecological processes to identify key stressor information). It is expected that such recommendations will be useful in helping to improve urban water system structure and function, especially regarding biodiversity restoration, and also create better habitats for macroinvertebrates.

The MMI that we have developed addresses some of the shortcomings of previous studies, driving us to propose a novel idea for a scientific and cost-effective bioenvironmental evaluation tool. We achieved this by dynamically assigning metric weights through the GAM method, which eliminates the need to sample the reference site during construction of the metric system, and allows for rapid evaluation of the ecological health of water bodies based on actual sampling results. This method is expected to be effective in application to assessment of the water ecosystem of a megacity. However, most Chinese urban freshwater ecosystems are still in the early stages of reconstruction and many water replenishment/transfer projects are still ongoing (e.g., the South-to-North Water Transfer Project, the Water Diversion Project from the Yellow River to Beijing, and the West–South Water Transfer Project). This motivated us to further propose improved methods, because the responses of most aquatic biota to the ecological modification of waters remain unclear. In the future, we expect to incorporate additional physical environment metrics, biological trait metrics, and taxa into this system. However, given the locality of the biotic–environmental relationship, it is necessary to work in each region/area after a framework is constructed, and to construct a metrics system that matches the local habitat characteristics.

5. Conclusions

The Beijing megacity water system is subject to considerable anthropogenic interference and ecological modification, but comprehensive assessments of the impacts of these phenomena are lacking. To this end, we conducted a systematic survey of the macroinvertebrates and environment in four typical urban aquatic habitat types in Beijing, and developed an MMI based on macroinvertebrates to assess the ecological quality of a megacity freshwater ecosystem under different ecological pressures. The results showed that most of the mountain rivers, with relatively little anthropogenic disturbance, had good ecological quality that could support strong macroinvertebrate communities. In contrast, the plain rivers that have undergone ecological modification and the few mountain rivers with strong anthropogenic interference had poorer hydraulic conditions and ecological quality that have caused loss of sensitive taxa (e.g., EPT taxa) and increase in widespread or tolerant taxa (e.g., *C. chinensis* and *P. acuta*). Compared with reservoirs, lake sites indeed showed minor increases in macroinvertebrate diversity and better ecological quality after ecological modifications (e.g., substrate reshaping). To enhance the conservation of macroinvertebrate diversity in urban water bodies, we propose emphasizing the hydrodynamic conditions and connectivity between water bodies. Additionally, managers should improve macroinvertebrate habitats by reducing water pollution and improving the substrate of rivers or lakes, with the aim of providing ecologically meaningful support for integrated management of megacity water systems. The MMI we developed represents a quantitative and integrated assessment method for coupling biological and environmental variables in megacity freshwater ecosystems, is sensitive to different gradients of disturbance, and can rapidly evaluate the impacts of both anthropogenic disturbances and ecological modification projects in megacity freshwater ecosystems. Thus, the MMI represents a promising integrated ecological assessment tool that is widely applicable to the management, conservation, and restoration of megacity freshwater ecosystems. In the future, sustainable ecological engineering strategies will be urgently needed to improve the ecological quality of urban water systems.

CRediT authorship contribution statement

Congcong Wang: Investigation, Data curation, Formal analysis, Methodology, Writing – original draft. **Xiongdong Zhou:** Conceptualization, Methodology, Writing – review & editing. **Mengzhen Xu:** Conceptualization, Funding acquisition, Methodology, Writing – review & editing. **Linyuan Zhang:** Investigation, Methodology. **Xinjue Hou:** Investigation, Methodology. **Zhongsuo Wang:** Conceptualization, Funding acquisition, Methodology, Writing – review & editing. **Yao Yang:** Methodology, Writing – review & editing. **Yaqi Luo:** Methodology, Writing – review & editing.

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.

Data availability

Data will be made available on request.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.ecolind.2023.110235.

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