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Research Article

Biological and Ecological Characteristics of Fish in the Downstream Area of the Mamaya Hydropower Station and Reservoir Section of the Dongqing Hydropower Station on the Beipan River, China

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As key indicator species for aquatic ecosystem health under cascade hydropower development, shifts in fish diversity directly reflect the cumulative impacts of hydropower projects on river connectivity, hydrological regimes, and habitat integrity, providing critical evidence for assessing ecological equilibrium and formulating conservation strategies. To assess the effects of hydropower development on the fish community structure and resource status in the mid-reach of the Beipan River and optimize the ecological sustainability of cascade hydropower operations, a field survey was conducted between the Mamaya hydropower station and the Dongqing hydropower station from 2020 to 2023. The results revealed that 2776 fish samples were recorded during the survey period, representing 38 species from 31 genera, 9 families, and 4 orders. Analysis of the relative importance index revealed *Onychostoma sima* as the dominant species, with a significantly higher index than those of the other species. Across the study period, the Margalef richness index (D_{Ma}), Shannon–Wiener diversity index (H'), Pielou evenness index (J), and Simpson index (C) ranged from 4.790 to 5.758, 0.064–0.083, 0.822–0.895, and 0.908–0.950, respectively, with peaks observed in 2021. The abundance–biomass comparison curve (ABC) suggested an undisturbed fish community structure in 2020, transitioning to moderate disturbance from 2021 to 2023. The length–weight relationships (LWRs) of 28 fish species were analyzed. The LWRs of the four species in the Beipan River Basin have not been previously documented, and the maximum lengths of the seven species were recorded for the first time. When a value in the LWR associated with fish fertility was used as an indicator to assess changes in fish status and nutrition between 2020 and 2023, the 10 largest species presented the highest A values from 2020 to 2021, followed by a decline from 2022 to 2023. The construction of cascade hydropower stations is considered the main driving factor of fish habitat change, fish diversity, and fish resources in the Beipan River. The establishment of aquatic germplasm resource protection areas, strengthening law enforcement, controlling the sand mining period, and monitoring, rescuing, and breeding rare and endemic fish in river basins are suggested. This study supplemented the basic data of fish resources in the middle reaches of the Beipan River. It proposed suggestions for enhancing fish diversity protection to provide protection measures and a basis for the protection of fish resources for power stations and river basin managers.

Keywords: abundance–biomass comparison curve; Beipan River; fish diversity; fish resources; LWRs

1. Introduction

The Beipan River, a major tributary of the upper Xijiang River in the Pearl River Basin, traverses Yunnan and Guizhou Provinces with substantial hydropower potential. To exploit this resource, a cascade of 11 hydropower stations has been planned along the river's mainstream. However, the construction of these stations has profoundly altered the aquatic ecosystems of the Beipan River basin by disrupting longitudinal connectivity, altering natural flow regimes, and degrading critical habitats essential for fish life-history processes. These changes severely disrupt the migratory corridors necessary for feeding, overwintering, and spawning, thereby posing major challenges to the survival of migratory species [1, 2]. The Mamaya hydropower station, the second facility in the midstream cascade (downstream of Maokou), links the upstream Guangzhao and downstream Dongqing hydropower stations. With the commissioning of the Mamaya hydropower station, the entire midstream cascade has become fully operational, further intensifying hydrological alterations and permanently obstructing natural fish migration routes, with cascading ecological consequences [1, 3].

In the context of ongoing hydropower development along the Beipan River and the contemporary emphasis on integrated clean energy initiatives that combine water, wind, and solar energy, understanding the current status of fish resources, their diversity, and community variations is critical for developing effective conservation strategies and optimizing habitat spatial configurations. Length–weight relationships (LWRs) of fish serve as a crucial parameter for assessing fish physiological conditions, facilitating the estimation of length or weight for unmeasured or endemic species, and are vital for evaluating fish population health, which underpins fisheries management and conservation efforts [4–7]. However, the limited data on fish growth characteristics within the basin limit the ability to provide robust theoretical support for habitat conservation and ecological restoration. Despite relatively extensive data on fish resources in the midstream region of the Beipan River, analyses of fish species before and after the construction of the Mamaya hydropower station are limited due to the predominantly qualitative nature of available datasets focusing on fish ecological habits and the lack of systematic continuity in research processes [1, 8, 9]. Research typically focuses on specific periods; for example, between 2009 and 2010, Zhou et al. conducted four surveys in the Dongqing Reservoir, identifying 13 species and highlighting severe habitat fragmentation in the lower Beipan River, where conditions no longer supported the reproductive cycles of fish species laying drifting eggs [8]. In July 2014, Mo et al. systematically surveyed fish resources in the Dongqing Reservoir, documenting 20 species and noting that, despite reduced fish density linked to cascade hydropower stations, the population growth structure remained stable [1]. Between 2013 and 2015, Feng et al. conducted six sampling campaigns in the Dongqing Reservoir, capturing 668 fish from 3 orders, 8 families, 29 genera, and 39 species [9]. Their findings identified dominant species and revealed a shift in the fish community, with species adapted to still or slow-moving waters increasingly dominating the reservoir ecosystem. In general, existing surveys and research materials provide only a cursory description of fish species composition in the mainstream of the Beipan River, with no specialized studies focusing on the structure of fish communities and their diversity within this reach. Evidence suggests that investigations into the current status of fish resources can yield essential data on species composition and resource abundance, which are crucial for the management, conservation, and sustainable utilization of fishery resources. Therefore, research on the present state of fish community structures in the mainstream of the Beipan River is imperative. Furthermore, against the backdrop of hydroelectric development, which has obstructed fish migration pathways, compiling current data on fish resources and identifying the principal anthropogenic factors impacting these resources are vital for understanding compensatory habitat functions in the mainstream of the Beipan River.

To elucidate the status of the fish resources in the watershed encompassing the Mamaya hydropower station and the Dongqing hydropower station reservoir area following the operation of the Mamaya hydropower station, this study conducted a fish resource survey from 2020 to 2023. This study compared and analyzed the current status and community structure characteristics of fish populations downstream of the Mamaya hydropower station nearly a decade after its operation, evaluating the impact of cascade development on fish resources along the mainstream of the Beipan River. Additionally, this research established the LWRs of fish in this watershed for the first time. The findings of this study will provide information on the ecological impact of hydropower projects and contribute to the development of strategies for fish population protection within the mainstream of the Beipan River.

2. Materials and Methods

2.1. Study Area. Fisheries surveys were conducted from 2020 to 2023 during July through October each year in the mainstream of the watershed below the Mamaya Dam and the Dongqing Reservoir (24°07′–25°31′N, 105°30′–105°46′E). By integrating historical fishery resource survey methods, specifically the requirements for fishery resource survey points outlined in the "*Standard for the investigation of reservoir fishery resources,*" it is recommended that sampling sections and points be comprehensively arranged based on river length, width, and shoreline configuration, with the option of selecting sampling points on one or both sides. In accordance with the requirements for fish survey points in the "*Technical Code for Investigation and Assessment of Aquatic Ecosystem for Hydropower Projects,*" it is suggested that for fishery resource surveys in the open waters of reservoirs, survey points should be rationally established in the mainstream or important tributary flowing water areas of the reservoir. To ensure spatial representation along the longitudinal gradient of the river and to increase sampling efficiency, this study established 17 fishery resource monitoring points across 10 fish resource monitoring transects, with 1–3 points per monitoring transect determined by actual channel width profiles and each transect spanning a 1 km radius (Figure 1).

2.1.1. Survey Sampling and Data Collection. To catch a wide range of fish species of different lengths and to comply with local fish protection policies, fish sampling employed a combination of fixed-size ground traps (0.2 m high, 0.25 m wide, 8 m long, mesh size 5 mm, 10 traps per site) and fixed gill nets (10–30 m long, 1.5–2.5 m wide, mesh size 20–60 mm, 10 nets per site). Collection was conducted twice daily, at 7:00 a.m. and 5:00 p.m., with a total trapping duration of 10 h per day. During the survey, each fish caught was measured by an electronic balance and ruler to determine total length (TL), standard length (SL), and body weight (BW) onsite. The lengths and weights were recorded to the nearest 0.1 cm and 0.1 g, respectively. Species identification and recording were performed following Wu (1989), with species names verified through FishBase [10].

2.2. Data Analysis and Processing

2.2.1. Index of Relative Importance (IRI). The IRI was calculated to determine the dominance of fish species in the study area, utilizing the percentages of abundance, biomass, and frequency of occurrence. The calculation was performed using the following formula [11]:

$$\mathrm{IRI} = (N\% + W\%) \times F\%. \tag{1}$$

In this formula, N% represents the percentage of the total number of individuals of a specific species among the captured fish, W% denotes the percentage of the total biomass of that species, and F% indicates the frequency of occurrence of the species across the sampling sites. Fish species with IRI values exceeding 1000 were classified as dominant species. Species with IRI values between 100 and 1000 were categorized as important species, those with values between 10 and 100 were considered common species, and species with IRI values below 10 were identified as rare species [12].

2.2.2. Fish Diversity Analysis. This study employs the Margalef richness index (D_{Ma}), Shannon–Wiener diversity index (H'), Pielou evenness index (J), and Simpson index (C) to evaluate fish diversity, providing insights into the diversity characteristics of the studied watershed. The specific details and formulas for these indices are as follows [13–15]:

1. Margalef richness index (D_{Ma}): This index reflects the species richness within a biological community and is calculated as:

$$D_{\mathrm{Ma}} = \frac{(\mathrm{S}\text{-}1)}{\ln \mathrm{N}}, \tag{2}$$

where S represents the total number of captured fish species, and N represents the total number of individuals captured across all species.

2. Shannon–Wiener diversity index (H'): This index reflects the complexity of community structure, combining species richness and evenness in a comprehensive manner. The formula for this index is as follows:

$$H' = -\sum_{\mathrm{i}=1}^{\mathrm{s}} P_i \ln P_i, \qquad P_i = 100\%\left(\frac{N_i}{N}\right), \tag{3}$$

where H' is the Shannon–Wiener diversity index, P_i is the proportion of individuals of species i relative to the total number of individuals, and N_i represents the cumulative number of individuals of species i.

3. Pielou evenness index (J): This index quantifies the evenness in the distribution of individuals among species within a community and is calculated as follows:

$$J = \frac{H'}{\ln S}, \tag{4}$$

where S is the number of captured fish species and H' is the Shannon–Wiener diversity index.

4. Simpson index (C): This index measures the probability that two consecutive individuals sampled from the community belong to the same species. It ranges from 0 to 1, where lower values indicate higher community diversity, and higher values reflect lower diversity:

$$C = 1 - \sum_{\mathrm{i}=1}^{\mathrm{s}} P_{\mathrm{i}}^2. \tag{5}$$

In this formula, P_i is the proportion of individuals of species i relative to the total number of individuals sampled.

Origin 9.0 software was used to analyze the LWRs of the fish, the IRI, and the fish diversity metrics. Statistical significance was determined at $p < 0.05$ for all analyses.

5. The abundance–biomass comparison curve (ABC), derived from k-dominance curves for biomass and abundance plotted on the same graph, was used to assess community disturbance levels [16]. In an undisturbed community, the biomass curve lies above the abundance curve. For moderately disturbed communities, the curves overlap or intersect, with the abundance curve predominating. The ABC value, which quantifies the difference between the two dominance curves, was calculated using the following formula:

$$W = \sum_{i}^{s} \frac{(B_i - A_i)}{(50 \times (S - 1))}, \tag{6}$$

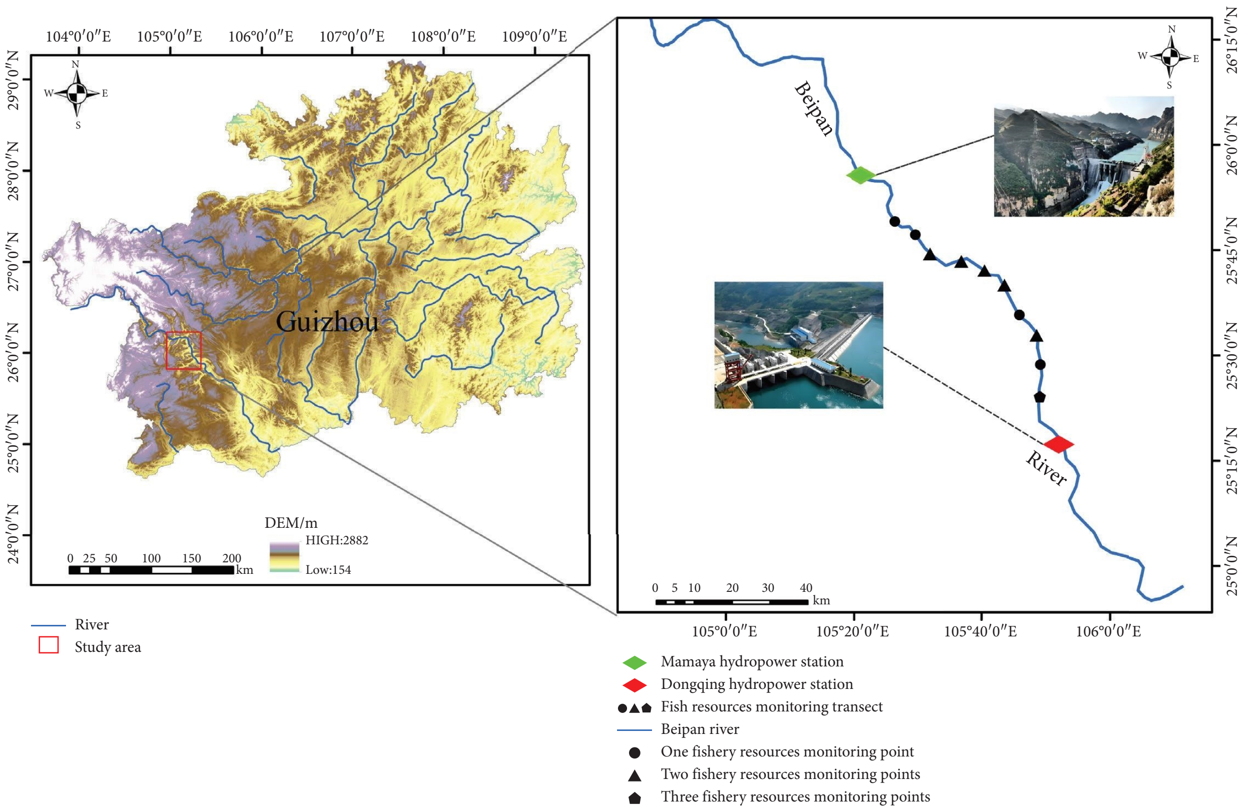

FIGURE 1: Survey area and locations of the Mamaya and Dongqing hydropower stations.

where S represents the number of fish species, B_i represents the cumulative biomass percentage of species ranked i, and A_i represents the cumulative abundance percentage of species ranked i. In this study, the percent mantissa represents the abundance, whereas the percent weight corresponds to the biomass.

2.2.3. LWR. To ensure adequate sample sizes for LWR analyses, species with fewer catches were excluded. The natural logarithm transformation was applied to estimate the LWR between SL and BW. The 95% confidence intervals (CIs) for the regression parameters a and b, as well as the coefficient of determination (R^2), were also calculated.

$$\mathrm{Log}\,(\mathrm{BW}) = \log a + b \log (\mathrm{SL}), \quad (7)$$

where BW represents the wet weight of each fish (g), SL represents the SL of the fish (cm), a represents the intercept, and b represents the slope.

Before conducting linear regression analysis, a log–log plot was used to identify and remove outliers, minimizing the fitting error of the data. Data analysis and individual figure processing were carried out using Origin Pro 2021 and SPSS 25.0.

3. Results

3.1. Current Status of Fish Resources. According to the statistical results of the fish resource survey conducted from 2020 to 2023, 2776 fish, weighing 353.89 kg, and representing 4 orders, 9 families, and 31 genera were captured (Table 1). Among these were 35 native fish species and 3 nonnative species (*Oreochromis mossambicus*, *Leiocassis longirostris*, and *Ictalurus punctatus*). The Cyprinid family dominated the samples with 2182 individuals belonging to 26 species and a total weight of 314.36 kg. This accounted for 68.42% of the total species, 56.44% of the total abundance, and 88.83% of the total weight, respectively. The Bagridae family dominated the samples with 134 individuals belonging to 4 species and a total weight of 3.94 kg. This accounted for 10.53% of the total species, 4.83% of the total abundance, and 1.11% of the total weight, respectively. The Siluridae family dominated the samples with 55 individuals belonging to 2 species and a total weight of 15.30 kg. This accounted for 5.62% of the total species, 1.98% of the total abundance, and 4.32% of the total weight, respectively. Cobitidae, Cranoglanididae, Gobiidae, Serranidae, Cichlidae, and Amblycipitidae had 1 species each.

Following the construction of cascade hydropower stations in the Beipan River Basin, the number of fish species gradually decreased, although the rate of decline slowed. Before the construction of the cascade stations (before 1989), there were 91 fish species in the study area [1, 9]. Between 2009 and 2011, this number decreased to 45 species, a reduction of 50.55% [8]. From 2013 to 2015, the number further declined to 39 species, representing a 13.33% decrease from the previous period [9]. From 2020 to 2023, the number further decreased to 38 species, a 2.56% reduction from the previous period. The fish resource survey from 2020 to 2023 revealed a slight decrease in the number of fish

Table 1: Statistical analysis of catch composition: taxonomic distribution, species proportions per family, and numeric/gravimetric metrics across fish families.

Families of fish	Number of species	Species percentage (%)	Total individuals	Number percentage (%)	Weight (kg)	Weight percentage (%)
Cyprinidae	26	68.42	2182	78.60	314.36	88.83
Cobitidae	1	2.63	29	1.04	0.10	0.03
Siluridae	2	5.26	55	1.98	15.30	4.32
Bagridae	4	10.53	134	4.83	3.94	1.11
Cranoglanididae	1	2.63	20	0.72	3.65	1.03
Gobiidae	1	2.63	121	4.36	0.51	0.14
Serranidae	1	2.63	19	0.68	4.67	1.32
Cichlidae	1	2.63	182	6.56	6.95	1.96
Amblycipitidae	1	2.63	34	1.22	4.40	1.24

species. In four consecutive years, 34 fish species, or 89.47% of the total fish population, were detected together. The highest species number was in 2021, when 37 species were monitored, followed by 36 in 2020 and 2022, whereas only 30 were detected in 2023 (Figure 2).

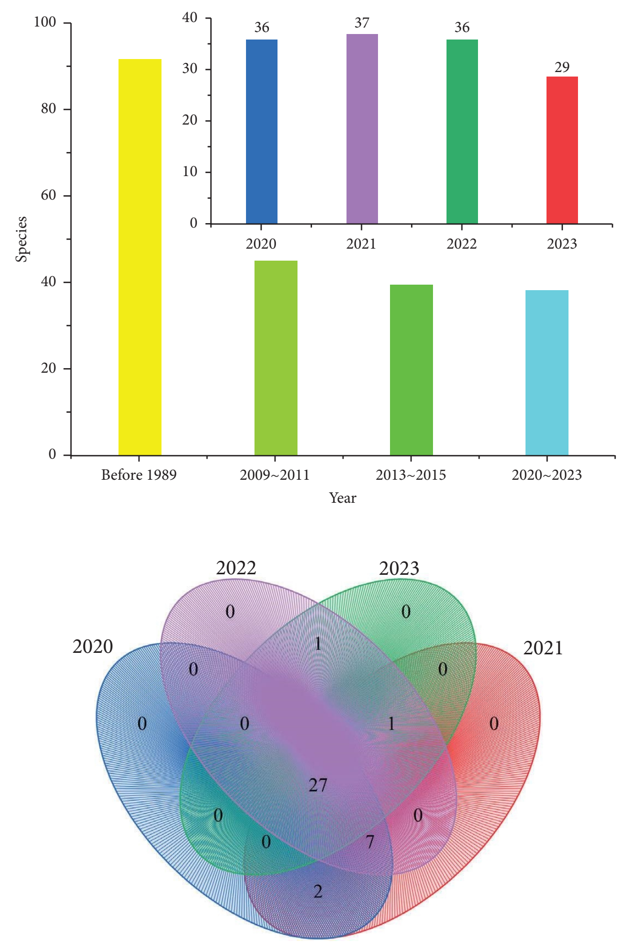

Figure 2: Number of fish species collected in different sampling years and the number of common fish species captured across different years within the study area. *Note:* Histograms represent the number of fish species in different sampling years; Venn diagrams represent the number of fish species captured in common across different sampling years.

3.2. Fish Diversity. As shown in Table 2, IRI analysis revealed that the dominant species in the study area was *Onychostoma sima,* with an IRI value of 4729.23, representing 7.46% and 25.00% of the total number and biomass, respectively. Three important species were identified, with IRI values ranging from 108.32 to 210.06, which primarily included the species *Spinibarbus sinensis*, *Hemibarbus maculatus*, and *O. mossambicus*. The survey additionally identified 10 common species ($10 \leq IRI < 100$) and 24 general species ($IRI < 10$), collectively constituting the majority of the fish assemblage.

Overall, fish diversity in the section of the Beipan River from below the Mamaya hydropower station to the Dongqing Reservoir is relatively low, with the Margalef richness index (D_{Ma}) varying between 4.790 and 5.758, as presented in Table 3. The highest values were recorded in 2021, followed by 2022, 2020, and 2023. The Shannon–Wiener diversity index (H'), Pielou evenness index (J), and Simpson index (C) all displayed a decreasing trend over the years, nearing parallel, with peak values observed in 2021, ranging from 0.064 to 0.083, 0.822 to 0.895, and 0.908 to 0.950, respectively.

The ABC curve illustrates that in the survey area from 2020 to 2023, the starting point of the biomass advantage curve was significantly greater than that of the abundance advantage curve, with W values being positive and slightly greater than 0. In 2020, the biomass-dominance curve was consistently above the abundance-biomass curve, indicating an undisturbed fish community structure with slow-growing, large individual species. From 2021 to 2023, the intersection of the biomass dominance curve and the abundance-biomass curve suggested moderate disturbance to the fish community, with increased disturbance and an unstable state (Figure 3).

From 2020 to 2023, both the cumulative abundance and cumulative biomass curves exhibited an overall increasing trend that gradually decelerated. However, when the cumulative curve rate reached 90% (indicated by the black dashed line in Figure 4), the cumulative abundance curve presented the highest number of fish species in 2020 and the lowest number in 2023. In contrast, the cumulative biomass curve peaked in 2021 and was lowest in 2023. In the cumulative abundance curve, the curve for 2023 rose more rapidly at lower species abundances than did the curves for

Table 2: Statistics on the relative importance index (IRI), numbers, weights, community statuses, and feeding habits of various fish species.

Species	IRI					Relative abundance (%)	Percentage of weight (%)	Community status	Feeding habits
	2020	2021	2022	2023	Mean IRI				
Onychostoma sima	11,498.361	2847.843	2163.163	2407.537	4729.226	7.46	25.00	I	O
Spinibarbus sinensis	9.612	57.923	251.284	521.409	210.057	1.73	7.55	II	O
Hemibarbus maculatus	90.681	158.592	209.791	27.977	121.760	3.06	5.95	II	C
Oreochromis mossambicus	5.992	8.944	406.165	12.197	108.325	6.56	1.96	II	O
Pseudohemiculter dispar	1.474	0.264	20.035	318.897	85.168	5.48	1.61	III	O
Onychostoma gerlachi	93.573	46.854	117.629	49.680	76.934	7.93	6.13	III	H
Semilabeo obscurus	2.942	198.077	1.735	/	67.585	2.38	3.89	III	H
Chanodichthys erythropterus	22.388	201.394	32.359	1.708	64.462	2.49	0.62	III	C
Hypophthalmichthys molitrix	9.383	11.136	192.930	6.630	55.020	2.95	2.79	III	O
Silurus cochinchinensis	8.259	56.988	50.321	1.360	29.232	1.69	3.31	III	C
Spinibarbus hollandi	24.720	55.603	9.639	3.773	23.434	0.94	1.08	III	O
Aristichthys nobilis	14.619	26.735	45.603	5.289	23.062	2.74	2.67	III	O
Ptychidio jordani Myers	0.485	0.309	0.258	56.087	14.285	5.84	0.93	III	O
Acrossocheilus yunnanensis	0.134	2.968	46.884	0.201	12.547	0.40	20.36	III	O
Opsariichthys bidens	0.299	5.741	2.518	22.066	7.656	4.50	0.61	IV	O
Ctenopharyngodon idella	0.255	2.531	9.761	15.650	7.049	1.19	1.52	IV	O
Acrossocheilus iridescens longipinnis	0.653	8.254	/	/	4.454	2.16	1.08	IV	O
Garra orientalis	3.016	8.192	2.209	0.403	3.455	0.65	0.77	IV	O
Siniperca kneri	4.882	2.444	0.179	/	2.502	0.68	1.32	IV	C
Ictalurus punctatus	0.621	6.464	0.133	/	2.406	1.22	1.24	IV	O
Rhodeus ocellatus	3.873	1.176	0.671	0.165	1.471	6.05	0.23	IV	O
Carassius auratus	1.482	0.413	1.408	2.075	1.345	0.94	0.98	IV	O
Rhinogobius giurinus	1.420	2.848	0.454	0.043	1.191	4.36	0.14	IV	C
Silurus asotus	/	1.021	0.597	1.871	1.163	0.29	1.02	IV	C
Pelteobagrus vachelli	0.565	0.523	0.030	2.783	0.975	2.34	0.56	IV	C
Cyprinus carpio	2.154	0.566	0.457	0.251	0.857	0.65	0.78	IV	O
Pelteobagrus fulvidraco	0.599	0.921	0.841	/	0.787	0.90	0.46	IV	C
Mylopharyngodon piceus	0.635	0.496	0.979	0.909	0.755	0.72	0.82	IV	O
Cranoglanis bouderius	1.225	0.946	0.234	0.462	0.716	0.72	1.03	IV	C
Semilabeo notabilis	0.185	0.572	0.112	/	0.290	0.72	0.67	IV	O
Crossocheilus bamaensis	0.463	0.499	0.142	0.007	0.278	2.45	0.20	IV	H
Hemiculter leucisculus	0.514	0.007	0.000	0.002	0.131	6.41	0.12	IV	C
Procypris mera	0.106	0.141	/	/	0.124	0.14	1.94	IV	O
Leiocassis crassilabris	/	/	0.171	0.034	0.102	1.44	0.04	IV	C
Discogobio yunnanensis	0.034	0.046	0.001	0.000	0.020	4.36	0.09	IV	O
Leiocassis longirostris	0.035	0.000	0.017	/	0.018	0.14	0.06	IV	C
Hemiculterella sauvagi	0.004	0.047	0.003	0.003	0.014	4.11	0.13	IV	C
Botia robusta	0.000	0.001	0.001	/	0.001	1.04	0.03	IV	O

Note: I: Dominant species; II: important species; III: common species; IV: rare species. Species with IRI values higher than 1000 were identified as dominant. It is an important species between 100 and 1000. A total of 10–100 are common species; fewer than 10 are rare species. C: carnivore; H: herbivore; O: omnivore./": The fish species was not caught that year.

Table 3: Margalef richness index, Shannon–Wiener diversity index, Pielou evenness index, and Simpson index of fish species in different years.

Year	Margalef richness index (D_{Ma})	Shannon–Wiener diversity index (H')	Pielou evenness index (J)	Simpson index (C)
2020	5.536	0.080	0.854	0.930
2021	5.758	0.083	0.895	0.950
2022	5.706	0.072	0.826	0.922
2023	4.790	0.064	0.822	0.908

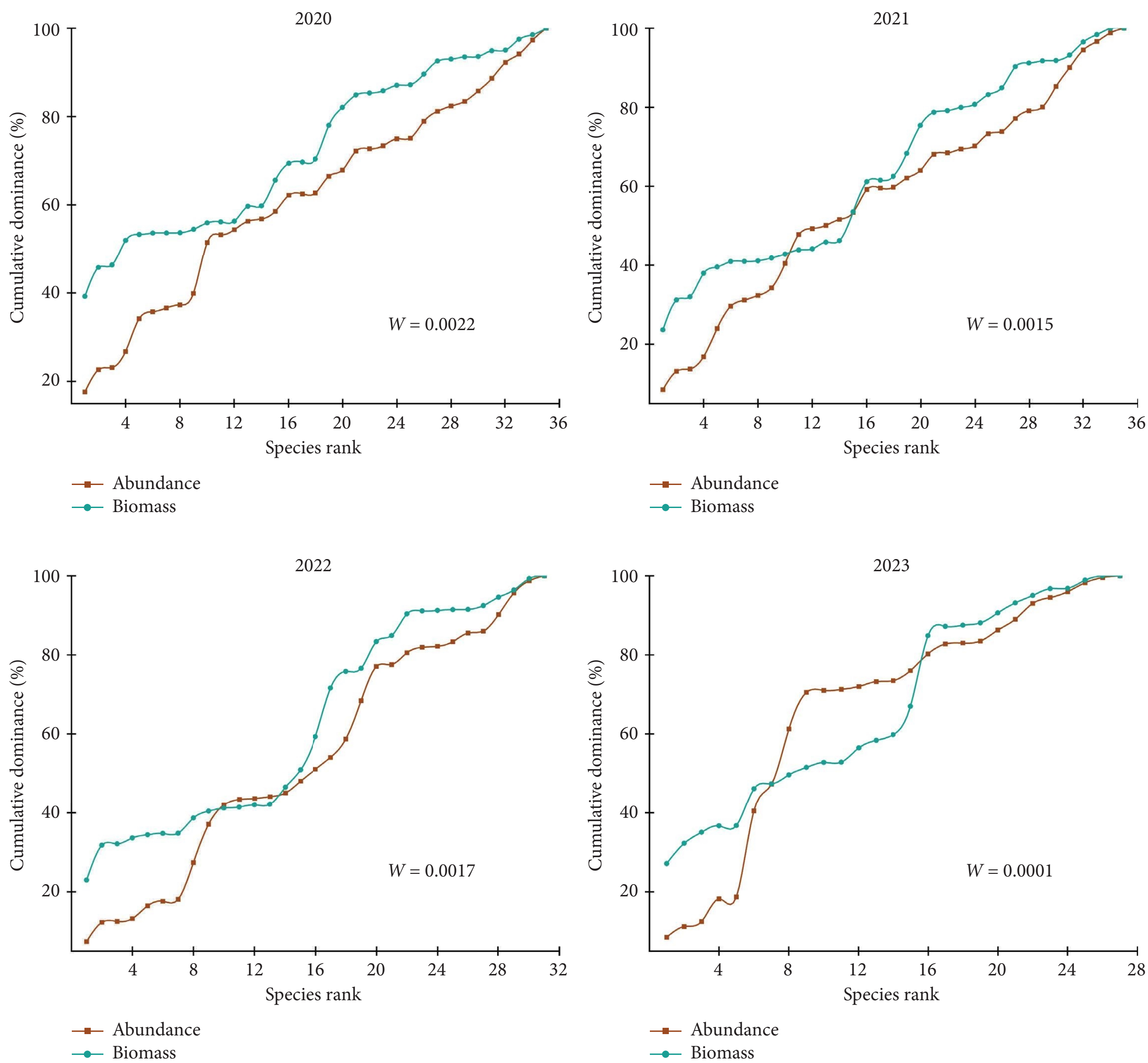

Figure 3: ABC curves and W statistic values of fish catches in the lower reaches of the survey area from 2020 to 2023.

2020–2022, which showed a more similar rate of increase. In the cumulative biomass curve, the curve for 2020 rose more slowly at lower biomass levels, whereas those for 2021–2022 accelerated, indicating an increase in biomass distribution. In contrast, the curve for 2023 flattened at both lower species ranks and higher biomass levels but overall remained below the biomass distribution levels observed from 2020 to 2022.

3.3. Samples and LWRs. As shown in Table 4, the LWRs for all the fish species were significant ($p < 0.05$), with R^2 values ranging from 0.9749 to 0.9999. The parameter b for the LWRs of the 28 species ranged from 2.308 to 3.793, which is consistent with the expected range (2.5–3.5) reported by Ma et al. [18], Yang et al. [19], and Wang et al. [20] in natural environments.

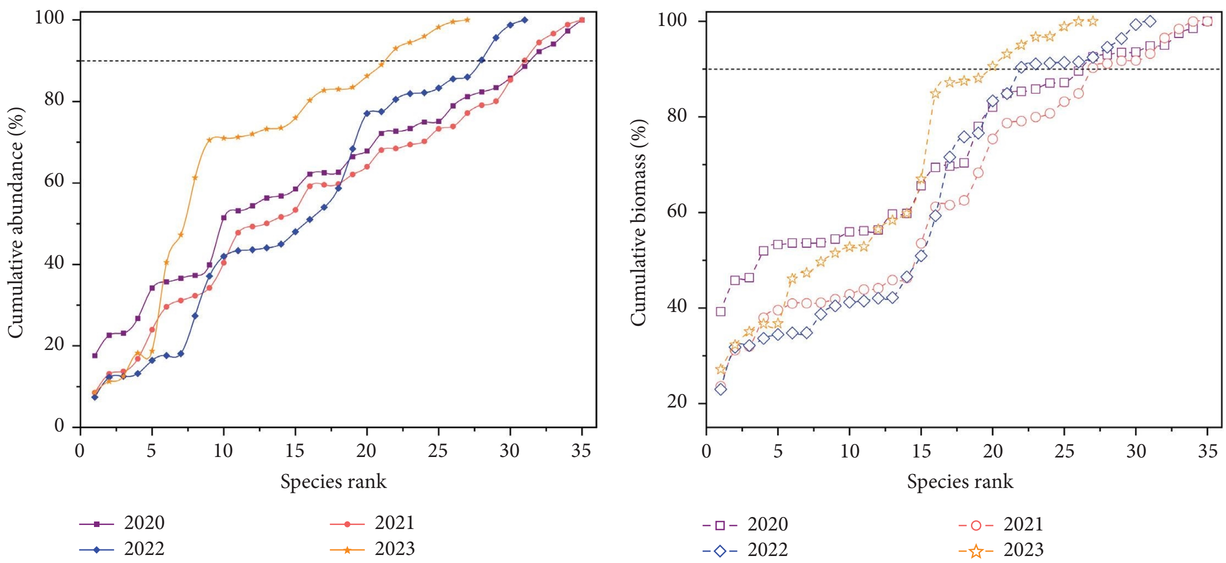

Figure 4: Dominance curves of cumulative abundance and cumulative biomass from 2020 to 2023.

An analysis of the 10 fish species with the highest cumulative catches from 2020 to 2023 and comparisons of changes in their LWR revealed significant differences in the LWR across the species ($p < 0.05$), as shown in Table 5. The results indicate that the *a* value in the LWR decreased annually, being largest in 2021 and smallest in 2023. In contrast, the *b* value gradually increased, reaching its lowest value in 2023 and its highest value in 2024. Notably, the body length and weight of these 10 fish species tended to decrease over time, with the smallest measurements observed in 2020 and fluctuations in the *a* and *b* values reflecting these changes (Figure 5). These patterns suggest potential correlations with alterations in the age structure of the fish population within the study area. A reduction in average body length and a shift in dominant age groups were noted annually, which may pose increasing risks to the population, potentially leading to severe outcomes, including extinction.

4. Discussion

4.1. Composition and Diversity Changes in the Fish Communities

4.1.1. Status of Fish Resources. A comparison with historical data revealed significant changes in the fish community of the Beipan River, primarily characterized by (1) a reduction in species. Historical records document 91 fish species in the river [9, 21]. Surveys conducted in September and November 2009 and March and May 2010 revealed 45 species [8]. From 2013 to 2015, the number further declined to 39 species, representing a 13.33% decrease from the previous period [9]. However, during this four-year continuous survey, only 38 species were collected. Despite the limited number of specimens in this study, the general principle that a greater number of species is associated with higher collection frequency supports the conclusion of a sharp decline in fishery resources throughout the Beipan River system [21]. (2) Changes in dominant species. Historical data indicate that the dominant species in the Beipan River are primarily *Schizothoracids* and *Acrossocheilus beijiangensis.* However, the dominant species is now primarily *O. sima* [22]. While silver carp remain common in the Beipan River Basin, species such as *Mylopharyngodon piceus*, *Ctenopharyngodon idellus*, and *Aristichthys nobilis* have become uncommon. Compared with previous studies on the fish species composition at the confluence of the Pearl River and the Beipan River, our study revealed a broadly similar taxonomic structure at the family level but fewer species within these families [23]. Apart from endemic species to the Beipan River, all other species can also be found in the mainstream sections of the Pearl River, indicating that the fish community of the Beipan River represents a nested subset of that of the mainstream of the Pearl River [24]. Li et al. (2014) similarly reported that higher-order streams are nested subsets of lower-order stream fish communities in their study of spatial patterns in fish assemblages in mountain streams of the Wànhé River [25]. The observed phenomenon can be attributed not only to the differentiation of local habitat characteristics but also, critically, to the isolating nature of riverine habitats, which is a key factor in the reduction in fish species diversity within families in the Beipan River [15]. In this study, the Beipan River, as a principal tributary of the Pearl River, shares significant similarities in its midstream fish species composition with adjacent sections of the Pearl River. This similarity stems from its direct connection to the Pearl River. However, differences in local habitat conditions such as water depth and flow rate, along with the construction of hydropower stations in both the Pearl River basin and the Beipan River basin, create barriers that limit the migration of certain endemic Pearl River species into the Beipan River [2, 8, 9]. Consequently, species such as *Siniperca chuatsi*, *Parabramis pekinensis,* and *Schizothorax yunnanensis* are rarely found in the Beipan River, leading to notable differences in species composition between the two rivers [26]. Compared with previous studies, the progressive contraction of fish species within

Table 4: Length–weight relationships (LWRs) of 28 fish species from the Beipan River, China.

Family	Fish species	N	Standard length (cm): SL range (cm)	Standard length (cm): Mean SL ± SD (cm)	Body weight (g): BW range (cm)	Body weight (g): Mean BW ± SD (g)	Regression parameters: a	Regression parameters: 95% CL of a	Regression parameters: b	Regression parameters: 95% CL of b	R^2
Cyprinidae	*Onychostoma sima*[b,f]	212	12.20–51.51	30.37 ± 12.14	49.30–1175.00	422.76 ± 343.84	0.1295	0.1151–0.1457	2.3078	2.3428–2.2727	0.9982
	Hemibarbus maculatus[f]	85	3.20–36.00	21.52 ± 10.74	1.00–680.30	247.65 ± 217.33	0.0228	0.0177–0.0294	2.8697	2.7848–2.9547	0.9953
	Acrossocheilus longipinnis[a,b,f]	45	14.70–30.30	23.57 ± 5.47	31.70–650.00	293.57 ± 47.66	0.0004	0.0003–0.0004	3.7932	3.7365–3.8499	0.9998
	Hemiculterella sauvagi[b,f]	94	5.80–15.20	11.96 ± 3.41	2.00–64.00	34.04 ± 22.68	0.0046	0.0028–0.0078	3.4872	3.2748–3.6996	0.9962
	Pseudohemiculter dispar[a,f]	66	4.90–51.51	15.5 ± 5.70	1.20–69.00	37.59 ± 25.76	0.0133	0.0101–0.0176	2.8077	2.7039–2.9116	0.9989
	Rhodeus ocellatus[b,f]	13	1.20–10.20	5.54 ± 3.30	0.02–16.60	4.95 ± 3.09	0.0152	0.0084–0.0273	3.0208	2.6672–3.3745	0.9898
	Opsariichthys bidens[f]	14	8.00–11.50	10.15 ± 0.99	8.44–23.80	17.14 ± 1.25	0.0159	0.0085–0.0297	3.0049	2.7353–3.2745	0.9749
	Ptychidio jordani Myers[f]	152	8.91–15.81	12.01 ± 2.22	14.70–66.21	30.6 ± 18.81	0.0039	0.0026–0.0059	3.5280	3.3901–3.6649	0.9991
	Hemiculter leucisculus[f]	168	3.04–22.46	9.76 ± 6.95	0.60–107.40	24.37 ± 21.26	0.0327	0.0287–0.0372	2.6000	2.5399–2.6601	0.9999
	Discogobio yunnanensis[b,f]	121	4.30–16.16	9.78 ± 2.13	2.35–70.52	21.81 ± 12.11	0.0550	0.0430–0.0703	2.5832	2.4751–2.6914	0.9805
	Sinocrossocheilus bamaensis[f]	58	4.20–14.85	9.78 ± 3.69	1.59–69.79	26.56 ± 23.99	0.0177	0.0114–0.0274	3.0641	2.8675–3.2608	0.9969
	Semilabeo notabilis[c]	18	12.90–42.34	27.68 ± 9.18	19.64–980.61	336.51 ± 302.19	0.0044	0.0040–0.0049	3.2858	3.2560–3.3156	0.9997
	Garra orientalis[f]	14	4.60–16.40	10.14 ± 4.62	1.90–110.18	39.24 ± 34.61	0.0187	0.0117–0.0297	3.0522	2.8493–3.2552	0.9960
	Spinibarbus hollandi[b,f]	26	4.60–40.68	26.92 ± 12.67	2.02–1421.58	638.56 ± 556.16	0.0211	0.0179–0.0250	2.9919	2.9396–3.044	0.9995
	Chanodichthys erythropterus[f]	69	6.20–57.50	26.49 ± 14.48	1.27–1095.31	199.69 ± 174.38	0.0057	0.0049–0.0066	2.9833	2.9380–3.029	0.9992
	Acrossocheilus yunnanensis[b,f]	11	4.48–44.30	30.85 ± 11.38	1.50–1700.00	766.58 ± 527.40	0.0149	0.0140–0.0158	3.0744	3.0566–3.0922	0.9998
	Onychostoma gerlachi[a,f]	212	15.95–31.05	23.81 ± 4.61	89.00–806.92	380.73 ± 226.01	0.0087	0.0076–0.0100	3.3324	3.2892–3.3756	0.9994
	Spinibarbus sinensis[f]	48	5.60–61.00	21.69 ± 18.19	2.10–2850.00	556.47 ± 521.93	0.0112	0.0081–0.0153	3.0282	2.9150–3.1414	0.9987
Cobitidae	*Sinibotia robusta*[f]	19	5.00–9.05	7.56 ± 1.08	2.70–18.00	10.92 ± 4.81	0.0108	0.0089–0.0133	3.3680	3.2683–3.4677	0.9987
Siluridae	*Pterocryptis cochinchinensis*[f]	47	13.28–29.25	22.77 ± 5.93	46.17–441.71	249.02 ± 148.32	0.0342	0.0229–0.0512	2.8019	2.6723–2.9315	0.9998
Bagridae	*Pseudobagrus vachellii*[f]	63	8.00–27.75	18.66 ± 6.36	6.14–220.86	90.12 ± 74.64	0.0155	0.0099–0.0242	2.8862	2.7315–3.0410	0.9952
	Tachysurus fulvidraco[d]	24	5.20–21.00	14.73 ± 5.35	4.80–140.08	70.16 ± 48.54	0.0725	0.0547–0.0962	2.4812	2.3744–2.5880	0.9969
	Pseudobagrus crassilabris[f]	38	4.70–19.20	9.60 ± 6.63	2.30–69.10	20.55 ± 12.48	0.0246	0.0093–0.0649	2.7521	2.3571–3.1291	0.9904
Cranoglanididae	*Cranoglanis bouderius*[a,d]	16	10.00–30.74	23.07 ± 9.16	17.00–525.00	281.11 ± 214.05	0.0284	0.0108–0.0747	2.8359	2.5218–3.1500	0.9908
Gobiidae	*Rhinogobius giurinus*[e]	121	1.87–6.50	3.89 ± 1.50	1.13–8.30	4.21 ± 2.36	0.0139	0.0132–0.0146	3.1944	0.0132–0.0146	0.9999
Serranidae	*Siniperca kneri*[f]	19	8.20–33.45	23.29 ± 8.72	15.61–500.45	245.97 ± 177.26	0.0671	0.0513–0.0876	2.5359	2.4497–2.6222	0.9977
Cichlidae	*Oreochromis mossambicus*[f]	182	8.20–24.31	15.97 ± 5.45	5.70–95.71	38.2 ± 32.87	0.0168	0.0121–0.0233	2.7063	2.5862–2.8265	0.9961
Amblycipitidae	*Ictalurus punctatus*[f]	24	8.20–49.51	29.38 ± 12.66	6.4–1200.54	399.98 ± 384.67	0.0152	0.0114–0.0204	2.8959	2.8084–2.9834	0.9970

Note: The values for a and b (length–weight relationship parameters) are provided, as are the 95% confidence intervals for a and b and the R-squared coefficient of determination in the log–log LWR. a, intercept of the LWR; b, slope of the LWR; $\log(BW) = \log a + b\log(SL)$ (SL, standard length: BW, body weight). 95% CI = 95% confidence limits (for both equation parameters); N, number of individuals; R^2, coefficient of determination.

[a]No LWR reference in FishBase [17].

[b]New records of maximum total length in FishBase.

[c]Endangered.

[d]Vulnerable.

[e]Near threatened.

[f]Least concern.

Table 5: Length–weight relationships (LWRs) of the top 10 most abundant fish species caught from the Beipan River, China.

Fish species	Year	Standard length (cm)		Body weight (g)		Regression parameters				
		SL range (cm)	Mean SL ± SD (cm)	BW range (cm)	Mean BW ± SD (cm)	*a*	95% CL of *a*	*b*	95% CL of *b*	R^2
Onychostoma sima	2020	38.52–51.51	44.17 ± 4.70	463.99–1175.00	665.55 ± 209.73	0.1311	0.1084–0.1589	2.3105	2.2603–3.3607	0.9992
	2021	32.45–46.50	39.60 ± 4.13	390.48–889.64	624.87 ± 148.80	0.1252	0.1030–0.1521	2.3115	2.2585–2.3646	0.9991
	2022	23.50–39.90	33.25 ± 5.64	174.70–590.44	403.10 ± 148.36	0.1166	0.1047–0.1300	2.3152	2.2842–2.3462	0.9997
	2023	12.20–23.50	18.54 ± 3.67	49.30–190.07	115.67 ± 48.73	0.1050	0.0900–0.1224	2.3796	2.3268–2.4324	0.9992
Oreochromis mossambicus	2020	16.80–24.31	20.34 ± 2.21	28.06–95.71	46.36 ± 12.41	0.0240	0.0224–0.0257	2.5051	2.4821–2.5282	0.9997
	2021	10.80–18.32	14.14 ± 1.95	10.41–42.38	21.99 ± 8.16	0.0193	0.0186–0.0201	2.6414	2.6262–2.6567	0.9999
	2022	8.20–14.90	11.75 ± 2.30	5.70–30.14	16.50 ± 8.21	0.0159	0.0151–0.0168	2.7838	2.7616–2.8060	0.9998
	2023	9.00–14.50	11.44 ± 1.68	6.31–24.70	13.15 ± 5.70	0.0122	0.0118–0.0126	2.8434	2.8312–2.8557	0.9999
Rhodeus ocellatus	2020	16.80–22.45	19.79 ± 1.79	53.20–107.44	80.14 ± 16.98	0.0522	0.0461–0.0591	2.4539	2.4123–2.4955	0.9993
	2021	10.50–18.20	14.30 ± 2.80	21.16–91.32	51.88 ± 26.22	0.0395	0.0374–0.0416	2.6717	2.6516–2.6918	0.9999
	2022	9.60–16.50	13.66 ± 2.11	6.24–25.85	16.48 ± 6.17	0.0159	0.0149–0.0169	2.6396	2.6154–2.6638	0.9998
	2023	3.04–15.60	9.42 ± 3.97	0.60–119.23	37.20 ± 16.24	0.0163	0.0160–0.0166	3.2399	3.2319–3.2479	0.9999
Pseudohemiculter dispar	2020	10.87–15.81	13.17 ± 1.85	18.35–66.20	38.08 ± 17.68	0.0054	0.0051–0.0058	3.4061	3.3788–3.4335	0.9998
	2021	10.01–13.25	11.27 ± 0.95	12.61–33.35	19.58 ± 5.92	0.0038	0.0035–0.0042	3.5138	3.4730–3.5546	0.9997
	2022	9.21–12.58	11.10 ± 1.05	10.64–32.52	21.59 ± 7.03	0.0040	0.0036–0.0044	3.5566	3.5163–3.5968	0.9997
	2023	8.91–12.25	10.67 ± 1.21	7.71–24.62	15.71 ± 6.29	0.0029	0.0025–0.0034	3.6054	3.5367–3.6742	0.9991
Opsariichthys bidens	2020	12.78–16.16	14.42 ± 1.00	39.58–70.51	53.71 ± 9.19	0.0954	0.0720–0.1263	2.3690	2.2636–2.4743	0.9951
	2021	12.42–15.82	14.09 ± 1.11	37.30–68.18	51.63 ± 10.05	0.0767	0.0676–0.0871	2.4568	2.4089–2.5047	0.9991
	2022	7.21–12.05	8.97 ± 1.54	8.45–32.53	15.88 ± 7.67	0.0451	0.0430–0.0472	2.6477	2.6263–2.6692	0.9998
	2023	4.30–8.30	6.78 ± 1.36	2.35–14.86	9.16 ± 4.37	0.0402	0.0381–0.0425	2.7913	2.7625–2.8201	0.9999
Rhinogobius giurinus	2020	5.91–6.50	6.10 ± 0.20	6.29–8.30	6.95 ± 0.69	0.0403	0.0333–0.0488	2.8441	2.7381–2.9502	0.9966
	2021	5.37–6.30	5.68 ± 0.35	4.79–7.78	5.75 ± 1.10	0.0316	0.0285–0.0351	2.9878	2.9277–3.0480	0.9992
	2022	2.78–6.15	4.42 ± 1.00	0.55–7.22	2.91 ± 2.00	0.0209	0.0204–0.0214	3.2129	3.1968–3.2289	0.9999
	2023	1.87–5.86	4.29 ± 1.23	1.13–4.54	2.03 ± 1.45	0.0119	0.0116–0.0121	3.3614	3.3471–3.3758	0.9999
Acrossocheilus yunnanensis	2020	12.71–15.10	14.12 ± 0.82	35.1–64.04	50.58 ± 9.57	0.0074	0.0065–0.0083	3.3325	3.2873–3.3778	0.9995
	2021	6.45–14.8	11.16 ± 2.57	2.96–51.71	23.45 ± 15.44	0.0049	0.0048–0.0051	3.4340	3.4184–3.4496	0.9999
	2022	5.80–12.30	10.09 ± 2.30	2.00–30.59	18.02 ± 10.47	0.0035	0.0033–0.0036	3.6188	3.6031–3.6346	0.9999
	2023	/	/	/	/	/	/	/	/	/
Hemibarbus maculatus	2020	12.71–34.69	24.71 ± 8.32	36.81–600.54	292.95 ± 212.29	0.0318	0.0307–0.0329	2.7754	2.7647–2.7862	0.9999
	2021	14.70–29.10	21.02 ± 6.09	47.85–321.73	155.04 ± 112.64	0.0255	0.0243–0.0268	2.8027	2.7869–2.8185	0.9999
	2022	9.71–24.10	16.02 ± 6.09	16.81–237.95	99.63 ± 90.55	0.0219	0.0211–0.0227	2.9227	2.9098–2.9355	0.9999
	2023	3.20–17.50	10.07 ± 4.30	1.63–158.89	45.77 ± 40.25	0.0188	0.0183–0.0194	2.9970	2.9839–3.0101	0.9999
Garra orientalis	2020	19.23–57.50	32.57 ± 10.86	52.11–1095.87	286.38 ± 284.39	0.0097	0.0067–0.0140	2.8934	2.7863–3.0005	0.9966
	2021	13.82–24.69	19.29 ± 3.32	14.20–77.47	40.55 ± 19.23	0.0069	0.0065–0.0073	2.9075	2.8861–2.9288	0.9999
	2022	10.12–15.65	12.23 ± 1.54	4.77–17.75	8.83 ± 3.61	0.0048	0.0045–0.0052	2.9825	2.9553–3.0096	0.9998
	2023	6.20–24.30	12.62 ± 5.68	1.23–98.86	21.27 ± 19.78	0.0034	0.0032–0.0035	3.2277	3.2109–3.2445	0.9999
Semilabeo obscurus	2020	10.85–20.50	16.01 ± 3.34	12.23–69.21	38.61 ± 19.58	0.0191	0.0181–0.0200	2.7127	2.6947–2.7306	0.9999
	2021	13.98–18.20	15.45 ± 1.34	24.01–50.20	32.34 ± 8.19	0.0170	0.0152–0.0190	2.7501	2.7097–2.7906	0.9995
	2022	11.82–15.70	13.81 ± 1.12	12.71–28.24	19.99 ± 4.55	0.0135	0.0120–0.0152	2.7745	2.7290–2.8200	0.9993
	2023	4.90–18.24	13.43 ± 4.51	1.20–54.71	28.44 ± 18.95	0.0120	0.0117–0.0122	2.9018	2.8924–2.9113	0.9999

Note: The values for *a* and *b* (length–weight relationship parameters) are provided, as are the 95% confidence intervals for *a* and *b* and the *R*-squared coefficient of determination in the log–log LWR.

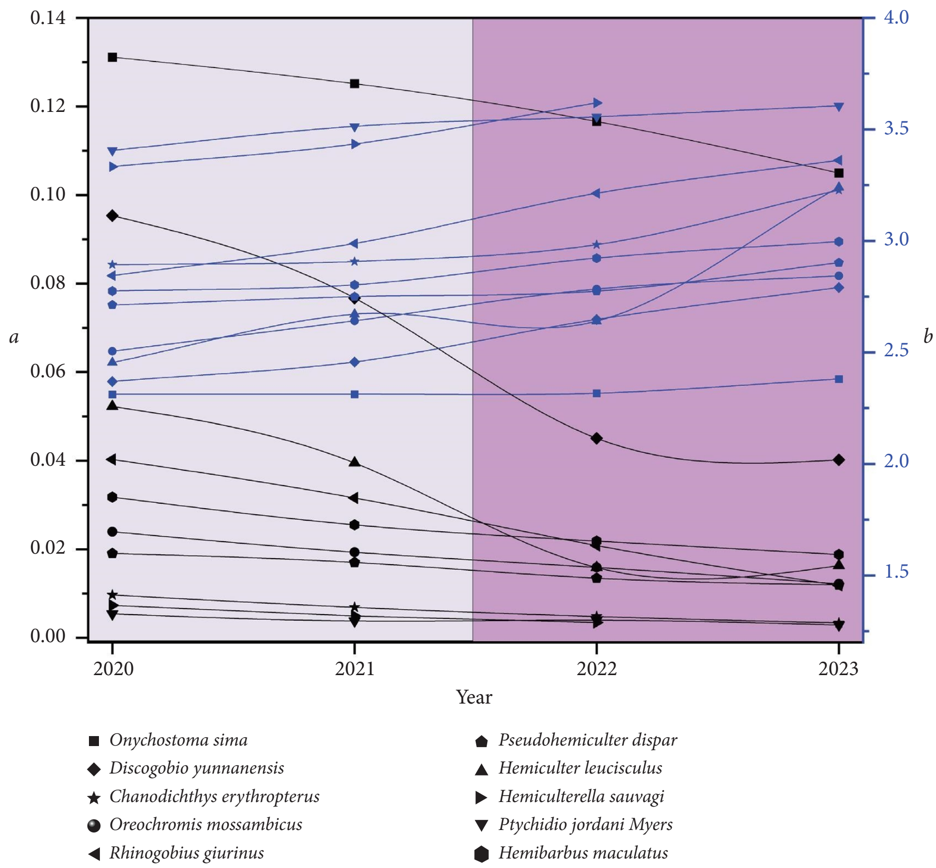

Figure 5: The variation trends of coefficient *a* and coefficient *b* in the LWRs from 2020 to 2023.

surveyed river segments likely stems from reservoir impoundment at bendway reaches, where elevated water levels and altered flow velocities eliminate original fish habitats, degrade spawning conditions and food resources, and ultimately decrease fish population abundance [27, 28].

Furthermore, studies have demonstrated that riverine phytoplankton, as primary producers in aquatic ecosystems, play vital roles in energy flow and nutrient cycling. Phytoplankton biomass is strongly correlated with fish productivity and serves as a major food source for fish within this basin [29, 30]. During hydropower operations, turbine discharges alter reservoir flow dynamics and vertical stratification, thereby modifying hydrodynamic processes and inducing water quality transformations. For example, reservoirs prioritizing power generation typically employ deep-water releases, generating conditions characterized by rapid drawdown, high discharge rates, and shortened retention times. While such conditions increase self-purification capacity, they simultaneously suppress the growth of phytoplankton, zooplankton, and algae by limiting resource availability [31]. Consequently, reduced phytoplankton taxa available as fish prey decrease fish dietary diversity, ultimately degrading fish species richness [32, 33].

4.1.2. Fish Distribution and Community Structure. Over four consecutive years of fish resource surveys, a consistent decline in fish species was observed from the Mamaya hydropower station dam site to the Dongqing hydropower station dam upper section. From an ecological standpoint, the majority of fish species in the lower reaches of the Beipan River are omnivorous [34] and primarily occupy the middle and lower water columns. This distribution likely reflects adaptations to limited food availability and fast-flowing river conditions. The development of the integrated hydropower and solar energy base within the Beipan River cascade basin has caused significant water level fluctuations in the Dongqing Reservoir and irregular discharges from the Mamaya hydropower station. These dynamics have resulted in frequent and unpredictable transitions between rapid, slow, and still water zones, influencing the spatiotemporal distribution of fish and severely impacting spawning conditions [35].

In terms of community structure, the dominant fish species were limited to a few, predominantly Cyprinidae, including *O. sima*, *Onychostoma gerlachi*, and *H. maculatus*. This represents one additional dominant species compared with the findings of Feng et al. [9]. The dominant species composition in this section of the river appears to be shaped by specific ecological pressures. Following the completion of the Mamaya hydropower station, the number of downstream habitats gradually decreased, leading to increased competition for scarce food resources and increased interspecies competition. These ecological constraints have contributed to the observed decline in the dominant fish species within the basin [36].

4.1.3. Current Status of Fish Biodiversity. The species diversity index serves as an indicator of biomass levels within an ecosystem [37]. The Margalef richness index (D_{Ma}) and Pielou evenness index (J) reflect fish species diversity and distribution patterns within aquatic systems, where higher values typically indicate greater structural complexity and stability in fish communities. This study's results align with the findings of Wang et al., with both D_{Ma} and J values falling within documented baselines, suggesting that moderately complex community structures and ecological stability persist in the investigated watershed [38, 39].

In this study, the Shannon–Wiener diversity index (H') for the fish in the surveyed river segments exhibited a consistent annual decline. It was significantly lower than the values reported by Feng et al. between 2013 and 2015 [9]. Based on Magurran's benchmark for diversity indices (1.5–3.5), the diversity index in the study area has experienced notable changes over the past decade [40]. Compared with other reaches of the Pearl River Basin, the investigated watershed has relatively low levels of fish diversity [41, 42]. Various factors may influence this variation, including reservoir operation duration [43], habitat characteristics [44], and tributary conditions [45]. During the initial stages of reservoir operation, rapid habitat alterations often lead to significant reductions in fish population diversity. Over time, as fish species adapt and invasive species establish themselves, diversity may gradually increase [45]. For example, at Brkopordo Dam in Suriname, fish diversity and evenness decreased sharply during the first 4 years of operation, but after several decades, diversity and evenness, while not reaching predam levels, exceeded those observed in the early years [46].

The persistent decline in fish diversity observed in this study may be attributed to nutrient limitations, reservoir capacity, and human engineering activities. Although a series of fish habitat protection measures were taken after the construction of the Mamaya hydropower station, the improvement in fish diversity was not significant, which may have been due to the sharp decline in the fish population and the reduction in biodiversity caused by overfishing before the completion of the dam [47, 48]. Additionally, extensive hydropower development across the Beipan River Basin has profoundly altered the ecological environment, hindering the adaptation of native species and resulting in population declines. Despite ongoing habitat restoration initiatives along the Beipan River mainstream, recovery is expected to be a lengthy process. As fish gradually adapt to the altered environment and invasive species proliferate, fish diversity may increase over time. Future research will focus on examining the relationships between fish diversity and environmental factors in the basin, identifying key influencing factors, and providing recommendations to enhance current fish habitat restoration efforts.

4.2. LWRs. This study represents the first report on the LWRs of 28 fish species in the Beipan River Basin. Based on FishBase [17] and the literature, the LWRs for *Acrossocheilus iridescens longipinnis*, *Pseudohemiculter dispar*, *O. gerlachi*, and *Cranoglanis bouderius* in the Beipan River Basin have not been previously documented (Table 4). In addition to the maximum TL in FishBase, this study records the first maximum lengths for species including *O. sima*, *Acrossocheilus longipinnis*, *Hemiculterella sauvagi*, *Rhodeus ocellatus*, *Discogobio yunnanensis*, *Spinibarbus hollandi*, and *Acrossocheilus yunnanensis*. Parameters a and b in the LWR equation ($BW = a \times SL^b$) generally fall within 0.001–0.05 and 2.5–3.5, respectively, as noted by Froese [49]. The parameters estimated in this study align with these typical ranges. When b exceeds 3.0, it reflects populations dominated by well-nourished adult samples; b at 3.0 indicates consistent growth patterns across sizes, whereas values below 3.0 suggest faster growth in length relative to weight among smaller samples [7, 50]. However, deviations in b were observed compared with findings from other studies [51, 52]. Such variations can result from environmental factors (e.g., habitat, seasonality, and spawning periods), anthropogenic influences (e.g., sample size and measurement accuracy), and physiological factors (e.g., feeding rates and maturity), leading to seasonal fluctuations in LWR parameters [20]. The variations in b among the studied species likely reflect differences in fishing methods, species, and timings of postcapture measurements. For example, the b value for *O. gerlachi* closely matches the findings of Kuang for specimens from the same river basin [53, 54]. Conversely, discrepancies were noted for other species due to differing environmental conditions at capture sites. For example, *H. maculatus* captured in various basins by Yang and Liu presented both $b > 3$ and $b < 3$ outcomes, reflecting site-specific environmental impacts [19, 55].

The condition factor is a commonly used metric derived from LWRs to evaluate the body condition and nutritional status of fish populations within a specific region. The a value in LWRs correlates with the richness of fish populations, with increases in this parameter often reflecting a more suitable habitat environment [20, 56]. Studies have demonstrated that the condition factor is directly proportional to the a value, with fish populations in regions offering superior nutritional conditions exhibiting higher condition factors and a value [57]. In this study, the analysis of the 10 most abundant fish species revealed higher a values from 2020 to 2021, followed by a decline from 2022 to 2023. This trend coincided with a reduction in average body size, indicating a shift toward smaller fish populations. Such changes may compromise ecosystem functional diversity and resilience, potentially through habitat alterations or overfishing. The construction of hydropower stations in the middle and lower reaches of the Beipan River has fragmented aquatic habitats, significantly impairing conditions for floating eggfish to complete their breeding cycles, thereby exerting severe impacts on fish populations and their structure [8]. Although habitat restoration measures have been implemented, their effectiveness remains limited, likely because there is insufficient time to achieve stable improvements in fish diversity [58]. Additionally, the release of cold water from dams during power generation may reduce condition factors and, consequently, a values. Increased fishing pressure can further influence fish populations by accelerating growth rates, thereby reducing conditions. Environmental variables, food availability, and physiological conditions also play critical roles in shaping the condition factor and LWRs parameters [59]. Variations in fishing intensity, food resources, or environmental conditions can alter growth patterns and significantly affect fish populations [60].

4.3. Influencing Factors and Protection Suggestions for Fish Populations After Reservoir Construction. The abundance biomass curve (ABC) indicates that the catch structure of the middle reaches of the Beipan River has been in a moderately disturbed state since 2021, suggesting significant human impacts on fish resources in this region. Anthropogenic activities such as hydropower construction, overfishing, and sand mining have been shown to severely affect fish community structures [15, 61]. Eleven cascade power stations are planned along the mainstream of the Beipan River, with five already constructed and operating [9, 22]. These hydropower cascades fragment the continuous river ecosystem into reservoir-river systems, significantly altering the hydrological conditions in the reservoir and downstream areas. This fragmentation obstructs migratory fish species, disrupts their reproductive cycles and life histories, and hinders communication among fish populations [2, 21]. Furthermore, dam construction damages the specialized habitats of rare and endemic fish species adapted to the unique environment of the Beipan River, exacerbating their endangered status. Cascade development in the Pearl River, for example, has led to notable declines in migratory and sticky fish species, increases in slow-flow and static fish, an increase in alien species, and a heightened risk of extinction for rare and endemic species [62]. In terms of water pollution, although certain sections of the Beipan River have historically been affected by industrial and agricultural nonpoint source pollution, intensive industrial pollution remediation efforts by governmental and local authorities between 2000 and 2015 significantly improved the water quality within the basin. The water quality now generally meets Class II–III standards, resulting in a less pronounced impact of pollution and water quality on fish resources [63]. With respect to artificial sand mining, some areas of the Beipan River basin experience illegal rock and sand mining activities, which degrade fish habitats and directly harm fish populations. Consequently, sand mining in the lower reaches of the Beipan River is likely to severely impact fish resources in this region. The data presented in Table 2 indicate that the majority of large- and medium-sized fish captured in the middle reaches of the Beipan River are juvenile, highlighting the severe consequences of overfishing in the surveyed area. Field investigations and interviews with fishermen revealed a high frequency of electric fishing within the Beipan River basin, with reports of 2–3 groups of electric fishing personnel operating simultaneously in river sections spanning 2–3 km. Thus, overfishing in the middle reaches of the Beipan River is identified as a significant factor contributing to the depletion of fish resources in this area.

The construction of the Mamaya hydropower station obstructed fish migration routes, decreased habitats for both target and nontarget fish species downstream, modified the hydrodynamic conditions of numerous historic spawning grounds, and ultimately led to a reduction in the reproductive capacity of mainstream fish [45]. Research on the biodiversity of the Porto Primavera Reservoir of the Amazon River suggests that tributary habitat substitution and artificial ecological restoration are crucial strategies for restoring fish diversity in the main river channels [57, 59]. The Beipan River benefits from two strategically positioned tributaries, the Dali Tree River and the Ximi River, which enhance tributary habitat substitution and ecological restoration, vital for improving fish diversity in the main channel. The Pearl River, a vital water source and aquatic biodiversity hotspot in China, provides excellent living conditions and breeding grounds for rare and endangered species as well as important aquatic economic species [15]. As a significant tributary of the Pearl River, the study of fish diversity in the Beipan River is crucial for restoring fisheries resources and protecting the aquatic ecosystem within its watershed. However, due to factors such as dam construction, sand mining, and overfishing, the fish resources in the Beipan River have shown a declining trend, characterized by reduced proportions of economically valuable fish, miniaturization, and juvenilization of fish populations. To increase fish diversity conservation, we suggest that the managers of the Mamaya and Dongqing hydropower stations and other cascade managers pay attention to the following aspects:

1. Establishing aquatic germplasm resource reserves is an effective approach to fish resource protection and plays a significant role in fisheries conservation. Currently, there is the "National Aquatic Germplasm Resource Reserve for Endemic Fishes in the Jiuwan Section of the Beipan River." We suggest further designating ecological red lines based on the unique habitat characteristics of the Beipan River, targeting rare and endangered endemic fish species in key habitats to prevent habitat loss and degradation.
2. Fisheries regulatory authorities should enforce fishery laws ("*Regulations of the People's Republic of China on the Protection of Aquatic Wildlife,*" "*Fisheries Law of the People's Republic of China*") and fishing moratorium regulations strictly, intensify law enforcement, severely crack down on illegal fishing practices such as poaching and electrofishing, and eliminate unlawful fishing activities.

   Research should be conducted on fish migration corridors and the restoration of spawning grounds to promote river connectivity, hydraulic regulation, and the ecological restoration of spawning grounds. Existing cascade hydropower stations with fish passage facilities should be retrofitted, or collection and transportation fish systems should be added to facilitate fish population exchange. Hydroecological scheduling is implemented according to the needs of fish reproduction.

3. For rare and endangered endogenous fish species, such as *Semilabeo obscurus, Ptychidio jordani Myers, C. bouderius, and O. gerlachi*, fish conservation centers should be established, while scientific research, monitoring, rescue, and breeding programs for these species should be conducted.
4. Given that the main breeding season for most fish species in the Beipan River is from April to September, sand mining activities should cease during this period. Sand mines should be located on the basis of ecological surveys and analysis results, avoiding establishment within spawning grounds. Where possible, sand mines should be placed upstream of concentrated spawning areas to minimize impacts on fish populations.

Data Availability Statement

The data used to support the findings of this study are included in the article.

Conflicts of Interest

The authors declare no conflicts of interest.

Author Contributions

Yongmeng Wang: conceptualization, methodology, formal analysis, writing–review and editing, and writing–original draft. Zaixing Zhao: conceptualization, methodology, and data curation. Jian Shen: data curation and supervision. Meng Wang: validation and methodology. Zhijun Jin: supervision, resources, and investigation. Chenyu Lin: formal analysis, writing–review and editing, supervision, and funding acquisition. Xiaotao Shi: resources and investigation. Hao Chen and Yu Tao: supervision, investigation. Hao Xia: project administration and investigation. Li Chang: writing–review and editing, supervision, and funding acquisition.

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