Chapter 2

Ecological Indicators for Surface Water Quality - Methodological Approaches to Fish Community Assessments in China and Germany

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2.1 Introduction

This chapter provides an overview about the implemented water quality assessment regulations and methods with a special focus on Germany and China. In the first part, the general situation and the currently used ecological indicators are presented. In the second part, we exemplarily discuss the fish community assessment by different methodological approaches. Specifically active and passive fishing techniques are compared, which both, single or in combination, are used in Chinese or European fish monitoring surveys.

2.1.1 *The Need for Ecosystem Health Assessment*

Human activities and consequently the impact on freshwater ecosystems has increased significantly within the past decades, resulting in an unneglectable decline in biodiversity and ecosystem integrity worldwide [13]. As a matter of fact, aquatic habitats associated with 65% of continental discharge have been recently classified as moderately to highly threatened [52, 60, 61], which demonstrates that aquatic ecosystems have become highly dysfunctional due to high anthropogenic pressure. As summarized by Maloney [35], there has been a 81% global decrease in freshwater

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vertebrate populations over the last 30 years (see also [36]) and an average of 60% of protected species and 77% of habitat types are considered to be in unfavourable conservation status in Europe, with an even higher proportion in rivers, lakes and wetlands [19]. Furthermore, as the human population grows, human contact with and impacts on aquatic ecosystems increase. At present, more than 50% of the human population lives within 3 km of a freshwater system [26]. This has raised concerns over the impacts of anthropogenic activities on aquatic environments [59]. During the past centuries, the requirements for human economic development were often prioritized, sometimes neglecting negative impacts of human activities on aquatic ecosystems and the services they provide. Progress to align both, development and protection of freshwater resources was made during the last decades and the urgent necessity to preserve and restore a good ecological status of aquatic habitats has been recognized [19]. Various monitoring programs to determine water quality status and need for action were developed and implemented worldwide, with the USA and also European countries such as Germany. Recently, countries with a rapid economic development, such as China, are increasingly implementing monitoring tools for ecosystem health status assessment for environmental protection and to preserve biodiversity.

2.1.2 Current Situation in China and Germany

China's rapid economic and social development during the last decades resulted in increasing anthropogenic pressure on freshwater resources. Therefore already in the 11th Five Years Plan (2006–2010) China consistently improved its measures to control pollution and protect the environment within their major watersheds. Major threats to aquatic ecosystems in China are mainly caused by large scale hydromorphological modifications, water scarcity and severe water pollution from industrial, agricultural and municipal emissions. According to the 2017 Report on the State of the Ecology and Environment in China, 23.8% of 1,940 monitored surface water sections were rated with grade IV and V –utilizable for industry, agriculture and recreation without skin contact only– whereas 8.3% did not meet grade V standards –not suitable for human contact or any use– (Table 2.1; [38]). As a developmental goal it was legally implemented in China to improve surface water quality until 2020, with less than 5% classified as grade V or worse, and more than 70% shall meet grade III or better [42].

Historically, European countries such as Germany had a long history of water management and protection. European countries were facing similar environmental problems during their rapid economic development in the 20th century. In the 1970s, when those countries' surface waters were highly polluted, measures were implemented to improve the environmental conditions sustainably, e.g. by improving waste water treatment and regulating emissions into surface waters. A Europe-wide approach was finally implemented in 2000 by the European Union, the EU Water Framework Directive (WFD) [16], where the good ecological status or potential

Table 2.1 Chinese water quality grades and respective water uses (Source: [38])

Grade	Use
I–II	First-class protection zones for drinking water resources; habitats of rare aquatic species; fish and shrimp spawning grounds
III	Second-class protection zones for drinking water resources; fish and shrimp overwintering ground; migration channels; aquaculture areas; swimming sites
IV	Industrial use; recreational purpose (without skin contact)
V	Irrigation (agri-/horticulture, landscape)
V+	Hardly any function except regulating local climate

for surface water bodies was defined as overall goal to be reached by the year 2015. Nevertheless, European countries have largely failed to reach the initial goals in water quality status of surface waters by today. Only 7.7% of Germany's rivers reached the European Water Framework Directive's binding environmental objective of good ecological and chemical status by 2015. 34.4% of the 8,995 monitored sites exhibited a poor ecological status—major deviations from surface water body type-specific values of biological quality elements and biological communities. Further 19.9% were classified as bad—severe alterations to surface water body, type-specific values of biological quality elements and absence or strong degradation of the ecological integer biological communities.

While Germany utilizes biological, hydromorphological and physico-chemical metrics to determine water quality since the early nineties of the last century, China's water monitoring is predominantly based on physico-chemical variables using a dense net of automated monitoring stations covering the main river basins. Those analyses are the most direct approach to define chemical water quality, but on the other hand they are of limited use to allow predictions about aquatic organisms' responses to pollution nor pollution-related influences on ecosystem level [50]. Biological monitoring can provide insights into the organism-habitat relationship by means of indicator species. The presence or absence of such species enables the detection of pollution and the evaluation of biological threats to ecosystem health [23]. Since aquatic organisms cope with chemical, physical and biological habitat alterations over their entire lifecycle, they additionally allow identification of the long-term ecological status of environments. Furthermore, steep pollution and disturbance gradients are appropriately determined by biological data [21]. Acknowledging the relevance of ecological methodologies for water quality assessment, China launched a pilot national monitoring program to assess ecological integrity in 2010.

2.1.3 Ecological Indicators of Ecosystem Integrity

The biological integrity of freshwater ecosystems, here river or lotic systems, is typically considered by separately assessing so-called biological quality elements (QEs)

such as fish, macroinvertebrates, macrophytes and algae. In combination these quality elements are used to create a comprehensive picture of the current status of the respective system with the component of the worst assessment determining the status class (one-out-all-out principle). Each of these biological groups can be utilized to derive structural indicators according to their presence or absence in the assessed ecosystem [45]. This allows to draw conclusions about the systems current condition, e.g. good, bad or poor, and additionally facilitates the comparison of the current status with past and future conditions for monitoring purposes. Structural indicators are used to determine ecosystem integrity and allow to draw conclusions about the stability of the ecosystem, e.g. the capacity of an ecosystem to maintain its organization, in rivers mostly linked to river dynamic processes such as sediment relocation or dead-wood dynamics. From a scientific point of view, most commonly used indicators are, for example, species richness and diversity, characteristic species assemblage or trophic structure [35]. For monitoring purposes these attributes are especially useful since their assessment can be realized by point-in-time measurements within the respective habitats [43]. Diversity for instance is measured by sampling organisms from a specific community (e.g., fish, aquatic invertebrates, macrophytes) and considering the number of unique taxonomic groups and the number of organisms per group. By applying a standardized sampling design following pre-determined methodological protocols a spatial and temporal comparability is warranted. In river ecosystem assessment, the fish species diversity at a representative site is a structural metric that reflects ecosystem integrity and availability of typical habitats as well as habitat connectivity. Variations in diversity are identified by comparing the current abundance and density of aquatic species in an ecosystem with that of a reference site or historical ecosystem state. Furthermore, this metric can be used to determine potential risks to ecosystem integrity if significant changes in biological structure are detected or specific taxa are missing [31], or on the contrary to measure restoration success [45]. Other commonly used structural metrics include biomass, species richness or dominance or presence or absence of indicator species. Hydromorphological and water physico-chemical criteria are additionally used to aid the biological metrics.

2.1.4 Biological River Assessment Approaches Applied in China and Germany

Legal and institutional frameworks as described in the previous Chapter 1 highlight that management, protocols, analyses and reports of river monitoring results vary between states, commissions, districts and municipalities. Therefore, the comparison of quality elements (QEs) used in river assessments in Germany and China is based on surveys carried out nation-wide. This includes China's Environmental Quality Standards for Surface Water [39] and river monitoring under the WFD in Germany.

2.1.4.1 China

China's rivers are assessed according to the national Environmental Quality Standards for Surface Water [39]. This standard lists 109 chemical and physical parameters of which 24 are classified as basic parameters, five as supplementary parameters, and 80 as specific parameters. Basic parameters are implemented for rivers, lakes, channels, and reservoirs, whereas supplementary and specific parameters are applied to drinking water resources. The standard's basic parameters comprise heavy metals, nutrients, water temperature, pH value, dissolved oxygen, chemical oxygen demand (COD), and biochemical oxygen demand (BOD). For each of the basic parameters, the standard sets thresholds, pursuant to the standard's surface water quality grades (Table 2.1). The grade is attributed on basis of the one-out-all-out principle. Thus, the worst grade attained by any considered parameter represents the final water quality grade. Grade I and II are generally in agreement with the WHO-Drinking Water Guideline values and most recent Chinese Drinking Water Standards [18]. The standard further provides general guidelines pertinent to monitoring procedures and mandate local WRABs to implement and supervise its adaption.

The research process on river health in China was initiated in 1992, when [67] described water quality of Jiuhua River using the Ephemeroptera, Trichoptera, Plecoptera (EPT) and the Family Biotic Index (FBI). Since the early 2000s, researchers in China [62, 63, 70] started to focus on urban river health assessments in relation to China's fast urbanization process. Furthermore, the administrative agencies responsible for the Yellow River, Yangtze River and Pearl River Basin suggested concepts to promote river health [22, 30, 69]. Broader studies on the ecological status and water quality of rivers based on macroinvertebrates were conducted and pilot schemes for monitoring and assessment of those were implemented. These include, amongst others, [28, 44] and [64]. Evidently, the biological assessment of rivers is gradually brought to attention of Chinese river management. Up until the year 2019, no nation-wide assessment program exists, but this is supposed to be changed in the future. The utilized structural metrics for biological quality elements, currently applied mainly for academic purposes, include aquatic macroinvertebrates, fish and benthic algae. For macroinvertebrates the calculation of "family richness", "macroinvertebrate BMWP score" and "EPT family richness" is used. For fish communities the "fish species index", "fish biotic index" and "fish Berger Parker index" are used and for benthic algae the "algal biotic index" and "algal Berger Parker index".

2.1.4.2 Germany

For several years, Germany has been monitoring its running waters according to the demands of the WFD, requiring the assessment of QEs as described below. Hydro-morphological and physico-chemical QEs support biological QEs, but cannot replace them. The river status class (Table 2.2) is obtained from a ratio, which represents the relationship between the values of biological parameters observed for a given stream or river and the values of these parameters in the undisturbed reference condition

Table 2.2 General definition of rivers' ecological quality in Germany (adapted from: EC 2000)

Status	Normative definition
High	Physico-chemical/hydromorphological quality elements No, or only very minor, anthropogenic alterations to river type-specific undisturbed reference conditions Biological quality elements Values reflect river type-specific undisturbed reference conditions, and show no, or only very minor, evidence of distortion; river type-specific conditions and communities
Good	Biological quality elements Low levels of distortion resulting from human activity are evident, but deviate only slightly river type-specific undisturbed conditions
Moderate	Biological quality elements Moderate deviations from river type-specific undisturbed reference conditions; moderate signs of distortion resulting from human activity; significantly more disturbed than under good status condition
Poor	Biological quality elements Major deviations from river type-specific undisturbed reference conditions; relevant biological communities deviate substantially from those of river-type specific undisturbed reference conditions
Bad	Biological quality elements Severe alterations from river type-specific undisturbed reference conditions; absence of large proportions of relevant biological communities of river-type specific undisturbed reference conditions

pursuant to the river or stream type. Reference conditions have been defined for 25 stream and river types [51]: four types for the eco-region of the Alps and Alpine foothills, eight types for the Central German Highlands, nine types for the North German Lowlands and four ecoregion-independent types. The worst assessment result of a biological QE determines the overall assessment result (one-out-all-out principle).

Aquatic macroinvertebrates are indicators of local environmental conditions. Since 1991, the method of assessing water quality and the organic impact in running waters based on macroinvertebrates has been regulated by the German standard DIN 38 410 [8]. It is based on one of the traditional saprobic procedures [47]. Samples are taken accordingly to the international standard DIN-EN-ISO 10870 [9]. Calculation, classification and presentation are also standardized. The ecological status can be determined with the assessment system PERLODES [37]. More detailed information for the assessment of running waters using macroinvertebrates as passive indicators is given in [23].

The analysis of macrophytes is a particularly well-suited tool for monitoring long-term trends in trophic conditions [53]. Nationally, the multi-habitat-sampling of macrophytes follows a standardized protocol. Besides coverage, vitality, and sociability of each species, environmental data such as shading, flow velocity, average water depths and substrate type are recorded. In Germany, the assessment of the

subcomponent macrophytes is enabled by the assessment system PHYLIB [2]. Alternatively, the NRW-procedure [57] can be used for the assessment of rivers based on macrophytes. Both assessment methods define states of potentially natural vegetation by means of a community approach. The determination of river water quality stems from the occurrence of non-typical vegetation units. PHYLIB is also applied to the subcomponent phytobenthos. Sampling and microscopy follow standardized protocols. During sampling, species coverage and substrate types are recorded. The microscopy protocol requires the determination of taxonomic unit and each unit's frequency of occurrence.

The establishment of a nationwide classification and assessment system of streams using phytoplankton was firstly initiated by the adoption of the WFD. Standardized methods for sampling, handling of samples, and analysis are described in the manual by [40]. It is restricted to medium-sized and major rivers. For the assessment of plankton-dominated streams, the software PhytoFluss is available.

Fish are sensitive to modifications of river morphology and lateral as well as longitudinal river channel connectivity. The survey of fish population characteristics is performed by electrofishing. Six QEs are standardized recorded: species and guilds inventory, species abundance and guilds distribution, age structure, migration (index-based), fish region (index-based), and dominant species (index-based). Those QEs (metrics) are compared to a priori defined river-type reference fish biocenosis which is based on expert knowledge. Depending on the deviation of observed values of QEs from values of the reference, scores are achieved. The average score determines the status classification. The methodology in Germany is based on [15]. The assessment of the subcomponent fish is performed by application of the assessment system fiBS [14].

In contrast to biological elements, hydro-morphological QEs are not primarily decisive for assessing the status of a river, but are determining the distribution as well as habitat heterogeneity, availability and niches for biotic communities. Hydro-morphological QEs of the WFD are morphology, hydrological regime and continuity. The quality element morphology does not only focus on the riverbed (course development, longitudinal profile, bed structure), but also on the banks (cross-section, bank profile) and the floodplain [6]. Rivers are assigned to five structural classes. The allocation to the classes is based on the extent by which a river's morphology deviates from river-type-specific reference condition. A river's hydrology is classified using the parameters connection to groundwater bodies, discharge, and discharge dynamics. On the basis of data and expert knowledge, the intensity of given pressures, such as water abstractions or engineering measures, are related to the potential natural status. The classification into five classes (Table 2.2) is based on the one-out-all-out principle [7]. The procedure is currently undergoing practical trials [20]. A method to classify the continuity of watercourses is being developed by LAWA [20].

According to Annex V of the WFD, five physico-chemical quality elements of running waters need to be monitored: visibility, temperature, oxygen, conductivity, acidification, and nutrient conditions. Water body type-specific background values (very good status) and threshold values (good status/good ecological potential) are listed in Annex 7 to the Ordinance on Surface Waters. Measured values determine

the allocation to quality classes. High status is achieved if measured values adhere to river-type-specific background values. If measured values are within a range which guarantees river-type-specific correct functioning, a good status is achieved.

2.2 Methodological Comparison of Fish Community Assessment

As described previously, there are several advantages when biological metrics are complementary considered in surface water health assessment and monitoring. Since fish assemblages are considered to be a highly suitable metric for assessing river health [1, 24, 25], there are ongoing activities in China, predominantly on the academic level, to further develop and adjust fish-based indices for Chinese surface waters. Example applications in China using the “index of biological integrity” (IBI) for river health assessment were reported for the Yangtze River [33], the Liao River [48], and some local small rivers [22, 32]. Structural metrics such as the IBI demand a predetermined sampling strategy and the proper choice of methods to be applied. The choice of efficient sampling techniques is thereby a crucial factor for the success of sampling strategies [5, 41, 45].

The following section of this chapter provides a methodological comparison of fish communities along a 70 km stretch of the Fan River, Lioaning province in China. Specific objectives are the comparison of (i) Active vs. Passive sampling methods, (ii) German vs. Chinese sampling approach and techniques, (iii) Determination of fishing technique effectivity for overall community characterization.

2.2.1 Methodology

2.2.1.1 Choice of Sampling Methods

In contrast to less mobile organisms such as plants and some macroinvertebrates, the assessment of fish communities relies on efficient and non-selective sampling techniques [4, 5]. Since fishing techniques are applied from the early development of human societies on, there is a great variety of different active and passive fishing methods. Today applied standardized methods include electrofishing [27, 41, 49], hook-and-line methods, net-based methods and the use of traps [4, 5, 34, 46, 68]. Catch rates, selectivity patterns and species specificity may differ strongly, however, depending on the sampling method chosen. Thus, an accurate characterization of populations can be difficult [11]. Additionally, the efficiency of the respective fishing method most likely varies with the change of stream size as well as turbidity, current velocity and conductivity [24, 41, 54]. Assessment methods currently applied in

China comprise electrofishing for wadeable [22] or gill-netting for non-wadeable streams [17], as well as casting nets and seining [29]. Standardization of assessment is often not reported or impossible when investigations include catches from local fishermen [29].

2.2.1.2 Area of Experimental Investigation

For the methodological comparison, 6 characteristic sites within the Fan River were chosen together with the Chinese project partners (N 42.10826, E 12.448.517 – N 42.257.50, E 12.360.941). The selection of the sampling area was based on the bilateral workshops held between the German and Chinese partners of the SINOWATER project at the Chair of Aquatic Systems Biology in Freising, Germany in 2016. The Fan River is a tributary to the Liao River, one of the seven major river basins in China. Fan River is a well suited model stream, since within less than 100 km the headwater region and the confluent to the Liao River main stem can be reached. The sampling area covers a gradient from wadeable sections with low-flow conditions at the most upstream site to non-wadeable conditions at the confluent and an elevation gradient from 51 to 370 m above sea level (upper sites dominated by agriculture in the catchment, lower sites with influence of the city of Tieling and water abstraction for irrigation + large reservoir in-between). Site ID and pictures are given in Table 2.3. Sites 3–6 are characterized by a major increased discharge from Zhenziling Reservoir above site 3.

2.2.1.3 Description of Sampling Methods and Design

Herein three active and three passive methods were used for fish community assessment. The survey was conducted in May 2017, at this time of the year typically no extreme flood or drought events occur, which would have hampered sampling activities. Active methods comprised single-pass electrofishing as typically applied in Germany and China as well as seining. As passive methods common minnow traps were used as well as gill-netting and longline-fishing. Specifications of the respective methods are listed in Table 2.4.

The sampling design used in this investigation allowed to simultaneously apply the different fishing methodologies within the same section of the river, including replication of each of the methods and coverage of structural variation. Sampling sections were chosen to represent a wide range of environmental conditions from fast-flowing upstream sections to slow-flowing downstream sections. At each sampling site, a 500 m stretch was selected and divided into three main replicate sections of 150 m each. Within these 150 m sections, there were 30 m assigned for each of the active methods (electrofishing German (EG), electrofishing China (EC) and seining (SN)). Passive fishing methods, longline-fishing (L) and traps (T) were spaced

Table 2.3 Sampling sections for the methodological comparison in Fan River, Liaoning province, China

Site 1 - JiaHeChang	Site 2 - BaiQiZhai

Site 3 - XiaoTun	Site 4 - LaoBianTai

Site 5 – FanheBridge (City of Tieling)	Site 6 - HuangHeZi

in between the active fishing sections as well as at the most upstream and most downstream end of the whole sampling site resulting in 5 replicates per method. Gill-netting was only applicable at the most downstream site 6. A graphical representation of the sampling design is given in Fig. 2.1. At the most upstream part a 50 m section for time-based electrofishing was located. For each specimen, the total length (TL) was measured to the nearest 0.1 cm, total weight (WT) was measured with 1 g accuracy on a representative subsample of fish.

Table 2.4 Specification of applied active and passive fishing methods

Method type	Specification	
Active		
Electrofishing German (EG)	Graßl ELT62, 3 kW (300 V) engine-powered backpack-electrofisher connected to 1 handheld anode with a 30cm dipnet and a passive braided wire copper cathode. Fish were collected by an assistant person with a dipnet and transferred to a bucket.
Electrofishing Chinese (EC)	Two copper electrodes on telescope handles, one equipped with a 20 cm dipnet, powered by a 24V battery. Fish were collected by the operator and collected by a second person carrying a bucket.
Electrofishing Chinese - time based (EC30)	Same equipment as EC. Sampling was conducted by fishing a zigzag route across the sampling section for 30 minutes.	
Seining	15 m net, mesh-size 10 mm, lead-bottomline. Spreading of the net in 90° angle to the river bank towards the middle of the river, then pushing the net back in a half-circle towards the bank
Passive		
Trapping (T)	Minnow trap, diameter 30cm, length 60 cm. Commercial cat-food used as bait. Placed in deeper sections with the whole trap being submerged.	
Longline (L)	5 m fishing line -3 types barbed hooks (#1, #6, #10) replicated 5 times. Each hook size was equipped with different bait: fish filet, mussel meat or worms, and maggot.
Gillnet	20 m long, 1 m wide, 2 cm mesh. Brought into a stagnant deep area by boat.

Fig. 2.1 Schematic representation of the sampling design applied. EG: Electrofishing German, EC: Electrofishing China, EC30: 30 min Electrofishing china, SN: Seining, L: Longline fishing, T: Trapping, 1–3: replicate sections, MI: additional zone for macroinvertebrate sampling

2.2.1.4 Calculated Metrics

For comparison of the method-dependent catch, the following metrics are reported: Species abundance, Species richness (d), Shannon diversity index (H'), Evenness (J') Condition indices for each species (BMI), species biomass. To express the effectiveness of each method, the Catch per Unit Effort (CPUE) was calculated. For comparability the CPUE per 10 min is presented. Richness was calculated by counting all the taxa for each site, specific calculations are given below.

$$H = - \sum_{i=1}^S P_i \log_2 P_i \quad (2.1)$$

where S is the total number of taxa, P_i is the relative abundance of species i .

$$BMI = \frac{(W_T \times 100)}{L_T^3} \quad (2.2)$$

WT: Total weight (nearest g); LT: Total length (nearest 0.1 cm).

$$CPUE = \frac{N_i}{t_i} \quad (2.3)$$

N_i : Number of individuals per replicate section i ; t_i : time of fishing at replicate section i

$$d = \frac{S - 1}{\log_e N} \quad (2.4)$$

$$J' = \frac{H'}{\log_e S} \quad (2.5)$$

$$1 - \lambda' = 1 - \frac{\{\sum i N_i (N_i - 1)\}}{\{N(N - 1)\}} \quad (2.6)$$

S: Total number of taxa; N: Total number of individuals.

2.2.2 Results

2.2.2.1 Fish Community

A total of 33 fish species from 10 different families comprising 7799 specimen were collected by all methods (and sites) combined. Most of the sampling sites within the Fan River system showed to be dominated by Cyprinidae, comprising up to 85% of all fish specimen caught, followed by Nemacheilidae, Cobitidae and Gobiidae. The fish community was generally characterized by generalists like *Rhynchocypris lagowskii*, *Barbatula nuda* and *Zacco platypus*. These first two species represented 21.9% and 23.7% of the total fish biomass, respectively. They were found at every site except the most downstream site 6, HuangHeZi, where the fish assemblage was dominated by *Carassius auratus* and *Abbottina rivularis*, most likely due to the vicinity of the Liao River main stem. In general, no apex predators or fish larger 16 cm (total length) were caught by the methods applied or are simply missing in Fan River. A full list of the Fan River fish community is presented in Table 2.5. *Cobitis granoei*, *Pseudorasbora parva* and *Zacco platypus* were detected at all sites. Some species such as *Aphyocypris chinensis* (site 2), *Chanodichthys erythropterus* (site 6), *Hypomesus olidus* (site 6), *Opsariichthys bidens* (site 2), *Romanogobio tenuicarpus* (site 6) were only detected at single sites. In site 6 three of the species exclusively occurred, indicating a strong influence of the Liao River on the fish community.

For 9 species less than 10 individuals were found (*Chanodichthys erythropterus*, *Gymnogobius urotaenia*, *Tachysurus fulvidraco*, *Hypomesus olidus*, *Pungitius sinensis*, *Misgurnus mohoity*, *Silurus asotus*, *Oryzias sinensis*, *Opsariichthys bidens*).

In terms of metrics the German electrofishing method resulted in the highest value for α -diversity with $H' = 2.35$ in contrast to seining (1.65), Table 2.6. The catch from trapping resulted in the highest values for species richness with $d = 3.936$.

2.2.2.2 Method Efficiency

In comparison of overall method efficiency, none of the methods turned out to be suitable to cover the full species range detected at each site compared to all methods combined. The electrofishing methods, however, detected always at least >50% of the species, while seining detected less than 40%. The results using the German electrofishing method (EG) had the greatest overall coverage in terms of number of taxa and biomass, and was suited to detect 56–100% of the species present at the specific sites. This result is further corroborated by the comparison of the catch per unit effort (CPUE). The electrofishing methods in general was the most effective method to characterize fish assemblages. The calculated CPUE generally revealed

Table 2.5 List of fish species in the Fan River system. Total number of individuals (N), percentage of total catch (Proportion) and biomass is given

Species name	Sum N	Proportion [%]	Biomass [kg]
<i>Rhynchocypris lagowskii</i>	2484	31.8	4.56
<i>Barbatula nuda</i>	1683	21.6	3.00
<i>Abbottina rivularis</i>	481	6.17	1.49
<i>Carassius auratus</i>	377	4.8	4.94
<i>Cobitis granoei</i>	374	4.8	0.39
<i>Pseudorasbora parva</i>	340	4.4	0.66
<i>Rhinogobius giurinus</i>	337	4.3	0.52
<i>Misgurnus anguillicaudatus</i>	298	3.8	0.58
<i>Rhinogobius brunneus</i>	218	2.8	0.14
<i>Zacco platypus</i>	195	2.5	0.52
<i>Rhinogobius cliffordpopei</i>	194	2.5	0.22
<i>Rhodeus lighti</i>	187	2.4	0.19
<i>Microphysogobio chinensis</i>	164	2.1	0.27
<i>Gobio cynocephalus</i>	107	1.4	0.31
<i>Acanthorhodeus chankaensis</i>	91	1.2	0.22
<i>Micropercops swinhonis</i>	60	0.8	0.10
<i>Paramisgurnus dabryanus</i>	55	0.7	0.15
<i>Aphyocypris chinensis</i>	29	0.4	0.05
<i>Romanogobio tenuicorpus</i>	23	0.3	0.03
<i>Rhodeus lighti</i>	21	0.3	0.02
<i>Hemiculter leucisculus</i>	20	0.3	0.01
<i>Lefua costata</i>	17	0.2	0.02
<i>Oryzias sinensis</i>	9	0.1	< 0.01
<i>Opsariichthys bidens</i>	9	0.1	0.44
<i>Silurus asotus</i>	8	0.1	0.31

(continued)

Table 2.5 (continued)

Species name	Sum N	Proportion [%]	Biomass [kg]
<i>Misgurnus mohoity</i>	5	<0.1	0.03
<i>Pungitius sinensis</i>	4	<0.1	<0.01
<i>Hypomesus olidus</i>	3	<0.1	<0.01
<i>Gymnogobius urotaenia</i>	2	<0.1	<0.01
<i>Tachysurus fulvidraco</i>	2	<0.1	0.24
<i>Chanodichthys erythropterus</i>	1	<0.1	0.05

Table 2.6 Diversity indices of fish community in Fan River based on the result of the different sampling methods

Method	Total species S	Total individuals N	Species richness D	Evenness J'	Shannon diversity H'	Simpson diversity 1-Lambda
EC	24	1954	3.03	0.67	2.15	0.81
EC 30	27	1714	3.49	0.52	1.74	0.64
EG	28	3570	3.30	0.70	2.35	0.84
SN	11	125	2.07	0.69	1.65	0.69
T	24	345	3.93	0.74	2.37	0.83
L	0	0	-	-	-	-

significant differences between all methods (KW, $H(3) = 33.7$, $p < 0.001$)¹. Within the active methods, the seining resulted in a significant lower CPUE compared to electrofishing methods ($p < 0.05$) and the passive method trapping showed significant lower CPUE compared to each other method ($p < 0.001$). Seining indicated to be the least efficient method of the active methodologies, however still yielded a higher CPUE compared to passive methods. Longline fishing never yielded any catch, regardless the site in which this method was applied. The method efficiency comparison is presented in Table 2.7.

Gill-netting was not applicable in the upstream parts of the river with higher flow-velocities and shallow water depths. In the most downstream, deep and low flowing reaches, results from the gill-net differed in terms of species composition from all the other methods applied, presumably caused by being more selective towards pelagic species such as *Carassius auratus* or *Hypomesus olidus* which were hardly caught with the active fishing methods applied from the river bank as used in this comparison.

¹ Statistical comparisons to test for differences between groups were performed by using the Kruskal–Wallis rank-sum test, with Mann–Whitney U pairwise comparisons (Bonferroni corrected). Significance was accepted at $p < 0.05$.

Table 2.7 Method efficiency comparison. Total number of taxa detected for each method and site. Values in brackets represent percentage of taxa detected in relation to total number of taxa by all methods combined

Site	N species	EC	EC ₃₀	EG	G	L	SN	T
1	9	5 (56%)	7 (78%)	5 (56%)	N/A	0	2 (22%)	2 (22%)
2	19	15 (79%)	16 (84%)	16 (84%)	N/A	0	5 (26%)	10 (53%)
3	15	13 (87%)	11 (73%)	15 (100%)	N/A	0	5 (33%)	11 (73%)
4	16	11 (69%)	11 (69%)	13 (81%)	N/A	0	0 (0%)	10 (63%)
5	19	10 (53%)	12 (63%)	17 (89%)	N/A	0	7 (37%)	9 (47%)
6	21	9 (42%)	12 (57%)	15 (71%)	11 (52%)	0	5 (24%)	6 (29%)

Some of the species that were only detected by method EG comprise bottom-dwelling species or “rare” species with overall specimen <10 at all sites combined.

2.3 Discussion

The results of this study highlight potential benefits of integrating standardized biological monitoring tools of fish community assessments that can be combined with existing Chinese standards for physico-chemical metrics used to rank surface water bodies into water quality grades I to V+. The grading system stems from the perspective of water usage purposes, such as suitability for drinking or irrigation purposes. This is in contrast to the European system that primarily focusses on the alterations of the ecological status of its aquatic systems including the multiple impacts on them. With respect to the recent rapid economic development, impacts on aquatic ecosystems in China have heavily increased. In the year 2014 overall water quality did not meet the criteria for the general public and fishery use (at least Grade III) from 28.8% of the monitoring sections in major rivers [66]. In 2017, the physico-chemical water quality of rivers showed for 23.8% of the monitoring sections a quality grade IV and V, while 8.3% failed to meet the grade V standard [38]. Investigations by Chinese researchers have previously identified physical habitat alterations and chemical pollutants as key factors influencing fish communities in the Liao River Basin [17, 65]. They also reported that, compared with the historical data, almost half of the fish species have disappeared and some local species were replaced by invasive ones. To meet the goal to restore biological integrity in the Liao River system and other watersheds in China, it becomes obvious that monitoring tools should consider biological quality elements such as fish. Since biomonitoring methods are already regionally tested, it can be anticipated that China’s national assessment approach changes in the near future and suitable protocols will be developed soon. European monitoring programs (including Germany) comprise various types of quality elements. Biological water QEs are thereby primarily decisive for assessing a river’s

actual status. Physico-chemical and hydromorphological QEs support the biological QEs. Hereby, a large spectrum of the complexity of different biological and ecological characteristics as well as the large variety of pressures and impacts can be covered. One strength of this approach is the general comparability over space and time, which allows to detect deviations between the observed conditions and the respective reference conditions. Specific metrics, as applied in the European WFD cannot be used for a direct comparison here. For Chinese Rivers there is so far limited information on guilds distribution, migration (index-based), fish region (index-based), and dominant species (index-based) as used in Europe.

For a representative assessment not only the choice of methods is crucial, but also the consideration of site-specific characteristics [4, 5, 41]. Depending on present fish communities either passive or active bioindication methods [45] might be suitable, ideally comprising a combination of different methods. In the method comparison presented here, the differences between the applied sampling strategies could exemplarily be shown. However, no method was suitable to detect all the species present at a given site. Only a combination of active and passive sampling techniques provides a best-case coverage. Electrofishing-based methods showed to be most effective, regarding range, efficiency and the potential to detect rare species. In this direct comparison the German electrofishing method revealed to be more suitable in comparison to the other applied methods in terms of efficiency and CPUE. In general it can be stated that active methods such as electrofishing should preferentially be used for fish community assessment in running waters [10]. There were minor differences between the Chinese and the German electrofishing, however, most likely this is rather a function of the established voltage by the different power sources used. Ideally the power of the electrofishing gear is adequately chosen, based on the river-specific size, discharge, depth and conductivity [41]. This approach is already applied in European fish stock assessments like the European Water Framework directive monitoring programs [10]. For headwater areas and small wadeable rivers, the Chinese electrofishing gear proved to be sufficient to characterize the present fish community. However, when species richness (or age structure) is the target of the fish assessment, this method increases the risk of missing species (or size classes). Thereby the time-based approach EC30 provided a broader spectrum compared to the single-pass fishing approach. In contrast, the estimation of the ecological status does not require the determination of absolute fish densities, but relative species composition and abundance [58]. During electric fish sampling, species composition and relative abundance are likely to be biased because of differences in the catchability of different species and fish sizes [3, 11, 56], which demands sufficient efficiency of the methods. Based on the difficulty of detecting rare species, random sampling should be conducted in conjunction with targeted sampling of rare species or passive methods such as trapping (see also [55]). Methods such as the Longline fishing showed to be not suitable for fish community characterization in this study.

No historic data is available for the Fan River, however, the increasing anthropogenic pressure and resulting aquatic species decline is indicated by many experts. This highlights the overall value of biological monitoring for aquatic health status

assessment. More detailed information on biological integrity is urgently needed, especially for monitoring of the future status of these river systems.

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