ASSESSING IMPACTS OF A DAM CONSTRUCTION ON BENTHIC MACROINVERTEBRATE COMMUNITIES IN A MOUNTAIN STREAM

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ABSTRACT

Dam would influence the flux of water and sediment, eventually disturbing ecological connectivity. In Asia, especially in China, there are many dams, but great scientific uncertainty still exists about their effects on downstream macroinvertebrates. Therefore, we investigated macroinvertebrate assemblages, in upstream and downstream reaches of a run-of-river dam on a tributary of the Xiangxi River, central China. Surber samples and associated physicochemical measurements were taken from five sites during the period from June 2003 to June 2007. According to the metrics selection principle of benthic biotic integrity index (B-IBI), we selected macroinvertebrate species richness and number of EPT taxa from 27 biological metrics to assess the effect of dam construction. We divided sites into three habitats (H1-H3) according to the impact extent of the dam. Using partial least squares (PLS), we developed two predictive models (macrionvertebrate species richness and number of EPT taxa) based on the environmental variables of the unregulated habitat (H1). These models were then used to predict species richness and number of EPT taxa at impacted habitats (H2 and H3). To evaluate the impact of flow regulation residuals, i.e. the differences between observed and predicted values, were used to evaluate the impact of flow regulation. Significant impacts of flow regulation on macroinvertebrate species richness and number of EPT taxa were detected at impacted habitats (H2 and H3), where observed species richness were lower 8.7% and 28.1%, and EPT taxa also were lower 12.8% and 30.0%. Then, possible mechanisms for observed impacts were discussed. This study addressed the negative impacts of run-of-river dams on macroinvertebrates, which may lead to more sustainable management strategies.

KEYWORDS: flow regulation, benthic macroinvertebrates, partial least squares models, Xiangxi River

1 INTRODUCTION

Dams are widespread in the world because of their many benefits and services such as flood abatement, irrigation, recreation and hydropower, but their environmental costs are significant too [1]. Dam fragment river systems, modify the flow regime and the transport of sediment, and change the physical and biological structure of rivers both upstream and downstream [2-4], eventually disturbing river ecological connectivity [5]. Numerous studies have focused on the impact of large dams (>15 m high) and reservoirs on water chemistry, fish, zoobenthos and phytoplankton [6-9], but few have investigated the effects of small run-of-river dams [10].

The use of biological indexes to assess the impact of dam construction is more widely recognized than the use of water chemistry. Water quality only partially reflects environmental impact [11], while biological indicators reflect the intensity of anthropogenic stress and have been used as a tool in risk assessment and evaluation of human induced changes in freshwater ecosystems [12]. Biological indicators have widespread appeal to scientists, environmental managers, and the general public, and were mainly used to assess the condition of the environment [13].

Benthic macroinvertebrates in streams are relatively easy to collect and identify. Most of them are sensitive, with short life histories, can respond rapidly to environmental change, and have a diversity of functional and effect traits that provide a variety of responses to changing conditions [14]. Macroinvertebrates which are popular in stream waters and ecosystem health assessments, are also important indices for monitoring and assessing river environments [15]. Past studies have reported that benthic macroinvertebrates below dams show changes in their assemblage composition, but many of these studies are based on observations of particular environmental factors or a particular taxon [16]. Our research was aimed at finding the suitable metrics to assess the effects of dam construction.
2 MATERIALS AND METHODS

2.1. Study area

Field data was collected from Jiuchong River, which is the best habitat quality river of the three main tributaries (Jiuchong River, Gufu River and Gaolan River) of the Xiangxi River. Xiangxi is a 6th order river that originates from Shennongjia Mountain, where it discharges into the Yangtze River (Fig. 1A, B). It has a watershed area of 3099 km² and a natural gradient of 1,540 m from the headwaters to its confluence with the Yangtze River, being the biggest tributary of the Three Gorges Reservoir (TGR) in Hubei province [17]. The average annual precipitation within this watershed is 900–1200 mm [17].

We selected a small dam (1.5m high, 20 m across) that was constructed in August 2003 on the Jiuchong River (Fig. 1B) to study the impact of the dam construction on the benthic macroinvertebrate communities. We sampled monthly at 5 sites (JC1-JC5; Fig. 1C), from June 2003 to June 2007. JC1 and JC2 were located approximately 1 km upstream of the dam where river channels were in a natural state, free of its impacts. JC3 was just upstream from the dam and slightly influenced by its intake. JC4 and JC5 were upstream from the outlet of a small hydropower station (Fig.1C). According to the impact extent of the dam, we divided the sites into three habitats (H1-H3). H1 comprised of unregulated sites including the sites before dam construction and JC1 and JC2 after dam construction. JC3 after dam construction made up H2, JC4 and JC5 after dam construction made up H3.

2.2. Benthic macroinvertebrate and water sampling

We established 1-3 random sampling locations (30×30 cm² quadrates) in each site for benthic macroinvertebrate measurements. All stones within the Surber sampler frame (mesh = 420 µm) were scrubbed with a soft brush to remove attached organisms. In areas of unconsolidated substrata, the river bed was sampled to a depth of about 10 cm. Macroinvertebrates were separated from sand and mud, and preserved in 10% formalin. The biological samples were transported to the lab and identified according to taxonomic references [18, 19].

In each site where benthic macroinvertebrates were captured, we used a hydrolab Minisonde (Hach Environmental, Loveland, Colorado) to measure in situ variables that included pH, conductivity (Cond) and water temperature (WT). We also measured water depth, water width and current velocity (LJD-10 water current meter).

FIGURE 1 - Jiuchong River within the Xiangxi River watershed in the People’s Republic (P. R.) of China and locations of sampling sites. A.-Yangtze River. B.-Xiangxi River watershed. C.-Jiuchong (JC) River
We collected surface water samples in plastic containers to measure chemical variables according to the standard methods [20] in the lab: including total nitrogen (TN), ammonium (NH₄-N), nitrate (NO₃-N), total phosphorus (TP), orthophosphate (PO₄-P), hardness, calcium (Ca²⁺), chloride (Cl⁻), alkalinity (Alk), silicon (SiO₂) and chemical oxygen demand (COD).

2.3 Data analyses

Based on Kerans and Karr [21] and Barbour et al. [22], we compiled 27 macroinvertebrate metrics that were classified into six categories; richness measures, composition measures, functional feeding group measures, habitat measures and diversity indices (Table 1).

<table>
<thead>
<tr>
<th>Category</th>
<th>Metric</th>
</tr>
</thead>
<tbody>
<tr>
<td>Richness measures</td>
<td>No. total taxa (Species Richness)</td>
</tr>
<tr>
<td></td>
<td>No. EPT taxa</td>
</tr>
<tr>
<td></td>
<td>No. Ephemeroptera taxa</td>
</tr>
<tr>
<td></td>
<td>No. Plecoptera taxa</td>
</tr>
<tr>
<td></td>
<td>No. Trichoptera taxa</td>
</tr>
<tr>
<td></td>
<td>No. Diptera taxa</td>
</tr>
<tr>
<td></td>
<td>No. Chironomidae</td>
</tr>
<tr>
<td></td>
<td>No. (Crustacea + Mollusca) taxa</td>
</tr>
<tr>
<td>Composition measures</td>
<td>% EPT</td>
</tr>
<tr>
<td></td>
<td>% Ephemeroptera</td>
</tr>
<tr>
<td></td>
<td>% Plecoptera</td>
</tr>
<tr>
<td></td>
<td>% Trichoptera</td>
</tr>
<tr>
<td></td>
<td>% Diptera</td>
</tr>
<tr>
<td></td>
<td>% Chironomidae</td>
</tr>
<tr>
<td></td>
<td>% Oligochaeta</td>
</tr>
<tr>
<td></td>
<td>% (Crustacea + Mollusca)</td>
</tr>
<tr>
<td></td>
<td>% Dominant taxon</td>
</tr>
<tr>
<td></td>
<td>% Three most dominant taxa</td>
</tr>
<tr>
<td>Feeding measures</td>
<td>% Scapers</td>
</tr>
<tr>
<td></td>
<td>% Shredders</td>
</tr>
<tr>
<td></td>
<td>% Gatherers</td>
</tr>
<tr>
<td></td>
<td>% Filterers</td>
</tr>
<tr>
<td></td>
<td>% Predators</td>
</tr>
<tr>
<td>Habitat measures</td>
<td>No. clinger taxa</td>
</tr>
<tr>
<td></td>
<td>% Clingers</td>
</tr>
<tr>
<td>Biodiversity indexes</td>
<td>Shannon-Wiener index</td>
</tr>
<tr>
<td></td>
<td>Evenness index</td>
</tr>
</tbody>
</table>

We used four criteria to select biological metrics from the candidate ones. First, we eliminated metrics with medi-ans of zero before data analyses because low values could prevent identification of differences among groups [22, 23]. Second, we transformed (natural log) all metrics consisting of proportional data to meet the assumptions of normality. We selected metrics that differed significantly between references (H1) and impaired (H3) sites. We used two-tailed t-tests assuming equal or unequal variances, depending on t-test results, to determine whether differences existed (α=0.05) [24]. For metrics that met our first and second criteria, we evaluated the separation power of potential metrics using box plots. We defined separation power as the degree of overlap between boxes (i.e., 25th and 75th quartiles) in box plots of the values of the metric for both reference (H1) and impaired (H3) sites [22]. We assigned a separation power of three when boxes did not overlap between the H1 and H3 site groups, a value of two when interquartile ranges overlapped but did not reach medians, a value of one when only one median was within the interquartile range of the other box, and a value of zero when both medians were within the range of the opposite box. We then excluded metrics with Coefficient of variation (CV) > 1 due to their high deviations [23].

The relationship between environmental variables and the metrics that met our four criteria were modeled with partial least squares (PLS) using the software SIMCA-P 11.5 for Windows. PLS modeling combines ordination and regression [25], has some considerable advantages over multiple linear regression [26], and is an important modeling tool in many scientific fields [27-28].

We first built two separate PLS models to describe how selected metrics at unregulated sites (H1) were related to environmental variables. Subsequently, we modified the models by excluding variables directly affected by flow regulation. From these new models, we predicted species richness and number of EPT taxa at sites with reduced and unreduced, but regulated flow. We used the residuals, i.e. the difference between values observed and those predicted from the models, to estimate the impacts of flow regulation on macroinvertebrates species richness and number of EPT taxa. Effects were calculated using the formula [(observed value–predicted value)/predicted value]×100. The effect of flow regulation was regarded as significant, if 95% confidence intervals for the means of effects did not include zero, corresponding to $p <0.05$ [25, 26]. This procedure produces a measure of the validity of the model, called $Q^2$. A component is judged significant if $Q^2$ is larger than a critical value (in our study, $Q^2_{limit} = 0.05$, corresponding to $p <0.05$).

3 RESULTS

3.1. Metrics selection

Of the 27 candidate metrics, no metrics resulted in medians of zero. Eight metrics (Evenness index, % Ephemeroptera, % Plecoptera, % Chironomidae, % Filterers, % Gatherers, % Clingers, % Dominant taxon and % Three most dominant taxa) were rejected because Spearman tests indicated no significant differences ($p > 0.05$) between reference and impaired sites. We then compared the remaining metrics based on separation power. We selected species richness, number of EPT taxa and % Plecoptera because of their high separation power (Fig. 2), but we removed % Plecoptera because of high CV (CV = 1.31). Finally, species richness and number of EPT taxa were selected to assess the effect of dam construction.

3.2. Species richness and number of EPT taxa

The average macroinvertebrate richness for all sites was 25, and mean number of EPT at all sites was 13.6 (Table 2). Macroinvertebrate richness differed significantly among the three groups (ANOVA, $p < 0.05$. Mul-
multiple comparisons showed that H1 and H3 sites, H2 and H3 sites differed significantly (ANOVA, \( p < 0.05 \)), H1 and H2 sites were not significantly different (ANOVA, \( p > 0.05 \)). The number of EPT taxa differed significantly (ANOVA, \( p < 0.05 \)). Multiple comparisons showed that H1 and H3 sites, H2 and H3 sites differed significantly (ANOVA, \( p < 0.05 \)), H1 and H2 sites were not significantly different (ANOVA, \( p > 0.05 \)).

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![FIGURE 2 - Box plots of component benthic macroinvertebrate metrics for reference and impaired sites. Boxes show interquartile ranges (25th and 75th percentiles), middle lines are medians, middle squares are averages, whiskers show interquartile ranges (5th and 95th percentiles), and circles are outliers.](image)

![TABLE 2 - Mean and range of macroinvertebrate species richness and number of EPT taxa at all sites and each habitat group](image)

<table>
<thead>
<tr>
<th>sites</th>
<th>n</th>
<th>Species richness (range)</th>
<th>Number of EPT taxa (range)</th>
</tr>
</thead>
<tbody>
<tr>
<td>All sites</td>
<td>235</td>
<td>25.0 (0-50)</td>
<td>13.6 (0-27)</td>
</tr>
<tr>
<td>H1</td>
<td>97</td>
<td>28.9 (9-49)</td>
<td>15.4 (3-24)</td>
</tr>
<tr>
<td>H2</td>
<td>46</td>
<td>27.0 (12-47)</td>
<td>14.0 (6-27)</td>
</tr>
<tr>
<td>H3</td>
<td>92</td>
<td>22.0 (0-50)</td>
<td>11.4 (0-27)</td>
</tr>
</tbody>
</table>

### 3.3. Effects of flow regulation on macroinvertebrate metrics

Significant models were developed for species richness and number of EPT taxa. An analysis of species richness at unregulated sites (H1), with respect to 17 environmental variables, yielded one significant component (\( Q_{y}^2 = 0.165 \)). A similar analysis of number of EPT taxa produced one significant component (\( Q_{y}^2 = 0.151 \)) (Table 3). Two models indicated that Ca\(^{2+} \), WT, hardness and velocity were important for both species richness and number of EPT taxa (Fig. 3).

| TABLE 3 - Fractions of variance explained and fractions of variation of dependent variables predicted by PLS models for species richness and number of EPT taxa at unregulated sites (H1). \( R_{x}^2 \) is the fraction of the variance of all the independent variables and \( R_{y}^2 \) of all the dependent variables explained by the model. \( Q_{y}^2 \) is the fraction of the total variation of the dependent variable that can be predicted by the model |
|-----------------|-----------------|-----------------|-----------------|-----------------|-----------------|-----------------|
|                  | With 17 environmental variables | Excluding flow velocity |
|                  | \( R_{x}^2 \) | \( R_{y}^2 \) | \( Q_{y}^2 \) | \( R_{x}^2 \) | \( R_{y}^2 \) | \( Q_{y}^2 \) |
| Species richness | 0.104 | 0.393 | 0.165 | 0.101 | 0.370 | 0.126 |
| Number of EPT taxa | 0.115 | 0.334 | 0.151 | 0.113 | 0.334 | 0.148 |

![FIGURE 3 - Loadings for the most influential variables in PLS regression models of (a) macroinvertebrate species richness and (b) number of EPT taxa.](image)

To estimate the effects of flow regulation, we built new PLS models for species richness and number of EPT taxa based on H1 sites, excluding flow velocity, which was assumed to be directly influenced by flow regulation. One significant component was extracted for species richness (\( Q_{y}^2 = 0.126 \)) and one for number of EPT taxa (\( Q_{y}^2 = 0.148 \)) (Table 3). These models were then used to predict species richness and number of EPT taxa at the two other habitats (H2-H3). At H2 and H3, respectively, observed species richness of 8.7% and 28.1% and number of EPT taxa of 12.8% and 30.0% were significantly lower than predicted values (Fig. 4). The effects observed at H2 and H3 differ from zero (Fig. 5), indicating significant effect of flow regulation on species richness and number of EPT taxa.

### 4 DISCUSSION

#### 4.1. Metrics selection

Species richness and number of EPT taxa are the metrics for developing a benthic index of biotic integrity (B-IBI). The B-IBI is based on a series of structural and functional metrics of benthic macroinvertebrate assemblages, and thus helps quantify the impact of environmental deterioration [21, 29]. This biological indicator can assess the
condition of a site, evaluate the human induced changes in the freshwater ecosystem [30], and has been commonly used in biological assessment [31]. A potential disadvantage of using a multimetric B-IBI is that it can mask responses of individual metrics because of averaging [32]. An individual criterion could be based on a metric indicating a valued ecological attribute of sufficient public or ecological concern that merits specific management attention [33]. In our study, species richness and number of EPT taxa could be used to set the minimum criteria for the impact of dam construction standards.

4.2. Effects of flow disturbance

Water flow determines the stream characteristics [34], greatly affecting the benthic organisms [34, 35]. McIntosh et al. (2008) [36] suggested that diverted stream flow limits macroinvertebrate colonization and growth, expressed as reduced community density and biomass. Macroinvertebrate assemblage below dams often show a reduction of diversity with low taxon richness and predominance of particular species [16, 37]. In the Xiangxi River, biomass and density of benthic macroinvertebrates at impacted sites were lower than those at unregulated sites [38]. Furthermore Thomson et al. (2005) [3] indicated that macroinvertebrate species richness and number of EPT taxa in downstream reaches raised significantly following complete removal of a small run-of-river dam in Manatawny, Pennsylvania. Our study showed a similar result where observed species richness and number of EPT taxa at H2 and H3 were significantly lower than predicted values. Number of EPT Taxa have been commonly used in water quality assessments representing all species, i.e. RIVPACS, AUSTRIVAS [39]. Bona et al. (2008) [37] suggested that,
to assess water quality by EPT, was more suitable in high altitude streams.

The study of Velinsky et al. (2006) [40] in Pennsylvania suggested that the impacts of the dam on H2 sites were probably related to dam size and impoundment residence time. Our dam is a run-of-river structure with a relatively small impoundment area, and water flows directly into the intake weirs with a very low residence time. Wu et al. (2010) [41] suggested that benthic diatoms were slightly influenced by the dam at the just upstream sites from it in Xiangxi River. While in our study, observed species richness and number of EPT taxa of H2 were significantly lower than predicted values. The reason may be that macroinvertebrates have a drift behavior, enabling them to escape unfavorable conditions either actively or passively [42]. Because the H2 site is near the intake to the hydropower station, the drifting macroinvertebrates were easily sucked into the aqueduct.

4.3. Possible mechanisms for observed impacts

PLS models have been used to estimate the magnitude of effects of dams on benthic macroinvertebrates and diatoms [25-26, 41]. We tried to elucidate the mechanism causing the observed impacts, a necessary step since dams affect the environment in many different ways. For example, effects on Hydropsychids have been attributed to changes in temperature, food availability, substratum stability and flow variability [43-45]. Impacts on benthic macroinvertebrate communities have been ascribed to changes in flow variability [38]. Only when mechanisms have been identified, cost-effective remedial measures can be suggested [25].

One possible mechanism was that lower in discharge usually causes lower water velocity, water depth and wetted channel width; increased water temperature and sedimentation; and changes in thermal regime and water chemistry [46]. Eventually, decreases in flow cause immediately increased in macroinvertebrate drift [46] and decreased macroinvertebrate richness [47]. In our study, significant impacts of flow regulation on macroinvertebrate species richness and number of EPT taxa were detected at impacted habitats. Therefore, we hypothesize that habitat changes created by flow reduction play an important role in the decrease of macroinvertebrate species richness and number of EPT taxa.

Another potential reason is biotic interactions. In the Xiangxi River, biomass and density of diatoms at impacted sites were higher than those at unregulated sites [41]. Some sessile macroinvertebrate taxa have large biomass such as Simulium sp. and Antocha sp. below the dam [16]. Zhang et al. (1998) [26] found that observed richness of Simulium sp. at impacted sites was higher than predicted values. We did not find observed richness of Simulium sp. was higher than predicted values in our study, because Simulium sp. was rare specie in this study area. It suggested that increased benthic diatom density and biomass changed the food resources to ease the competition for food between diatom-eating macroinvertebrate species.

In conclusion, a combination of biotic interactions and habitat changes may explain why macroinvertebrate species richness was affected by flow regulation. Although other possible mechanisms for observed effects may exist, we hope this study can help estimate the effects of dam construction better, and lead to more sustainable management strategies of hydropower.

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