Article:

Fu, H, Xu, J, Zhang, H et al. (4 more authors) (2023) A meta-analysis of environmental responses to freshwater ecosystem restoration in China (1987–2018). Environmental Pollution, 316 (Part 2). 120589. ISSN 0269-7491

https://doi.org/10.1016/j.envpol.2022.120589

A meta-analysis of environmental responses to freshwater ecosystem restoration in China (1987-2018)

Hong Fu [1,2,3], Jun Xu [1*], Huan Zhang [1], Jorge García Molinos [4], Min Zhang [5], Megan Klaar [2], Lee E. Brown [2]

[1] Donghu Experimental Station of Lake Ecosystems, State Key Laboratory of Freshwater Ecology and Biotechnology of China, Institute of Hydrobiology, Chinese Academy of Sciences, Wuhan, 430072, P. R. China

[2] School of Geography and water@leeds, University of Leeds, Leeds, West Yorkshire, United Kingdom

[3] University of Chinese Academy of Sciences, Beijing 100049, P. R. China

[4] Arctic Research Centre, Hokkaido University, Sapporo, Japan

[5] College of Fisheries, Huazhong Agricultural University, Hubei Provincial Engineering Laboratory for Pond Aquaculture, Freshwater Aquaculture Collaborative Innovation Centre of Hubei Province, Wuhan, P. R. China

Correspondence authors: Jun Xu (xujun@ihb.ac.cn, Tel & Fax + 86 27 68781095)

Abstract

Understanding how abiotic and biotic components respond to aquatic ecosystem restoration is pivotal for sustainable development in the face of economic development and global environmental change. However, the post-restoration monitoring and evaluation of aquatic ecosystems across large spatial and temporal scales is underfunded or not well documented, especially outside of Europe and North America. We present a meta-analysis of abiotic and biotic indices to quantify post-restoration (2 month to 13 years) effects from reported aquatic restoration projects throughout the China-mainland, incorporating 39 lentic and 36 lotic ecosystems. Decreases in dissolved nutrients (total nitrogen, ammonia nitrogen and total phosphorus) post-restoration were rapid, but tended to slow after about 9.3 years. Response ratios summarizing biodiversity responses (incorporating phytoplankton, invertebrates, vascular plants, fish and birds) typically lagged behind abiotic changes, suggesting longer timescales are needed for biotic indices to recover. Time since restoration interacted with lentic project size, showing even with the same proportional efforts of restoration, larger lentic ecosystems responded much more slowly than smaller ones. Spatial heterogeneity, reflecting the effects of different restoration approaches (e.g., sewage interception, polluted sediment dredging, artificial wetlands, etc.), had a significantly stronger effect on biotic than abiotic indices, particularly in rivers compared to standing waters. This reflects the complexity of fluvial ecosystem dynamics, and hints at a limitation in the reinstatement of ecological processes in these systems to overcome issues such as dispersal limitations. Overall, the different timelines and processes by which abiotic and biotic indices recover after restoration should be taken into account when defining restoration targets and monitoring programs. Our study illustrates the value of long-term aquatic ecosystem monitoring, especially in China given the scale and magnitude of ongoing restoration investments in the country.

Keywords: lake; river; biodiversity; recovery timeline; reintroduction; water pollution

1 Introduction

An estimated 2.4 % of the Earth's land surface consists of freshwater ecosystems (Van Klink et al., 2020). These ecosystems host unique biodiversity and maintain important ecosystem services such as water and food supply, climate regulation and recreation (Janse et al., 2015), but are particularly vulnerable to degradation because rivers and lakes integrate the effects of all activities occurring within their catchments (Kummu et al., 2011). Due to ever-increasing global anthropogenic pressures, the restoration and conservation of freshwater ecosystems is now among the most pressing environmental concerns (Carvalho et al., 2019). Previous global studies have demonstrated improvements in biodiversity and ecosystem services following restoration of river, lake and estuarine ecosystems (Benayas et al., 2009; Jeppesen et al., 2005; Kail et al., 2015; Lu et al., 2019). For example, Jeppesen et al. (2005) reported the re-oligotrophication process followed by 35 North American and European lakes resulting from reductions of external nutrient (nitrogen and phosphorus) loading. In-lake total phosphorus (TP) concentrations reached equilibrium in most lakes about 10-15 years post-restoration due to the effect of internal loading, whereas decreases in total nitrogen (TN) loading had a much more immediate effect on in-lake TN concentration. Biological parameters also responded to the reduced loading, including reduced phytoplankton biomass and chlorophyll-*a* levels, shifts in community structure and enhanced zooplankton biomass. However, the changes in the recovery trajectories of abiotic and biotic indices caused by various restoration measures are still understudied in the literature and remain unclear due to a general lack of long-term monitoring data to understand restoration effects over time (Kail et al., 2015; Lu et al., 2019), especially within large geographical settings and in lotic ecosystems. A synthesis of river restoration projects across the USA, Bernhardt et al. (2005) highlighted the lack of long-term monitoring as a major impediment to evaluating restoration success. Furthermore, understanding differences between abiotic and biotic responses to restorations in different aquatic ecosystems (lentic/lotic) both in space and time has been suggested as a further research priority (Verdonschot et al., 2013). To address these research needs, we conducted a meta-analysis of aquatic ecosystem restoration projects across China to quantitatively assess the long-term temporal variation of a suite of abiotic and biotic indices frequently used as key indicators of the success of freshwater ecosystem restoration (Fu et al., 2021).

As the world's largest developing country, urbanization in China has proceeded rapidly. Since the onset of the national reform and opening-up policy in 1978, its annual urbanization expansion rate has increased from < 20% to > 57% in 2016 (Liang and Yang, 2019). However, urbanization and economic development has brought an acute problem of natural ecosystem degradation, especially water pollution (Liu et al., 2016). To ameliorate the negative impacts of accelerated aquatic environmental degradation, investments in ecosystem restoration for improving China's natural water quality increased dramatically from being negligible in 1994 to 1,000 billion RMB in 2014 (Zhou et al., 2017). Based on national records of dissolved oxygen (DO), chemical oxygen demand (COD), and ammonium (NH_4^+), Zhou et al. (2017) concluded that China's increasing gross domestic product (GDP) during the 2006–2015 period was not at the expense of its inland waters due to concurrent restoration efforts. However, the wider extent of improvement remains unknown as the study did not consider any changes in biological status, whilst other studies incorporating biotic indices (e.g. plants, fish, invertebrates, etc.) have focused either on single lakes (e.g. (Bai et al., 2020)) or specific regions of China (e.g. Taihu basin (Fu et al., 2021)). Further studies of national-scale responses to restoration are vital to provide guidance for government investment allocation, in particular by revealing if there are any regional variations in coupled response patterns of abiotic and biotic indices.

China is a geographically vast country with a wide diversity of aquatic ecosystems and environments. It spans 50° latitude and covers five climatic zones (Wang et al., 2016). Therefore, lotic and lentic elements of a watershed may strongly vary in the outcomes of restoration projects, depending on their specific hydrologic and biological conditions (Levi and McIntyre, 2020).

Here, we present a comprehensive national-level (China mainland) meta-analysis of the temporal trajectory of different abiotic (biological oxygen demand, nutrients like nitrogen and phosphorus,) and biotic (species richness/diversity and abundance/biomass) indices, used as indicators of restoration effects. The assembled datasets extend up to a maximum of 13 years after the restoration (individual studies were implemented between 1987 and 2018), and span 75 lentic and lotic freshwater ecosystems (Table S1). The study aimed to test the following hypotheses: (H_1) biotic indices would lag behind abiotic indices after restoration, but eventually become similar if restoration schemes are maintained over enough time. This reflects the likelihood that species require additional time to recolonize newly generated habitats (Watts et al., 2020) and subsequently establish populations following restoration. (H_2) project size and different types (lotic vs. lentic) of aquatic ecosystems would influence restoration effects, with larger ecosystems supporting more biodiversity but taking longer to recover (Fukami, 2004). (H_3) the response of abiotic indices to restoration was expected to be more predictable (i.e. with significantly smaller variability) in comparison to biotic indices, because of the complexity of organism life-history strategies (lifespan, fecundity, etc.) and different restoration schemes at large spatial scale. Finally, (H_4)

temperature can considerably shape aquatic ecological environments and biodiversity (Yang et al., 2018), therefore we expected that the rate of aquatic ecosystem recovery after restoration would vary in different climatic zones.

2 Methods

2.1 Literature search

We conducted a systematic literature search using the CNKI (China National Knowledge Infrastructure) search engine for studies published up to 19th December 2019 and matching the following search term combinations: (restor* OR rehabil* OR recover* OR reestab* OR repair*) * ecological AND (freshwater OR river OR lake OR stream OR wetland OR channel OR waterway OR watershed OR basin). This search, which yielded a total of 1705 publications, was conducted primarily in Chinese search engines because data from local restoration projects usually prioritize publication in Chinese journals following project funder requirements. Although projects publishing data in international journals are typically also available from technical reports or other forms of grey literature (PhD dissertations) in Chinese through CNKI, we also conducted a search of literature in the ISI Web of Science using the equivalent search terms. The suitability criteria for inclusion were: (1) the publication provided quantitative data on abiotic and/or biotic indices before the restoration and over a period of at least one month after completion of the restoration; (2) the publication stated the start and end date of restoration; (3) the publication concerned restoration of freshwater systems.

After applying these criteria, 78 studies (which 74 of them were from CNKI, 4 from Web of Science, Table S2) were retained, corresponding to 36 lotic and 39 lentic freshwater systems (Table S1). These provided information on 157 monitored sites within these monitored systems (Fig. 1), comprising a total of 1653 records of abiotic or biotic indices. The geographical distribution of documented projects, mostly concentrated on the eastern half of the country, reflects well the Chinese demographic pattern of a densely populated east and sparsely populated west (Chen et al. 2016). The timescales of the monitored restoration project ranged from 2 months to 13 years (3.69 ± 3.01 years) after restoration (two of them were less than half a year in duration) (Table S3).

2.2 Data extraction

For each publication meeting the search criteria, we documented the location of the restoration projects (latitude and longitude) (Fig. 1), start and completion date, and project size (i.e., the area for lentic ecosystems, and the ratio of restored stream length to bankfull width for lotic ecosystems (Miller et al., 2010). We attempted to categorize studies by the specific restoration measures but almost all were synthetic ecological restoration projects combined with schemes such as sewage interception, polluted sediment dredging, artificial wetlands, submerged macrophyte reintroduction, exclusion of fishing and/or riparian buffer zone restoration. The diversity of schemes incorporated into the analysis allows generalizations to be made about restoration effects, but for the feature of individual restoration measures (e.g., investments, amounts, etc.), the number of published studies typically remains too low to develop more focused meta-analysis. We extracted information on all variables relating to aquatic ecosystem restoration effects, whether or not these were explicitly the focus of restoration actions, before and after restoration. For abiotic parameters these included concentrations of ammonia nitrogen (NH_4^+-N), TN, TP and biological oxygen demand (BOD_5) in water. Biotic indices considered, which including abundance/biomass and richness/diversity of organisms, related to various taxonomic groups including vascular plants, phytoplankton, invertebrates, birds and fish (Table S1). These were incorporated into a combined meta-summary of organism responses following the approach of Benayas et al. (2009).

Studies were classified into two aquatic ecosystem types: lentic/standing (i.e. lakes, reservoir, wetlands, still channels) and lotic/fluvial (i.e. rivers and flowing channels) ecosystems. The final database contained 39 lentic and 36 lotic ecosystems documented in the publications retrieved by our literature search. Several studies reported data from the same ecosystems but with different time periods, and these were combined to avoid pseudo replication. Additionally, we deconstructed some studies which reported more than one ecosystem. Several restoration schemes were reported in more than one publication and these were combined. Where numerical data were not provided in a publication, data were extracted from the figures (> 60% publications) using the Graph digitizer software (Digitizelt, version 2.5, https://www.digitizeit.de). This software has been used widely in meta-analysis studies (Rasheduzzaman et al., 2020; Zhang et al., 2017), and proven to be reliable in extracting data from figures with high level of confidence (see Rakap et al. (2016)).

We focused on the annual accumulated mean daily temperature above 10 °C (AAT10) as an indicator of climatic zones, because it is a key criterion used to divide traditional physical geographical regions in China (Dong et al. 2009). AAT10 of each site was extracted using ArcGIS 10.2 (ESRI Company, Redlands, CA, USA) based on original data downloaded from the Resource and Environment Science and Data Center (https://www.resdc.cn/) at a grid resolution of 500 m. Our analyses then integrated annual averages for the time period since 1980 to 2020.

2.3 Quantifying restoration effects

A response for each comparison between degraded and restored sites was calculated within the same assessment, using the ratio Δr proposed by Benayas et al. (2009) and Miller et al. (2010) as a standardized measure of restoration effects (Eq. (1)).

$$\Delta r = (+/-)\ln(After\ \ Restoration/Degraded) \tag{1}$$

where After Restoration/Degraded means the value of a specific biotic or abiotic metric at the monitored site after and before restoration, or at a local reference degraded site that did not undergo restoration.

Measures of biotic indices include data reported as abundance/biomass, richness/diversity indices (e.g. alpha or beta diversity, evenness, etc.) depending on the study (Benayas et al., 2009) (Table S4). Therefore, the use of the response ratio enables integration of such heterogeneous data and is dimensionless, with positive values indicating an improvement of the original status, and negative values a degradation. Whilst an increase in biodiversity metrics is typically considered as a positive response to restoration, and a decrease in metrics indicates negative effects, this might not always be the case. For example, a decrease in overall richness or abundance may be seen if the loss of pollutant tolerant organisms outweighs their replacement by those found under restored conditions. As such, given that decreasing nutrients (NH_4^+-N, TN, TP), BOD_5 and density of phytoplankton/Oligochaeta in eutrophic environments are the targets of restoration, we reversed the sign of the resulting ratio (-Δr) for these parameters to make their interpretation more intuitive and keep consistency with that of other biological indices for which restoration targets an increase in value (+Δr).

2.4 Statistical analyses

Visual inspection of frequency histograms showed all response ratios of abiotic (ΔrNH_4^+-N, ΔrTN, ΔrP, ΔrBOD) and biotic indices (Δr biodiversity of birds, fish, invertebrates, phytoplankton and vascular plants) followed non-normal distributions. Therefore, we used Wilcoxon signed rank tests to examine whether the median response ratios of ecosystem indices were significantly different from zero. The density plots of the response ratio of abundance/biomass and richness/diversity of each organism were displayed, because we observed bi-modal distributions of almost all the organisms in our study except birds and fish.

The relationships between restoration effects (response ratio) and potential predictors were assessed by fitting a Linear Mixed Model (LMM, model 1) to the response ratios of abiotic and biotic indices, using "lme4" and "lmerTest" R packages (Bates et al., 2014; Kuznetsova et al., 2015). Predictors included categories of abiotic and biotic indices (including NH_4^+-N, TN, TP, BOD_5, birds, fish, invertebrates, phytoplankton and vascular plants), start date, monitored years (*t*) after restoration (d*t*), ecosystem type (lentic vs. lotic), and AAT10. We specified $(dt)^2$, the general category of environment indicators (abiotic vs. biotic), ecosystems type and AAT10 as fixed effects, while sub-categories of abiotic and biotic indices, ecosystem ID and start date of the restoration were include as random effects, plus $dt|$sites as a random slope effect to account for data collected from sites where different restoration schemes were implemented. The quadratic d*t* term accounted for non-linear variation of the abiotic and biotic index responses after restoration over time. To explore whether abiotic and biotic indices showed different variations along the years after restoration, an interactive term ($(dt)^2$ * general category of environment indicators) was specified in the model. The above model showed a significant effect of ecosystem type, therefore two additional LMM models were applied to the response ratio of abiotic and biotic indices for lentic (model 2) and lotic ecosystems (model 3), separately. Here, project size ($\log_{10}$ surface area in km^2 for lentic; the ratio of length to bankfull width for lotic) was included as a fixed effect and the other terms remained the same as model 1. Since we also wanted to explore whether the monitored years after restoration and project size showed interaction effects (hypothesis ii), $(\mathrm{d}t)^2$*lentic project size was added to model 2 as a fixed effect. No interaction effects were detected between the monitored years after restoration and lotic project size, therefore only lentic project size was included in our models (Table 1).

Exploration of responses among separate biotic indices was undertaken for phytoplankton (model 4) and invertebrates (model 5), while other biotic indices did not have enough observations for their own models. Relationship between the specific abiotic indices (NH_4^+-N, TN, TP, BOD) were evaluated alongside these separate biotic indices (Δr phytoplankton, Δr invertebrates), with the category of environment indicators including the specific abiotic indices and phytoplankton/invertebrates as a fixed effect, and other terms the same as model 1 (Table 1). While the significant difference between lentic and lotic ecosystems was tested again, one LMM model (model 6) was applied to the response ratio of phytoplankton and all abiotic indices for lentic ecosystem. Model terms were as per model 4, except project size was included rather than ecosystem type. Finally, a LMM model (model 7) was applied to the response ratio of invertebrates and all abiotic indices for lotic ecosystem, with similar terms as model 5 (Table 1). Other models for lentic and lotic phytoplankton and invertebrates were not included because of the limited number of observations. Only abundance/biomass sub data were used for model 4 to model 7, due to the limited richness/diversity data of each organism group.

For each model structure (Table 1), we performed model selection to search for the most parsimonious model based on the Akaike's Information Criterion (AIC). Model residuals were tested for compliance with model assumptions (Crawley, 2002), and spatial and temporal autocorrelation with Moran's tests (Birk et al., 2020).

To investigate the spatial heterogeneity and restoration scheme variance between abiotic and biotic indices response to restoration, we calculated the coefficient of variation (CV) of the response ratio of abiotic and biotic indices over the monitored years after restoration in the first three models (model 1 to model 3) and used a Kruskal-Wallis test to examine whether they differed. All data analysis was performed using R 4.0.1 (R Core Team 2020, https://www.R-project.org/).

3 Results

3.1 Overall response of abiotic/biotic indices after restoration

Restoration works were found to be efficient at recovering freshwater ecosystems from their initial degraded condition, as shown by their significant effect on almost all the assessed abiotic and biotic indices except for birds (Fig. 2, Fig. S2). Mean response ratios of the concentrations of NH_4^+-N, TN, TP and BOD_5 were overall positive (all $p < 0.001$, Fig. 2). Biotic indices for fish, invertebrates, phytoplankton and vascular plants were significantly higher after restoration, as illustrated by generally positive response ratios (all $p < 0.05$, Fig. 2). Furthermore, the biotic response of abundance/biomass and richness/diversity of each organism were different (Fig. S4). The improvement of aquatic ecosystems (denoted by positive response ratio of abiotic and biotic indices) increased with time elapsed since restoration (Fig. 3).

Examination of marginal effects showed that, lentic ecosystems had a significantly higher positive response to restoration compared to lotic ecosystems (n = 1653, marginal $R^2 = 0.10$, $p < 0.05$, Fig. 3 a). Post-restoration recovery of biotic indices almost always lagged behind abiotic indices in lentic and lotic ecosystems. For lentic ecosystems, the response ratio of abiotic indices reached its recovery peak 9.3 years from restoration, the response ratio of biotic indices was still rising by the end of the monitored period (n = 1130, marginal $R^2 = 0.11$, $p < 0.05$, Fig. 3 b). Nonetheless, the limited duration of the monitored years after restoration for lotic ecosystems (≤ 9 years, Fig. 3 c), meant the peaks of the response ratio for the abiotic and biotic indices were not reached in many instances and highlighting the need for longer-term monitoring efforts.

The response ratio of abiotic and biotic indices increased with smaller lentic project size (n = 1130, marginal $R^2 = 0.14$, $p < 0.01$, Fig. 4 a). A significant interaction between the monitored years after restoration and the size of lentic project was evident for the response ratio of all the abiotic and biotic indices (n = 1130, $p < 0.05$, Fig. 5). For example, higher abiotic and biotic index responses were associated with time after restoration, but these effects were much weaker for larger project size (Fig. 5), irrespective of the number of monitored years elapsed since restoration. However, this interactive effect was not detected for lotic ecosystems (n = 505, $p = 0.29$, Fig. 4 b).

The coefficient of variation for the response ratio of biotic indices (CV= 0.23 ± 0.03) was significantly higher than abiotic indices (CV= 0.18 ± 0.05) across all the freshwater ecosystems ($p < 0.001$, Fig 6), and was even obvious in lotic ecosystems (CV of Δr biotic =0.90 ± 0.39, CV of Δr abiotic =0.40 ± 0.08) (Fig. 6). The higher variability of the response ratio of biotic indices was particularly notable at the initial stage after restoration. No significant difference was found for the variance of AAT10 on abiotic and biotic variable responses to the restoration effort.

3.2 Specific abiotic and biotic responses after restoration efforts

Examination of the marginal effects showed that in lentic ecosystems the response ratio of NH_4^+-N, TN and TP concentrations peaked and then declined approximately 8 - 9 years after restoration. In contrast, the response ratio for BOD_5 and the abundance/biomass of phytoplankton increased consistently over time after restoration (n = 1087, marginal $R^2 = 0.13$, $p < 0.05$, Fig. 7c). In lotic ecosystems, the response ratio of all abiotic and biotic indices almost always increased over time because of the limited monitored years after restoration; however, the response ratio of BOD_5 gradually peaked and declined slightly around 6.5 years after restoration (n = 437, marginal $R^2 = 0.14$, $p < 0.05$, Fig. 7d).

4 Discussion

Long-term monitoring of freshwater ecosystems following restoration is often underfunded or not well documented, especially outside of Europe and North America (Jeppesen et al., 2005; Scamardo and Wohl, 2020), leading to scarce understanding of biotic and abiotic responses (Kail et al., 2015). Our study of long-term (up to 13 years) freshwater ecosystem responses following the restoration at a large spatial scale (China mainland) has showed that: (1) Over > 10 years post-restoration, the response of biotic indices always lagged behind abiotic indices in both lentic and lotic freshwater ecosystems; (2) post-restoration response of abiotic and biotic indices in lentic ecosystems was significantly greater than lotic, but smaller lentic ecosystems can be more easily restored than larger ones; (3) Spatial environmental heterogeneity coupled with different combinations of restoration measures and restoration efforts (e.g., investments, amount of each specific measures and position, etc.) drove the significantly higher variance of biotic index response ratios than abiotic indices, especially in lotic ecosystems.

By integrating some of the longest available monitoring time-series data, our results demonstrate that restoration projects effectively improved the abiotic and biotic conditions of aquatic ecosystems over time (Fu et al., 2021; Huang et al., 2019), except birds. Particularly, we observed that the response ratio of abundance/biomass and richness/diversity had different density distribution for each organism (fish, invertebrates, phytoplankton, vascular plants). It reflects the different dimensions of the biotic indices (quality (richness/diversity) vs. quantity (abundance/biomass)) response to post-restoration. Possible reason could be: the recovery time for one type of biotic indices lags the other, for example, perhaps increases in abundance of a few species are easier to attain than the increase in richness after restoration. However, we cannot get more detail for the limited sample size and asymmetry biotic data were documented (Table S4).

In agreement with our first hypothesis, we found the quantitative evidence of continuous lagged biotic responses at a long-term scale: the response ratio of abiotic indices declined in lentic ecosystems after about 9.3 years post-restoration, while the response ratio of biotic indices was still rising even in the longest post-restoration monitored sites (i.e., 13 years after restoration). However, in some situations, restoration effects could gradually vanish over time unless careful monitoring of changes is used to inform further restoration maintenance. As Kail et al. (2015) noted, macrophyte abundance increased at the beginning of some restoration schemes but decreased during the following years. The lack of persistence in some restored conditions might illustrate that conditions such as sediment transport and deposition or altered hydrodynamic processes were not successfully restored, that other long-term shifts (e.g. global warming) continue to impart changes (Boerema et al., 2016), or that further catchment development imparts further water quality issues over the long-term (Meals et al., 2010).

In many of the studies that we reviewed, restoration targeting pollution sources such as sewage interception usually was the first step of aquatic restorations. Additionally, common projects included targeting pollution-sinks such as removal of contaminated sediment, followed by submerged macrophyte reintroduction and riparian buffer zone planting. As a consequence, water quality improvements were typically rapid with pollutant loads reduced quickly. In contrast, the response lag for biotic indices likely relates to dispersal and establishment limitations which are common, and several recent reviews of metacommunity theory and practice in freshwaters have therefore advocated for the potential reintroduction of aquatic assemblages (Cid et al., 2021; Patrick et al., 2021). Although reintroduced organisms (e.g. macroinvertebrates, filter-feeding fish (*Hypophthalmichthys molitrix*, *Aristichthys nobilis*), plants) have been common in Chinese restoration projects (Table S1), multi-species communities require additional time to recolonize rapidly altered habitats/niche and establish viable populations (Lorenz et al., 2018). Augmented dispersal may not always translate into the establishment of stable local populations because some species may be unable to survive and successfully reproduce (Coulon et al., 2010). This could illustrate a need for managed reintroductions to consider temporally-staged assisted migrations in line with successional theory, as physical, chemical and biological components of the ecosystem change over time according to the starting conditions. Additionally, biotic time lags might be related to the carrying capacity of the ecosystem: water quality and habitat need to establish and succeed for a longer period of time to support a wider diversity of species than those introduced initially (Patrick et al., 2021). Finally, time lags of recovery of different species are highly variable because of different life-span and fecundity, with short-lived species expected to display short time-lags (Watts et al., 2020).

Response ratios of the concentrations of all nutrients (NH_4^+-N, TN, TP) peaked 8 - 9 years after restoration in lentic ecosystems. As a consequence, concentrations of phytoplankton were subsequently reduced significantly, linked to the decline of TP concentrations in water and probably accompanied by zooplankton and fish community structure change (Jeppesen et al., 2005). However, the response ratio of TP also showed a relatively rapid increase immediately post-restoration, most likely reflecting the widespread dredging of polluted sediment which often accompanied reduction of external nutrient loading. Thus, the response ratio of TP peaked earlier and decreased faster than NH_4^+-N and TN, in line with Li et al. (2022) findings for Lake Wuli, China. Here, sewage interception and denitrification reduced N in by >70%, but had less impact on P illustrating the important role of sedimentary cycling. Response ratios of BOD_5 and abundance/biomass of phytoplankton were still improving after 9.3 years in lentic ecosystems, and significantly positive correlations were evident with BOD_5 and all other biotic indices. These results are possibly caused by the interactions of vascular plants, invertebrates and phytoplankton leading to a more clear water state (Brett et al., 2017). Alternatively, the results may reflect more effective colonization of aquatic plants and the successful (stable) establishment of healthier habitat conditions as water quality has improved. In addition, peaks of the response ratio of abiotic or biotic indices in lotic ecosystems were not observed in our study (except for BOD_5) given the limited monitoring years after restoration (≤ 9 years). Further analysis of other organisms including birds, fish and vascular plants was not possible due to the limited sample size, and illustrates the lack of consistency of biological monitoring post-restoration.

Our analysis confirmed that the responses of abiotic and biotic indices in lentic ecosystems were significantly greater compared with lotic ecosystems. This is consistent with our second hypothesis, and supported by Verdonschot et al. (2013) who qualitatively concluded that the successful restoration rate of lakes from eutrophication and acidification was higher than most rivers. This finding reflects the complexity of hydrology, hydraulics and morphology in the lotic ecosystem, and river restoration can involve changes to the physical, chemical, biological and hydrological components of the system (Speed et al., 2016) as well as the core targets of restoration schemes. In lentic ecosystems, the reduction of external nutrient loadings, removal of contaminated sediments and direct point pollution sources can be addressed easier at a whole lake, provided the catchment area is not extensive. In particular, we demonstrated that smaller project size of lentic ecosystems can be more easily restored than larger ones. Moreover, our results demonstrate that interactions between time since restoration and the size of lentic projects can eventually result in different restoration effects. Therefore, even with the same proportional efforts of restoration, a larger project size of lentic ecosystems did not achieve the same proportional response as smaller systems (Fig. 5). This may be due to larger lentic ecosystems being able to support longer food-chain length and biodiversity, in addition to offering more complex and diverse habitats (Post et al., 2000). In contrast for lotic ecosystems, whole upstream catchment restorations will often be necessary to achieve positive responses within a selected restoration reach.

Our study illustrated that spatial heterogeneity and restoration scheme effects introduced more variability to biotic indices response after restoration than abiotic indices. This effect was especially strong in lotic ecosystems, in line with our third hypothesis. Whilst abiotic parameters can often be controlled in a strongly deterministic manner, organisms with different niches are influenced by physicochemical and biological factors as well as dispersal success in more complex ways, leading to greater stochasticity (Cid et al., 2021; Thompson and Townsend, 2006). Kail et al. (2015) also reported the high variability of the response ratio of fish, macroinvertebrates and aquatic macrophytes after river restoration (without incorporation with abiotic indices), and indicated that many factors (e.g., organism group, restoration measures) can contribute to the different variability range of response ratio. Possible reasons for the considerable high variability of the response ratio of biotic indices in lotic ecosystems compared to lentic ones could be due to the flow-biota-ecosystem processes nexus in lotic ecosystems. These linkages exert direct and indirect control on the dynamics of organism communities at local to regional scale. This can make it difficult to restore fragmented river network habitats at a local scale (Palmer and Ruhi, 2019), unless whole catchment complementary approaches are undertaken.

No significant influence of different climatic zones (AAT10) was detected on aquatic ecosystem restoration effects in our study, contradicting our expectations for hypothesis four. Possible reasons are likely to include the diversity of ecosystems considered amongst the multiple abiotic or biotic indices that were integrated in the meta-analysis. Stronger biogeographic responses linked to climate are more likely to be observed in studies where similar restoration interventions and identical monitoring protocols are implemented along a latitudinal gradient. In addition, the practice of augmented dispersal by incorporating species reintroduction of local plants and animals that then adapt to the local climate conditions will significantly blur the boundaries between natural, climatically driven processes and recovery from human modifications. Whilst an optimum annual accumulated mean daily temperature above 10 °C is considered to enable more successful biodiversity recovery (Dong et al., 2009), more data is required to validate this supposition. For example, in extremely warm environments, the stimulation of algal growth extends the duration of eutrophication and algal blooms (Nazari-Sharabian et al., 2018; Xiong et al., 2016), thus making conditions less favorable for ecosystem recover despite attempts at restoration. Further study is needed to understand the role of large-scale biogeographic effects on aquatic restoration recovery across China.

Overall, generally positive response ratios were observed across most aquatic ecosystems in our study, for a range of restoration schemes spanning lentic and lotic ecosystems. We highlight the importance of continued nutrient reductions (Lefcheck et al., 2018) and continuous long-term monitoring after restoration, especially for lotic ecosystems. The heterogeneity of available data despite decades of ecosystem restoration in China underscores the need for stricter monitoring and data reporting/sharing protocols after restoration, particularly for biotic indices. Such advances could be made following procedures that are utilized as part of chemical monitoring programs that form China's official standards for surface water (GB3838-2002).

5 Conclusion

Our findings provide quantitative evidence that abiotic and biotic indices recovery after restoration differ in lentic and lotic ecosystems over large spatial scales. We highlight that the response of biotic indices lags behind abiotic indices for a longer period (over 10 years) post-restoration, and the restoration effect can decline without continuous further restoration or maintenance projects. Our results suggest that lentic ecosystems are typically easier to restore than lotic ones, but larger lentic ecosystems need greater and disproportional restoration efforts compared to smaller ones. Moreover, considerably higher variability in the response ratio of biotic indices to restoration efforts was observed, particularly in lotic ecosystems. Finally, our results show that the response ratios were not related to climatic zones represented in China mainland. Our research shows the need for long-term and enhanced biological monitoring post-restoration, if river managers wish to improve future restoration effects. When defining restoration targets, we encourage attention to the different timelines for the recovery of abiotic and biotic indices after restoration.

Acknowledgments

This research was supported by the National Key R&D Program of China (Grant No. 2018YFD0900904), the Water Pollution Control and Management Project of China (Grant No. 2018ZX07208005), the International Cooperation Project of the Chinese Academy of Sciences (Grant No. 152342KYSB20190025), the National Natural Science Foundations of China (Grant No. 31872687) and the China Scholarship Council. Many thanks to Pamela Scobie for her revision of the English language of this paper. LEB and MK’s time contributions were funded by the School of Geography, University of Leeds. JGM was supported by JSPS KAKENHI Grant No. 19H04314.

Author contributions

Conceptualization, J.X.; methodology, J.X., H.F., L.E.B. and J.G.M.; formal analysis, H.F.; resources, H.F. and J.X.; writing—original draft preparation, H.F.; writing-review and editing, H.F., J.G.M., H.Z., M.Z., L.E.B., MK and J.X.; supervision, J.X., L.E.B., MK and M.Z.

Conflict of interest

The authors declare no conflict of interest.

References

Bai, G., Zhang, Y., Yan, P., Yan, W., Kong, L., Wang, L., Wang, C., Liu, Z., Liu, B. and Ma, J. 2020. Spatial and seasonal variation of water parameters, sediment properties, and submerged macrophytes after ecological restoration in a long-term (6 year) study in Hangzhou west lake in China: Submerged macrophyte distribution influenced by environmental variables. Water Res. 186, 116379.

Bates, D., Mächler, M., Bolker, B. and Walker, S. 2014. Fitting linear mixed-effects models using lme4. arXiv preprint arXiv:1406.5823.

Benayas, J.M.R., Newton, A.C., Diaz, A. and Bullock, J.M. 2009. Enhancement of biodiversity and ecosystem services by ecological restoration: A meta-analysis. Science 325(5944), 1121-1124.

Bernhardt, E.S., Palmer, M.A., Allan, J., Alexander, G., Barnas, K., Brooks, S., Carr, J., Clayton, S., Dahm, C. and Follstad-Shah, J. 2005. Synthesizing US river restoration efforts. Science 308(5722), 636-637.

Birk, S., Chapman, D., Carvalho, L., Spears, B.M., Andersen, H.E., Argillier, C., Auer, S., Baattrup-Pedersen, A., Banin, L. and Beklioğlu, M. 2020. Impacts of multiple stressors on freshwater biota across spatial scales and ecosystems. Nature Ecology & Evolution 4(8), 1060-1068.

Boerema, A., Geerts, L., Oosterlee, L., Temmerman, S. and Meire, P. 2016. Ecosystem service delivery in restoration projects: the effect of ecological succession on the benefits of tidal marsh restoration. Ecol. Soc. 21(2).

Brett, M.T., Bunn, S.E., Chandra, S., Galloway, A.W., Guo, F., Kainz, M.J., Kankaala, P., Lau, D.C., Moulton, T.P. and Power, M.E. 2017. How important are terrestrial organic carbon inputs for secondary production in freshwater ecosystems? Freshwater Biology 62(5), 833-853.

Carvalho, L., Mackay, E.B., Cardoso, A.C., Baattrup-Pedersen, A., Birk, S., Blackstock, K.L., Borics, G., Borja, A., Feld, C.K. and Ferreira, M.T. 2019. Protecting and restoring Europe's waters: An analysis of the future development needs of the Water Framework Directive. Science of the Total Environment 658, 1228-1238.

Cid, N., Erős, T., Heino, J., Singer, G., Jähnig, S.C., Cañedo‐Argüelles, M., Bonada, N., Sarremejane, R., Mykrä, H. and Sandin, L. 2021. From meta‐system theory to the sustainable management of rivers in the Anthropocene. Front. Ecol. Environ. 20(1), 49-57.

Coulon, A., Fitzpatrick, J.W., Bowman, R. and Lovette, I.J. 2010. Effects of habitat fragmentation on effective dispersal of Florida scrub‐jays. Conservation Biology 24(4), 1080-1088.

Crawley, M.J. (2002) Statistical computing an introduction to data analysis using S-Plus.

Dong, J., Liu, J., Tao, F., Xu, X. and Wang, J. 2009. Spatio-temporal changes in annual accumulated temperature in China and the effects on cropping systems, 1980s to 2000. Clim. Res. 40(1), 37-48.

Fu, H., Gaüzère, P., Molinos, J.G., Zhang, P., Zhang, H., Zhang, M., Niu, Y., Yu, H., Brown, L.E. and Xu, J. 2021. Mitigation of urbanization effects on aquatic ecosystems by synchronous ecological restoration. Water Res. 204, 117587.

Fukami, T. 2004. Assembly history interacts with ecosystem size to influence species diversity. Ecology 85(12), 3234-3242.

Huang, J., Zhang, Y., Arhonditsis, G.B., Gao, J., Chen, Q., Wu, N., Dong, F. and Shi, W. 2019. How successful are the restoration efforts of China's lakes and reservoirs? Environ. Int. 123, 96-103.

Janse, J., Kuiper, J., Weijters, M., Westerbeek, E., Jeuken, M., Bakkenes, M., Alkemade, R., Mooij, W. and Verhoeven, J. 2015. GLOBIO-Aquatic, a global model of human impact on the biodiversity of inland aquatic ecosystems. Environmental Science & Policy 48, 99-114.

Jeppesen, E., Søndergaard, M., Jensen, J.P., Havens, K.E., Anneville, O., Carvalho, L., Coveney, M.F., Deneke, R., Dokulil, M.T. and Foy, B. 2005. Lake responses to reduced nutrient loading–an analysis of contemporary long - term data from 35 case studies. Freshwater biology 50(10), 1747-1771.

Kail, J., Brabecb, K., Poppec, M. and Januschkea, K. 2015. The effect of river restoration on fish, macroinvertebrates and aquatic macrophytes: A meta-analysis. Ecological Indicators 58, 311–321.

Kummu, M., De Moel, H., Ward, P.J. and Varis, O. 2011. How close do we live to water? A global analysis of population distance to freshwater bodies. PloS one 6(6), e20578.

Kuznetsova, A., Brockhoff, P.B. and Christensen, R.H.B. 2015. Package 'lmertest'. R package version 2(0).

Lefcheck, J.S., Orth, R.J., Dennison, W.C., Wilcox, D.J., Murphy, R.R., Keisman, J., Gurbisz, C., Hannam, M., Landry, J.B. and Moore, K.A. 2018. Long-term nutrient reductions lead to the unprecedented recovery of a temperate coastal region. Proceedings of the National Academy of Sciences 115(14), 3658-3662.

Levi, P.S. and McIntyre, P.B. 2020. Ecosystem responses to channel restoration decline with stream size in urban river networks. Ecological Applications 30(5), e02107.

Li, Y., Shen, R., Liu, X., Su, H., Wu, F. and Chen, F. 2022. Impacts of nutrient reduction on temporal β-diversity of rotifers: A 19-year limnology case study on Lake Wuli, China. Water Res. 216, 118364.

Liang, W. and Yang, M. 2019. Urbanization, economic growth and environmental pollution: Evidence from China. Sustainable Computing: Informatics and Systems 21, 1-9.

Liu, S., Dong, Y., Cheng, F., Coxixo, A. and Hou, X. 2016. Practices and opportunities of ecosystem service studies for ecological restoration in China. Sustainability science 11(6), 935-944.

Lorenz, A.W., Haase, P., Januschke, K., Sundermann, A. and Hering, D. 2018. Revisiting restored river reaches–Assessing change of aquatic and riparian communities after five years. Science of the total environment 613, 1185-1195.

Lu, W., Font, R.A., Cheng, S., Wang, J. and Kollmann, J. 2019. Assessing the context and ecological effects of river restoration–A meta-analysis. Ecological Engineering 136, 30-37.

Meals, D.W., Dressing, S.A. and Davenport, T.E. 2010. Lag time in water quality response to best management practices: A review. Journal of environmental quality 39(1), 85-96.

Miller, S.W., Budy, P. and Schmidt, J.C. 2010. Quantifying macroinvertebrate responses to in‐stream habitat restoration: applications of meta‐analysis to river restoration. Restoration Ecology 18(1), 8-19.

Nazari-Sharabian, M., Ahmad, S. and Karakouzian, M. 2018. Climate change and eutrophication: a short review. Engineering, Technology and Applied Science Research 8(6), 3668.

Palmer, M. and Ruhi, A. 2019. Linkages between flow regime, biota, and ecosystem processes: Implications for river restoration. Science 365(6459), eaaw2087.

Patrick, C.J., Anderson, K.E., Brown, B.L., Hawkins, C.P., Metcalfe, A., Saffarinia, P., Siqueira, T., Swan, C.M., Tonkin, J.D. and Yuan, L.L. 2021. The application of metacommunity theory to the management of riverine ecosystems. Wiley Interdisciplinary Reviews: Water 8(6), e1557.

Post, D.M., Pace, M.L. and Hairston, N.G. 2000. Ecosystem size determines food-chain length in lakes. Nature 405(6790), 1047-1049.

Rakap, S., Rakap, S., Evran, D. and Cig, O. 2016. Comparative evaluation of the reliability and validity of three data extraction programs: UnGraph, GraphClick, and DigitizeIt. Computers in Human Behavior 55, 159-166.

Rasheduzzaman, M., Singh, R., Haas, C.N. and Gurian, P.L. 2020. Required water temperature in hotel plumbing to control Legionella growth. Water Res. 182, 115943.

Scamardo, J. and Wohl, E. 2020. Sediment storage and shallow groundwater response to beaver dam analogues in the Colorado Front Range, USA. River Research and Applications 36(3), 398-409.

Speed, R.A., Li, Y., Tickner, D., Huang, H., Naiman, R.J., Cao, J., Lei, G., Yu, L., Sayers, P. and Zhao, Z. 2016. A framework for strategic river restoration in China. Water International 41(7), 998-1015.

Thompson, R. and Townsend, C. 2006. A truce with neutral theory: local deterministic factors, species traits and dispersal limitation together determine patterns of diversity in stream invertebrates. J. Anim. Ecol. 75(2), 476-484.

Van Klink, R., Bowler, D.E., Gongalsky, K.B., Swengel, A.B., Gentile, A. and Chase, J.M. 2020. Meta-analysis reveals declines in terrestrial but increases in freshwater insect abundances. Science 368(6489), 417-420.

Verdonschot, P., Spears, B., Feld, C., Brucet, S., Keizer-Vlek, H., Borja, A., Elliott, M., Kernan, M. and Johnson, R. 2013. A comparative review of recovery processes in rivers, lakes, estuarine and coastal waters. Hydrobiologia 704(1), 453-474.

Wang, H., Wang, Q., Bowler, P.A. and Xiong, W. 2016. Invasive aquatic plants in China. Aquatic Invasions 11(1), 1-9.

Watts, K., Whytock, R.C., Park, K.J., Fuentes-Montemayor, E., Macgregor, N.A., Duffield, S. and McGowan, P.J. 2020. Ecological time lags and the journey towards conservation success. Nature ecology & evolution 4(3), 304-311.

Xiong, J., Wang, X.C., Zhang, Q., Duan, R. and Wang, N. 2016. Characteristics of a landscape water with high salinity in a coastal city of China and measures for eutrophication control. Ecological Indicators 61, 268-273.

Yang, K., Yu, Z., Luo, Y., Yang, Y., Zhao, L. and Zhou, X. 2018. Spatial and temporal variations in the relationship between lake water surface temperatures and water quality-A case study of Dianchi Lake. Science of the total environment 624, 859-871.

Zhang, H., Hollander, J. and Hansson, L.-A. 2017. Bi-directional plasticity: rotifer prey adjust spine length to different predator regimes. Scientific reports 7(1), 1-7.

Zhou, Y., Ma, J., Zhang, Y., Qin, B., Jeppesen, E., Shi, K., Brookes, J.D., Spencer, R.G.M., Zhu, G. and Gao, G. 2017. Improving water quality in China: Environmental investment pays dividends. Water Res. 118, 152-159.

Figures legends

Figure. 1. Spatial distribution of monitored sites (*n* = 157). The Hu Huanyong Line is traditionally used as a geographic boundary between the highly developed and densely populated Eastern region, where most restoration projects are located, and the less-developed and sparsely populated Western region in China. The inset shows the histogram of monitored years across all documented ecosystems. AAT10 (the annual accumulated mean daily temperature above 10 °C) used as a proxy for large scale climatic zones.

Figure. 2. Response ratios of abiotic (NH_4^+-N, TN, TP, BOD_5) and biotic (richness/diversity and abundance/biomass of birds, fish, invertebrates, phytoplankton and vascular plants) indices in restored compared with degraded (i.e., pre-restoration) aquatic ecosystems. All response ratios differed significantly from zero except for birds (Wilcoxon signed rank tests, all the p values < 0.05, effect size $r = 0.68$). The mean and standard deviation are given alongside the overall data distribution for each metric.

Figure. 3. Marginal effects of the response ratio of abiotic and biotic indices in lentic and lotic aquatic ecosystem over the monitored years after restoration (a) (model 1). Interaction effect between the monitored years after restoration (dt) and indicators category (abiotic VS biotic) on the response ratio of abiotic and biotic indices in lentic (b) (model 2) and lotic (c) (model 3) aquatic ecosystems.

Figure. 4. Marginal effects of the project size of (a) lentic aquatic ecosystem (results from model 2) and (b) lotic project size on the whole response ratios of abiotic and biotic indices (results from model 3). Lentic project size (km ^{2}) was $\log_{10}$ transformed.

Figure. 5. Interaction effect between monitored years after restoration and project/ecosystem size of lentic ecosystems on the response ratio of all the abiotic and biotic indices ($p < 0.05$, results from model 2). Lentic project size (km ^{2}) was $\log_{10}$ transformed.

Figure. 6. Differences of the coefficient of variation between the response ratio of abiotic and biotic indices in both lentic and biotic aquatic ecosystem with significant differences at $p < 0.01$ (Wilcoxon's test) (a) (model 1). The coefficient of variation (CV) between response ratio of abiotic and biotic indices along the years after restoration in (b) (model 2) lentic and (c) (model 3) lotic aquatic ecosystems.

Figure. 7. Marginal effects of the response ratio of individual abiotic and biotic indices in lentic and lotic aquatic ecosystems over the monitored years after restoration. (a), results from model 4; (b), results from model 5; (c), results from model 6; (d), results from model 7.

Tables legends

Table 1. Linear Mixed Models (LMM) used in this study. Rows in grey show the models include both lentic and lotic ecosystems. dt, monitored years after restoration.

Table 1. Linear Mixed Models (LMM) used in this study. Rows in grey show the models include both lentic and lotic ecosystems. dt, monitored years after restoration.

Model No.	**Dependent variable**	**Fixed effects**	**Random effects**	**Random slop effects**	**Ecosystem type included**
1	$-\Delta r$ abiotic and biotic	$(dt)^2$*abiotic vs. biotic; ecosystem type; AAT10	the category of each abiotic and biotic indices; ecosystem ID; start date of the restoration	dt|sites	lentic and lotic
2	$-\Delta r$ abiotic and biotic	$(dt)^2$*abiotic vs. biotic; $(dt)^2$*project size; AAT10	the category of each abiotic and biotic indices; ecosystem ID; start date of the restoration	dt|sites	lentic
3	$-\Delta r$ abiotic and biotic	$(dt)^2$*abiotic vs. biotic; project size; AAT10	the category of each abiotic and biotic indices; ecosystem ID; start date of the restoration	dt|sites	lotic
4	$-\Delta r$ abiotic & phytoplankton	$(dt)^2$* the category including each abiotic indices & phytoplankton; ecosystem type; AAT10	ecosystem ID; start date of the restoration	dt|sites	lentic and lotic
5	$-\Delta r$ abiotic & Δr invertebrates	$(dt)^2$* the category including each abiotic indices & invertebrates; ecosystem type; AAT10	ecosystem ID; start date of the restoration	dt|sites	lentic and lotic

6	$-\Delta r$ abiotic & phytoplankton	$(dt)^2$* the category including each abiotic indices & phytoplankton; project size; AAT10	ecosystem ID; start date of the restoration	dt|sites	lentic
7	$-\Delta r$ abiotic & Δr invertebrates	$(dt)^2$* the category including each abiotic indices & invertebrates; project size; AAT10	ecosystem ID; start date of the restoration	dt|sites	lotic