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

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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 months–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 down 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 that, 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.

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 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;

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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, 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. In 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) 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 less thatn 20% to more than 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 1000 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) claimed 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 lake (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 time 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 scales. 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 December 19, 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 (of which 74 were from CNKI and 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 (on average 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 a more focused meta-analysis. We extracted information on all

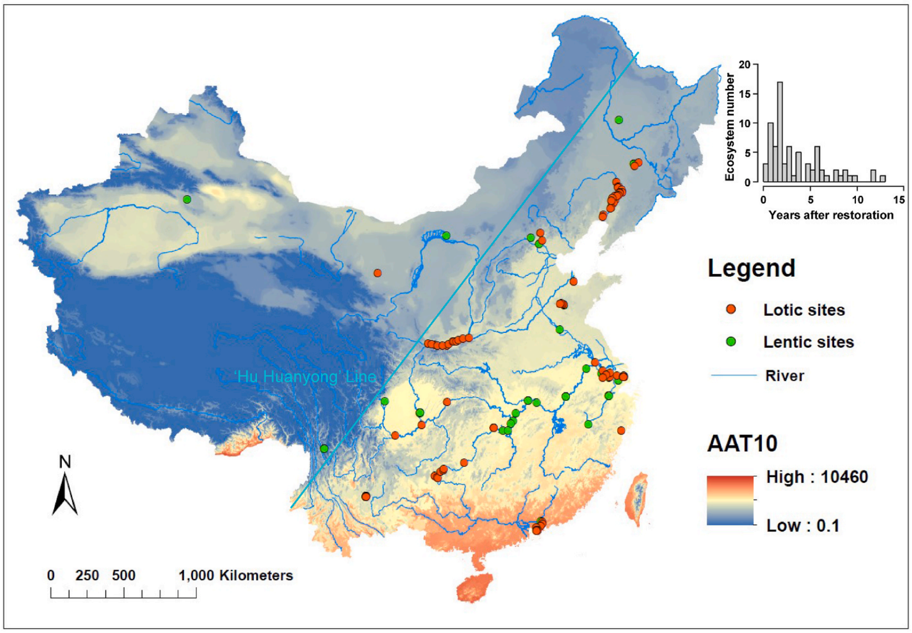

Fig. 1. Spatial distribution of monitored sites (n = 157). The Hu Huanyong Line is traditionally used as a geographic boundary between the highly developed and densely populated Eastern region, where most restoration projects are located, and the less-developed and sparsely populated Western region in 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.

variables related 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. The biotic indices considered, which include abundance/biomass and richness/diversity of organisms, relate 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 by 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 ecosystem but for different time periods. These were combined to avoid pseudo replication. Additionally, we deconstucted 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 (DigitizeIt, 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 from 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)).

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

where *After Restoration/Degraded* refers to the ratio of the values of a specific biotic or abiotic metric at the monitored site after and before restoration or, where the latter was not available, 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 (Δr NH_4^+-N, ΔrTN, ΔrP, Δr BOD) 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 (dt), 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 dt 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), $(dt)^2$*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 all other terms kept the same as for 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, which replaced 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 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, 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

Table 1
Linear Mixed Models (LMM) used in this study. dt referst to the number of monitored years after restoration.

Model No.	Dependent variable	Fixed effects	Random effects	Random slop effects	Ecosystem type included
1	-Δr 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	-Δr 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	-Δr 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	-Δr 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	-Δr 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	-Δr 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	-Δr 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

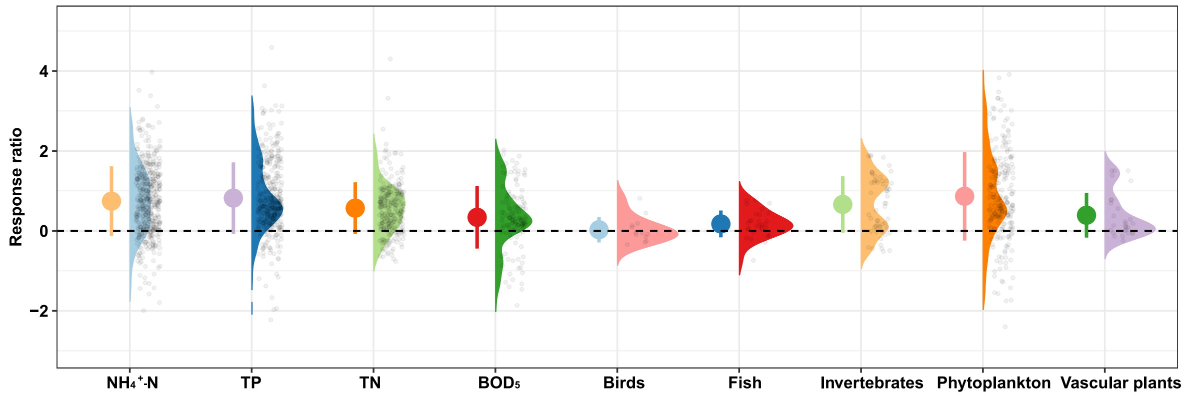

Fig. 2. Response ratios of abiotic (NH_4^+-N, TN, TP, BOD_5) and biotic (richness/diversity and abundance/biomass of birds, fish, invertebrates, phytoplankton and vascular plants) indices in restored compared with degraded (i.e. pre-restoration) aquatic ecosystems. All response ratios differed significantly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

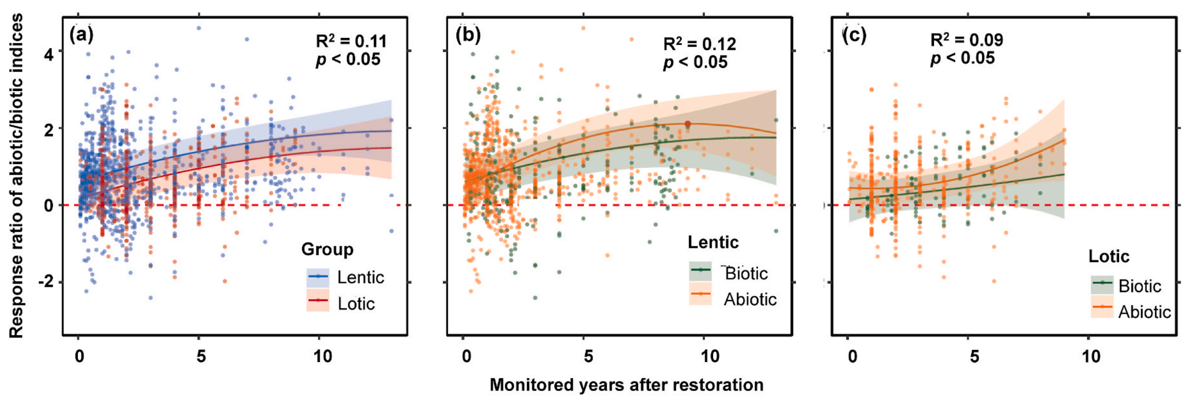

Fig. 3. Marginal effects of the response ratio of abiotic and biotic indices in lentic and lotic aquatic ecosystem over the monitored years after restoration (a) (model 1). Interaction effect between the monitored years after restoration (dt) and indicators category (abiotic VS. biotic) on the response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

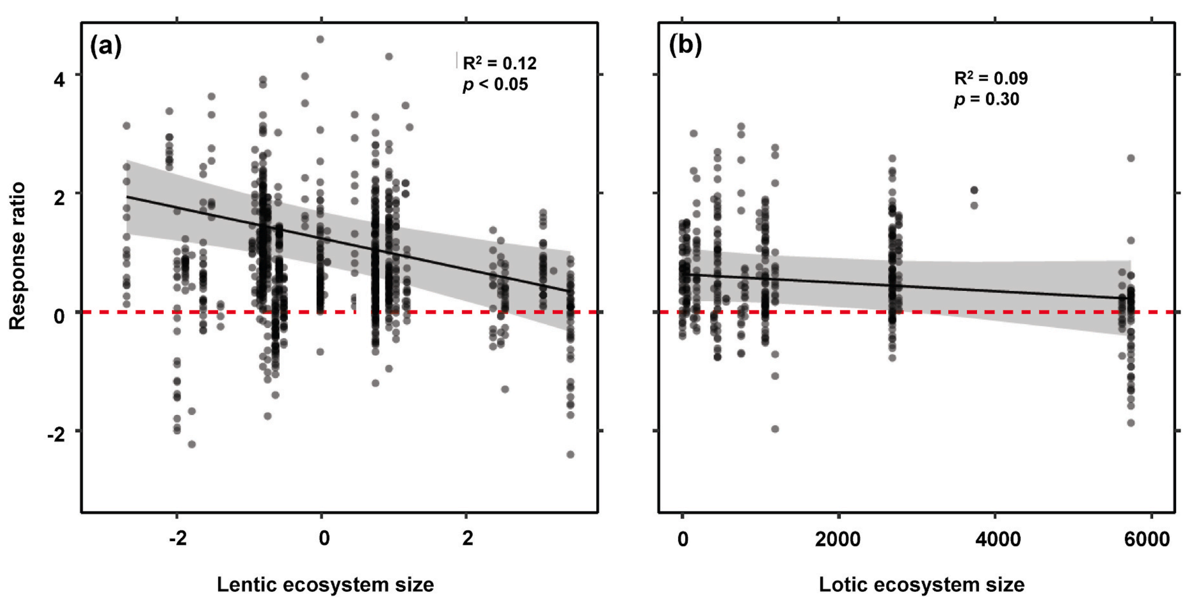

Fig. 4. Marginal effects of the project size of (a) lentic aquatic ecosystem (results from model 2) and (b) lotic project size on the whole response ratios of abiotic and biotic indices (results from model 3). Lentic project size (km ^{2}) was $\log_{10}$ transformed.

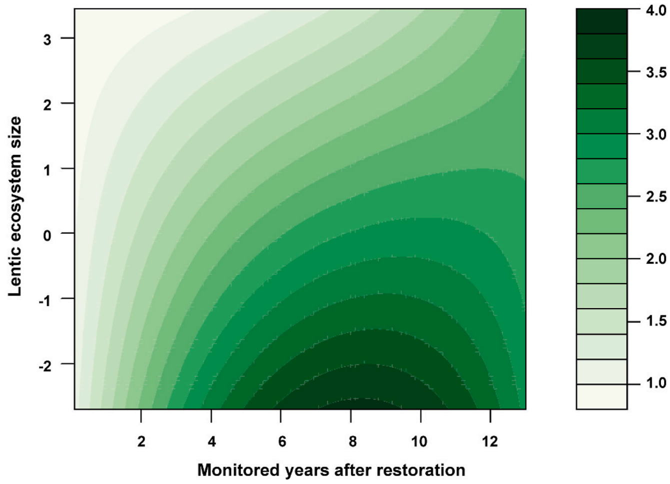

Fig. 5. Interaction effect between monitored years after restoration and project/ecosystem size of lentic ecosystems on the response ratio of all the abiotic and biotic indices ($p < 0.05$, results from model 2). Lentic project size (km^2) was $\log_{10}$ transformed.

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) 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 for 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). This reflects the different dimensions of the biotic indices (quality (richness/diversity) vs. quantity (abundance/biomass)) response to post-restoration. A possible reason could be that the recovery time for one type of biotic indices lags the other. For example, increases in abundance of a few species might be easier to attain than the increase in richness after restoration. However, we cannot get more detail given 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

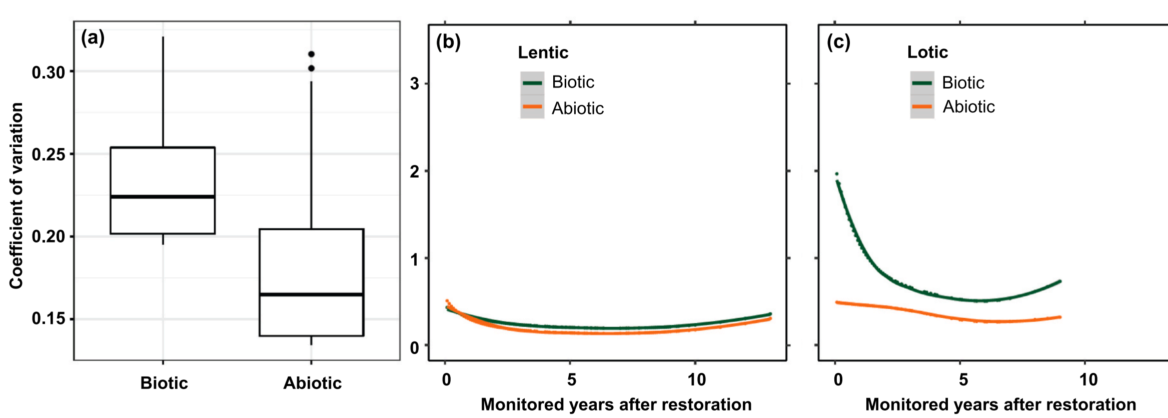

Fig. 6. Differences of the coefficient of variation between the response ratio of abiotic and biotic indices in both lentic and biotic aquatic ecosystem with significant differences at $p < 0.01$ (Wilcoxon's test) (a) (model 1). The coefficient of variation (CV) between response ratio of abiotic and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

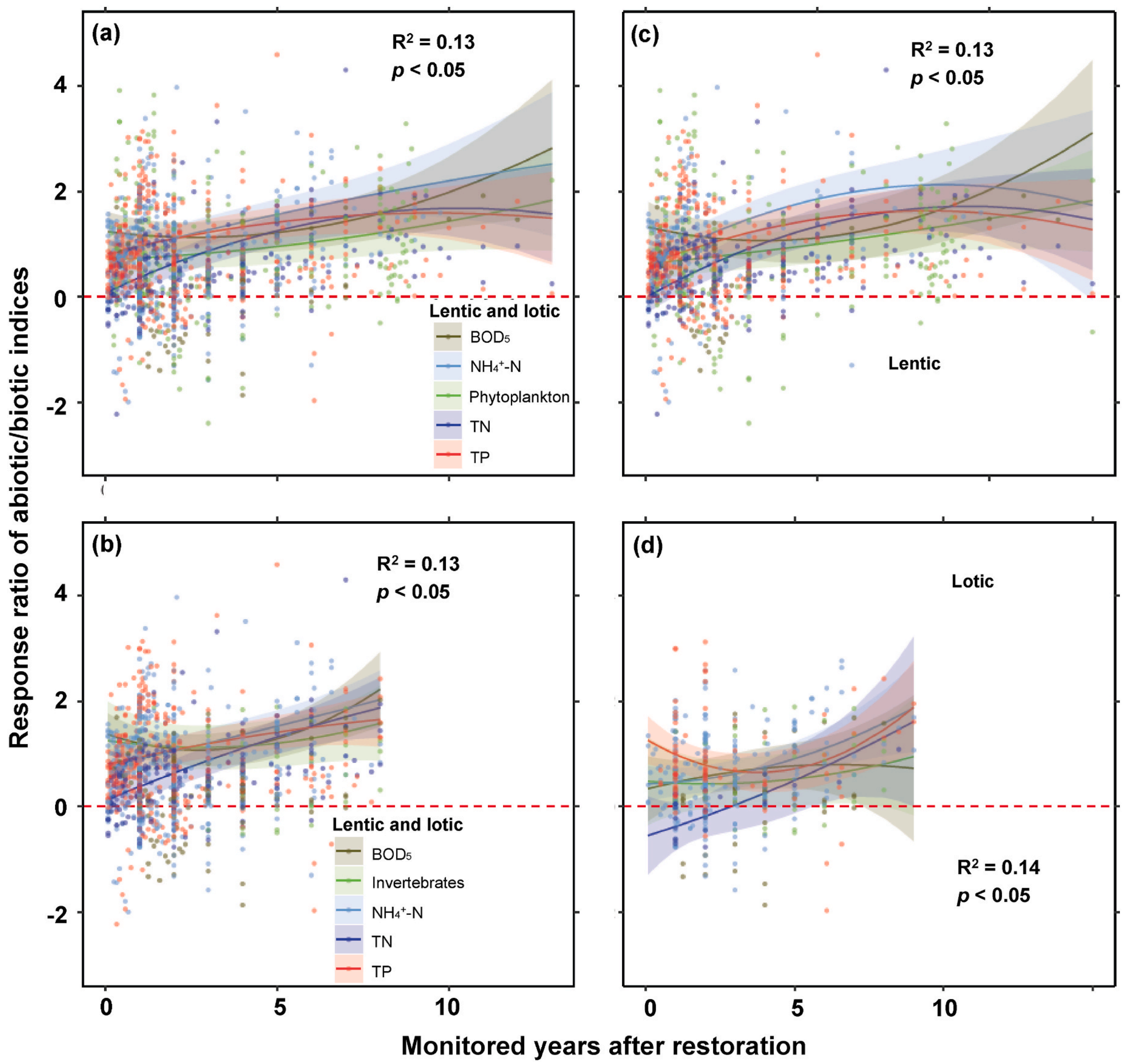

Fig. 7. Marginal effects of the response ratio of individual abiotic and biotic indices in lentic and lotic aquatic ecosystems over the monitored years after restoration for (a) model 4; (b) model 5; (c) model 6; (d) model 7.

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 nitrogen in by >70%, but had less impact on phosphorus 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 ($\leq$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.

Credit author statement

Conceptualization, J.X. and H.Z.; 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.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Data availability

Data will be made available on request.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.envpol.2022.120589.

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Mitigation of urbanization effects on aquatic ecosystems by synchronous ecological restoration

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ABSTRACT

Ecosystem degradation and biodiversity loss have been caused by economic booms in developing countries over recent decades. In response, ecosystem restoration projects have been advanced in some countries but the effectiveness of different approaches and indicators at large spatio-temporal scales (i.e., whole catchments) remains poorly understood. This study assessed the effectiveness of a diverse array of 440 aquatic restoration projects including wastewater treatment, constructed wetlands, plant/algae salvage and dredging of contaminated sediments implemented and maintained from 2007 to 2017 across more than 2000 km^2 of the northwest Taihu basin (Yixing, China). Synchronized investigations of water quality and invertebrate communities were conducted before and after restoration. Our analysis showed that even though there was rapid urbanization at this time, nutrient concentrations (NH_4^+-N, TN, TP) and biological indices of benthic invertebrate (taxonomic richness, Shannon diversity, sensitive taxon density) improved significantly across most of the study area. Improvements were associated with the type of restoration project, with projects targeting pollution-sources leading to the clearest ecosystem responses compared with those remediating pollution sinks. However, in some locations, the recovery of biotic communities appears to lag behind nutrients (e.g., nitrogen and phosphorus), likely reflecting long-distance re-colonization routes for invertebrates given the level of pre-restoration degradation of the catchment. Overall, the study suggests that ecological damage caused by recent rapid economic development in China could potentially be mitigated by massive restoration investments synchronized across whole catchments, although these effects could be expected to be enhanced if urbanization rates were reduced at the same time.

1. Introduction

Almost all natural ecosystems on Earth have been disturbed by human development (Sévêque etal., 2020). Billions of dollars are invested annually to restore degraded ecosystems (Zhang et al., 2000), but many countries continue to face a dilemma between the needs of economic development and ecosystem restoration (Liu etal., 2016b). Therefore, adequate assessment of the efficiency of restoration projects

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in maintaining and restoring natural ecosystem services in line with continuous sustainable development is needed.

There are few developing countries that have implemented as many and diverse ecosystem conservation and restoration projects in recent decades (Zhao et al., 2017), while maintaining rapid economic growth and urbanization, like China. Following the implementation of the reform and opening-up policy, China's urban population increased dramatically from 172.5 million in 1978 to 771.2 million in 2015 (Guan et al., 2018). This urban population growth has resulted in severe degradation of aquatic ecosystems as a consequence of land-use change, pollution and hydromorphological modification (Yang et al., 2019). To mitigate the severe ecosystem degradation, the Chinese government initiated major investments in eco-environmental conservation and restoration projects in 2000. The investment in total environmental restoration across the China mainland has increased from almost nothing in 1994 to 1 trillion RMB Yuan in 2014 (Zhou et al., 2017). Whilst these factors have made China one of the world's leading investors in ecosystem restoration, there is also a general perception that the national restoration policies and actions have contributed a lot to improve the status of water quality across China (Zhou et al., 2017). However, no study has yet attempted to describe the quantitative relationship between the indices of different restoration projects targeting either pollution sources (the place when pollution was generated) or pollution sinks (natural aquatic ecosystems like rivers) and ecosystem indices (nutrients, biological communities, etc.) across large space and time scale.

These investments in river restoration in China provide opportunities to enhance understanding of catchment-scale remediation schemes with varied restoration approaches, which have received comparatively less attention than restoration schemes focused on river sections (Ramchunder et al., 2012), or single types of restoration measures (Kail et al., 2015). To maintain continued, unreserved support from governmental institutions and the general public, the benefits from coordinated, large-scale ecological conservation and restoration efforts urgently need to be evaluated with communication of lessons learned to decision makers.

Here, we combine historical and present data to explore the relationship between a large set of different restoration projects (spanning a range of investments and removal amount of nutrients) and aquatic ecosystem responses in the Taihu basin (Yixing, China). Increasing urbanization intensity can complicate interpretation of aquatic environmental restoration effects over time, although impervious surface area provides a quantifiable index to incorporate this potentially confounding element into the study (Yang et al., 2019). The aim was to examine the effectiveness of ecosystem restoration using nutrients and macroinvertebrates as key indicators, spanning 10 years and across a large spatial-scale (> 2000 km^2); We hypothesized that (Fig. 1): (H_i) ecological damage caused by rapid economic development can be effectively mitigated by synchronous large-scale restoration projects; (H_{ii}) the recovery of biotic indices would lag behind change in abiotic indices (e.g., nutrients) following the implementation of restoration projects because of the extensive pre-restoration degradation of the catchment limiting

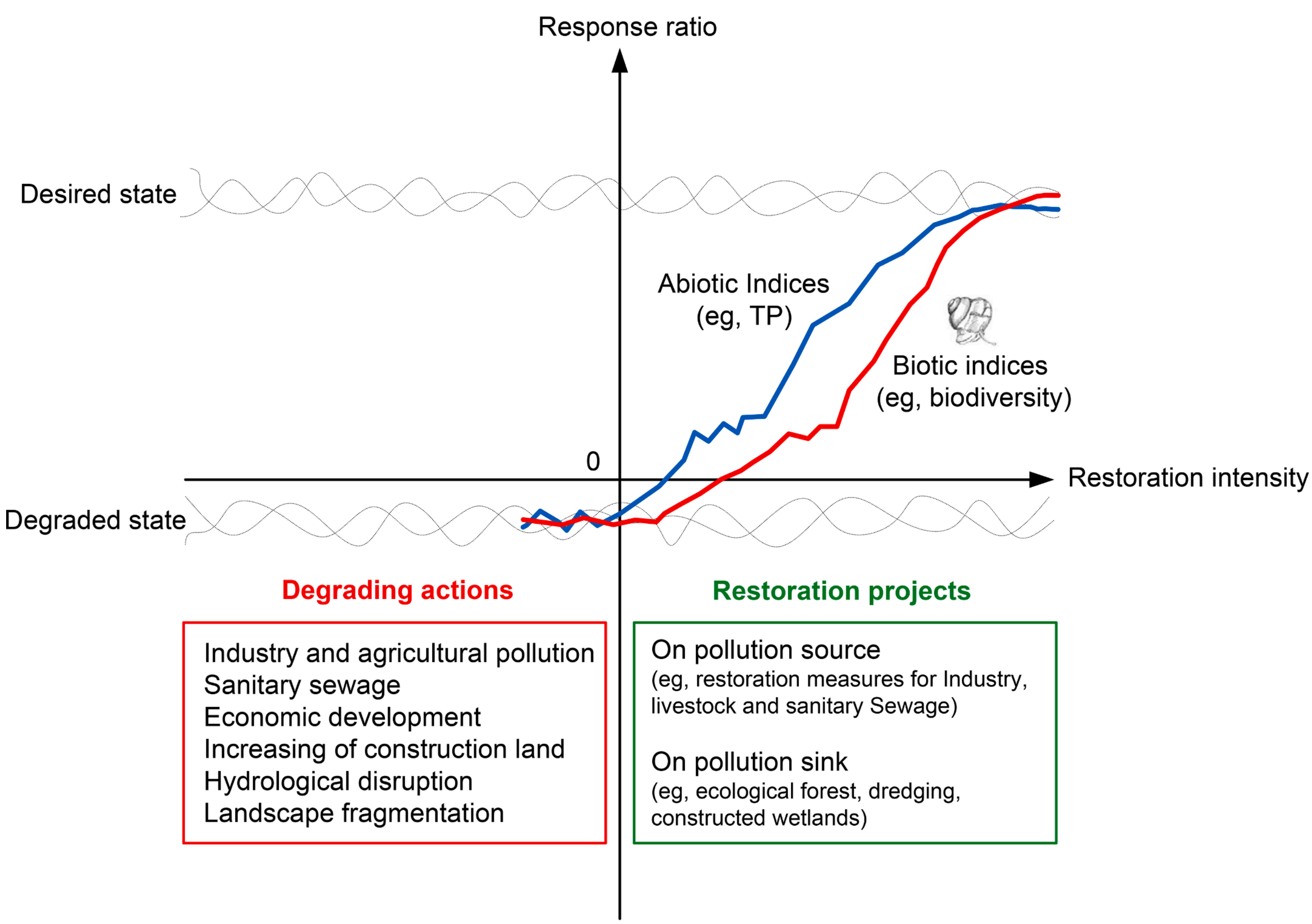

Fig. 1. Conceptual diagram of expected changes in aquatic ecosystems over time as a consequence of restoration. To demonstrate successful restoration, response ratios of abiotic and biotic indices should increase significantly relative to their respective values in the degraded state, ideally reaching predefined target levels corresponding to the desired restoration state.

potential for rapid recolonization; and (H_{iii}) the choice of restoration approach can be expected to result in different effects on the ecosystem restoration with, for example, approaches such as dredging modifying physical habitat and potentially exacerbating stress and delaying recovery.

To test these hypotheses, we gathered information on several hundreds of existing restoration projects conducted in the northwest sector of the Lake Taihu basin (China) over a period of 10 years. Several restoration project indices, nutrient concentration and biological indices of benthic macroinvertebrates in aquatic ecosystems were then computed, and their local trends assessed via a moving-window approach taking into account increases in urbanization intensity and the investment made on the restoration projects. The integrated assessment of multiple data sources provides a novel and thorough analysis on the role of environmental restoration project investments on the water qualities of a watershed under the influence of urban expansion.

2. Material and methods

2.1. Study region

The Taihu basin, located at the plain river network region in the downstream area of the Yangtze River, covers an area of approximately 36,895 km^2 (Fig. 2). The basin, representing 0.4% of China's land area, is heavily populated (40 million residents) and highly industrialized, supporting 11% of China Gross Domestic Product (GDP) (Yi et al., 2017).

This study focused specifically on the upstream areas of the northwest Taihu basin, covering the whole area of Yixing city (Fig. 2). The district covers a total area of 1996.6 km^2 (including 242.29 km^2 of Lake Taihu), 16.8% of which is occupied by water bodies. The catchment has a northern subtropical monsoon climate with an average annual temperature of 16.0 °C and abundant rainfall (1177 mm a year on average,). The urban area of 66.3 km^2 includes rivers with a density of approximately 2.27 km/km^2 (Wang et al., 2017).

Yixing provides an ideal case study to test our hypotheses for two main reasons. First, it represents the typical characteristics of the wider Taihu basin (Pan and Zhao, 2007), and includes nine of the 13 main tributaries to Lake Taihu, which together account for around 60% of the total flow into the lake Taihu. Second, Yixing has spent 8.21 billion RMB ($1.2B USD) on 440 different aquatic environment restoration projects throughout the catchment between 2007 and 2017.

2.2. Restoration project data

We collected data corresponding to restoration projects which were implemented and maintained between 2007 and 2017. The database contained $>$ 440 water environmental restoration projects from the Development and Reform Commission of Jiangsu province (Jiangsu Development and Reform Commission, 2008; Jiangsu Development and Reform Commission, 2014). Of these, one hundred projects that did not provide information on specific restoration measures, the project scale, or for which the measures taken could not be quantified and converted into removal quantity of nitrogen and phosphorus, were discarded (see below for an explanation of how we calculated these parameters). Similarly, projects that did not have direct impacts on the aquatic environment (garbage disposal, drinking water treatment) were removed from further analysis. We collected data on the location of the restoration works (latitude and longitude) (Fig. 2), type of restoration projects (Figure 5), starting and completion year, specific restoration measures, project scale, and total investments (Table S1). To eliminate the effects of inflation on the project investment costs, we used 2007 as the base year and made price adjustments to that baseline for other years' investments (Table S2) (Imai, 2018). Projects were classified according to targeted pollution paths: (i) restoration projects targeting pollution sources (e.g., treatment of industrial and agricultural (farming, aquaculture and livestock breeding etc.) wastewater or sanitary sewage), and (ii) restoration projects targeting pollution sinks (e.g., dredging of contaminated sediment, water hyacinth cultivation for removal of pollutants, harvesting of harmful blue-green algae, etc.). For the restoration projects that aimed to control wastewater pollution at source, we further divided those restoration projects into three different categories: (i) industry-focused, (ii) agricultural wastewater (mainly include livestock breeding and aquaculture in this study), and (iii) domestic sewage.

For each restoration project we calculated the removal quantity (in 10^4 t/a) of key nutrients including ammonia nitrogen (NH_4^+-N), total phosphorus (TP) and total nitrogen (TN), according to different subproject categories and by reference to various national or regional standards of wastewater discharging (for formulas see Table 1 and references therein). The main principle of the removal quantities calculation was to estimate nutrient removal from the sink of the water pollution in theory Table 2.

2.3. Field sampling

To assess relationships between aquatic ecosystems and nutrient removal efficiency, we monitored the recovery of 63 locations by sampling each site both before (2007) and after (2017) the implementation of the restoration works. Sampling sites were located in the limnetic zone of the lakes or the rivers of Yixing and collected between July and September during both sampling campaigns (Fig. 2).

Benthic macroinvertebrate samples were collected within a 100 m reach for each site using a 0.05 m^2 modified Peterson grab (three grabs per reach), and sieved *in situ* through a 250 μm mesh. The resulting sieved materials were stored in a cooler box and transported to the laboratory on the same day. In the laboratory, the samples were sorted on a white tray, and all specimens picked out and preserved in 7% formalin solution. Specimens were identified to the lowest feasible taxonomic level under a dissection microscope (Olympus® SZX10) according to several taxonomic keys (Morse et al., 1994; Wang, 2002).

Simultaneously with benthic macroinvertebrate sampling, four water samples were collected from an intermediate depth at each site, stored in an acid-cleaned plastic container (200 mL), and kept in a cool box for transportation to the laboratory. TN (mg/L), TP (mg/L) and NH_4^+-N (mg/L) were then measured using an ultraviolet spectrophotometer (PhotoLab S12, WTW Company, Munich, Germany). TP and TN were measured on the unfiltered samples, whereas NH_4^+-N was determined from samples filtered using 0.45 μm Whatman GF/F filters (Whatman, Kent, Great Britain). All storage, preservation, and chemical analyses were performed in the laboratory following national standard analytical methods for water and wastewater (National Environmental Protection Bureau, 2002).

2.4. Quantification of restoration effects

Nutrient concentrations: We used the response ratio Δr proposed by Benayas et al. (2009) as a standardized effect size of restoration effects (Eq. (1)). The response ratio is dimensionless with positive values indicating an improvement of the original degraded status and negative values denoting a degradation. Given that decreasing NH_4^+-N, TN and TP concentrations in eutrophic environments are the target of restoration, we reversed the sign of the resulting ration (-Δr) for all assessed nutrient parameters (NH_4^+-N, TN and TP) to make their interpretation more intuitive and keep consistency with that of the biological indices.

$$\Delta r = -\ln(After\ Restoration\,/\,Degraded) \quad (1)$$

Biological indices: By referring to the applications of biological indices in the Yangtze River Basin, China (Huang et al., 2015), taxonomic richness, Shannon–Wiener index (Simpson, 1949) and percentage of Oligochaeta were selected as representative indices to describe the

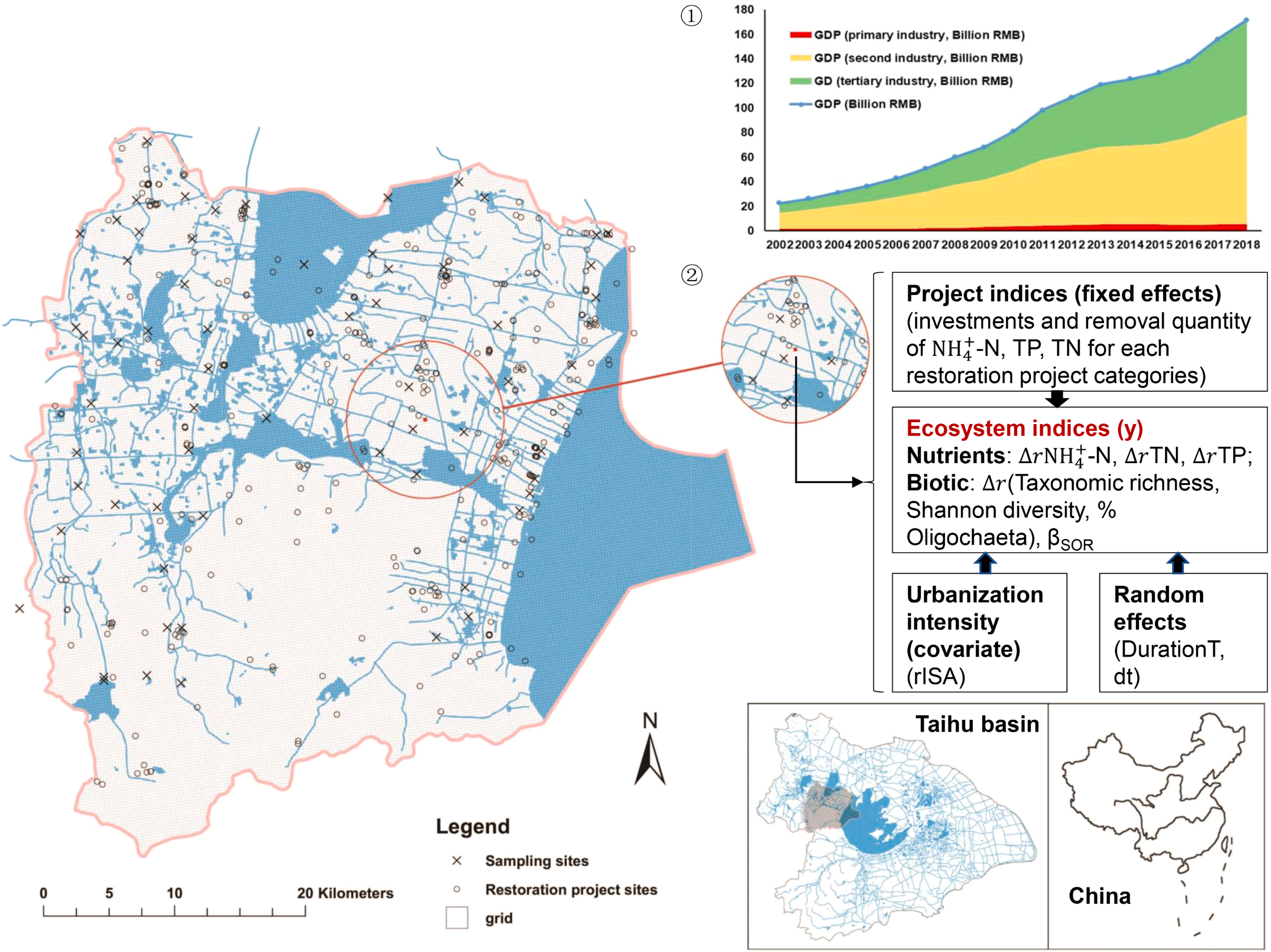

Fig. 2. Map of the study area (Yixing, China) showing the location of the sampling sites, restoration project sites and the spatial definitions considered. Insets refer to ① the changing trend of Gross Domestic Product (GDP) in Yixing from 2002 to 2018), and ② schematic of the calculation process where each squared grid (250 × 250 m) was considered the center of a 6 km radius window containing at least three sampling sites and nine restoration project sites. Indices were then calculated for each of the 4080 windows (see Methods for details on the type of indices and their calculation).

Table 1
Evaluation of removal quantity of nutrients include NH_4^+-N, TN and TP.

Name of sub projects	Class of restoration measures	Evaluation formulas	Units	References
Biogas digester	Source	Vb*swine heads*pollution coefficient* nutrients removal efficiency	10^4 t/a	/
Septic tank	Source	Vs/daily output of livestock sewage*pollution coefficient* nutrients removal efficiency	10^4 t/a	/
Keeping pigs and achieve standard discharge	Source	pig population*pollution coefficient* nutrients removal efficiency	10^4 t/a	(Chinese Research Academy of, 2020)
Sewage of livestock	Source	(mass of livestock sewage/daily output of livestock sewage) * pollution coefficient * nutrients removal efficiency	10^4 t/a	((Xie et al., 2014))
Fermentation bed	Source	area of fermentation bed*breeding density*pollution coefficient* nutrients removal efficiency	10^4 t/a	/
Removal of net cage culture	Source	area of net cage*pollution coefficient	10^4 t/a	(Chinese Research Academy of, 2020)
Renovation of wastewater treatment & water conservation and zero emission projects	Source	treatment scale*(influent of nutrients concentration-effluent nutrients concentration) * 365	10^4 t/a	(Department of Ecology and Environment of Jiangsu Province, 2004; Department of Ecology and Environment of Jiangsu Province, 2007; State Environmental Protection Agency of the People's Republic of China, 2002)
Rural population benefited by sewage treatment facilities	Source	population*pollution coefficient of rural people*365	10^4 t/a	(Wang et al., 2010)
Domestic wastewater treatment	Source	treatment scale*(influent of nutrients concentration-effluent nutrients concentration) * 365	10^4 t/a	(Ministry of Environmental Protection of the People's Republic of China, 2010)
Diversion of urban rain and sewage water	Source	(area of rain and sewage diversion/per capita occupation land)*pollution coefficient of urban people*365	10^4 t/a	(Ministry of Environmental Protection of the People's Republic of China, 2010)
Ecological forest	Sink	area of ecological forest*annual reduction of nutrients	10^4 t/a	(Sun et al., 2015)
Surface flow wetlands	Sink	area of wetlands* nutrients removal efficiency	10^4 t/a	(Li, 2017)
Dredging of contaminated sediment	Sink	Vd*bulk density*(1-moisture content of silt) * release coefficient of nutrients *average amount of nutrients of dry matter	10^4 t/a	(Liu et al., 2016a; Qin et al., 2005; (Zhu et al., 2008))
Cyanobacteria salvage	Sink	dealing rate of cyanobacteria*(moisture content) * nutrients content of dry matter*duration days of cyanobacteria bloom	10^4 t/a	(Zhang et al., 2009; Zhou, 2012)
Water hyacinth planting	Sink	area of water hyacinth* nutrients removal efficiency	10^4 t/a	(Liu et al., 2015; Zhao, 2010)

Notes: Vb, volume of biogas digester; Vs, volume of septic-tank; Vd, volume of dredging.

variation of benthic macroinvertebrate assemblages. A function of species richness and density (Nzengya and Wishitemi, 2000) was used to determine the Shannon diversity.

The Hilsenhoff Family Biotic Index (FBI) (Hilsenhoff, 1988) was applied to assess the ecological conditions of each site. FBI sore are assigned a tolerance number from 0 (very intolerant) to 10 (highly tolerant), and calculated by the following equation: $FBI = \sum^{[}(TV_i)(n_i)]/N$, where TV_i is the tolerance value of the ith taxon, n_i is the number of individuals in ith taxon, and N is the total number of individuals in the sample. The tolerance value of each family was obtained from Qin et al. (2014) and Wang and Yang (2004). Low FBI values reflect a higher abundance of sensitive invertebrate groups, thus a lower level of organic pollution.

We analyzed the changes in species composition between restored (2017) and degraded (2007) sites using the command beta.temp in the *R* package betapart (Baselga and Orme, 2012). This procedure computes the total dissimilarity (measured as Sørensen dissimilarity, β_{SOR}), and partitions it into turnover (β_{SIM}) and nestedness (β_{SNE}) components (Baselga, 2012). In the context of temporal variation of communities these two components reflect (i) the substitution of some species by others through time (β_{SIM}), and (ii) the loss (or gain) of species through time in a nested pattern (β_{SNE}).

Biological response ratios were based on a slightly modified formula:

$$\Delta r = \ln[(After\ Restoration + 1) / (Degraded + 1)] \tag{2}$$

where, in this case, the degraded and restored conditions were calculated using the biological indices of benthic macroinvertebrate (taxonomic richness, Shannon diversity, percent Oligochaeta and Hilsenhoff FBI). The addition of a unit (+1) to each term in the formula was needed because some sites it registered zero values.

2.5. Land use data and urbanization metric

Land use data for Yixing district was derived from 30-m resolution land use maps for 2007 and 2017 (taken as surrogates for existing conditions before and after implementation of restoration) provided by the Resource and Environmental Science Data Center of the Chinese Academy of Sciences (http://www.resdc.cn) (Fig. S1). The 26 original land use categories were simplified into six categories according to the land resource classification system of China's land use/land cove change (CNLUCC), namely farm land, building land (artificial surfaces), forest land, grassland, water body and barren land (Song and Deng, 2017). The land use transformation matrix for the Yixing district across the six land use categories between 2007 and 2017 is provided in Table S3.

The impervious surface area (ISA) of Yixing has increased from 4.36% in 2007 to 10.15% in 2017. Prior research has noted that when the ISA increases to a range between 10 and 25%, the impact on aquatic environments is significant (Schueler, 1994). However, the water environment in relation to the ISA may vary depending on regional conditions (Luo et al., 2018). Thus, we used the response ratio of impervious surface area (rISA = ln (ISA_{2017}/ISA_{2007})) as a co-variable in subsequent analyzes to assess confounding effects of land use change (urbanization) acting in opposition to restoration effects. Land use data and the

Table 2

Results of GLMM and LMM for nutrients (NH_4^+-N, TN, TP), the investments of different restoration project categories and the intensity of urbanization (expressed as the response ratio of impervious surface area (rISA)) on biological parameters (taxa richness, Shannon diversity,% Oligochaeta, β_{SOR}). Variables are only given when the correlation was significant ($p < 0.05$). Variables shaded in gray correspond to positive correlations. s_Livst_inv, investment targeting agricultural sewage; s_san_inv, investment targeting sanitary sewage; s_ind_inv, investment targeting industry waste water; sinkPinvstm, investment targeting pollution sink.

taxa richness GLMM (gaussian, link="log"), $N = 3022$, Marginal R^2: 0.73				
Variables	Estimates	SE	t	P
(Intercept)	1.44	0.13	10.79	<0.001
log(**NH_4^+**-N)	0.27	0.02	14.00	<0.001
TN	−0.36	0.02	−14.62	<0.001
TP	−0.67	0.02	−42.42	<0.001
s_Agric_inv	−0.28	0.04	−6.93	<0.001
s_san_inv	0.09	0.02	4.62	<0.001
s_ind_inv	−0.12	0.03	−3.63	<0.001
sinkPinvstm	−0.47	0.01	−43.78	<0.001
s_Agric_inv: rISA	0.74	0.08	9.56	<0.001
s_san_inv:rISA	−0.16	0.03	−4.70	<0.001
s_ind_inv:rISA	−0.39	0.07	−5.63	<0.001
Shannon diversity LMM, $N = 3022$, Marginal R^2: 0.63				
Variables	Estimates	SE	t	P
(Intercept)	1.76	0.07	23.66	<0.001
log(**NH_4^+**-N)	0.51	0.01	39.10	<0.001
TN	−0.09	0.02	−4.65	<0.001
TP	−0.56	0.01	−44.70	<0.001
s_Agric_inv	−0.25	0.003	−8.13	<0.001
s_san_inv	0.04	0.01	4.65	<0.001
s_ind_inv	−0.32	0.01	−25.77	<0.001
sinkPinvstm	−0.18	0.003	−21.88	<0.001
rISA:s_Livst_inv	0.41	0.07	5.98	<0.001
(-% Oligochaeta) LMM, $N = 3022$, Marginal R^2: 0.47				
Variables	Estimates	SE	t	P
(Intercept)	0.46	0.06	1.82	<0.001
log(**NH_4^+**-N)	−0.03	0.01	−4.33	<0.001
TN	−0.10	0.01	−8.19	<0.001
TP	−0.03	0.01	−3.90	<0.001
s_Agric_inv	0.05	0.02	2.72	<0.01
s_san_inv	−0.11	0.01	−14.65	<0.001
s_ind_inv	0.28	0.01	18.14	<0.001
sinkPinvstm	–0.26	0.01	−27.11	<0.001
s_Agric_inv:rISA	−0.22	0.04	−5.01	<0.001
s_san_inv:rISA	0.18	0.02	8.00	<0.001
s_ind_inv:rISA	−0.53	0.04	−14.94	<0.001
sinkPinvstm:rISA	0.34	0.02	10.36	<0.001
β_{SOR} LMM, $N = 3022$, Marginal R^2: 0.31				
Variables	Estimates	SE	t	P
(Intercept)	0.62	0.03	18.60	<0.001
log(**NH_4^+**-N)	−0.07	0.03	−14.21	<0.001
TP	−0.04	0.01	−8.96	<0.001
s_Agric_inv	0.09	0.01	7.44	<0.001
s_ind_inv	0.03	0.01	2.84	<0.001
s_san_inv	0.04	0.004	8.70	<0.001
sinkPinvstm	−0.06	0.003	−17.81	<0.001
s_Agric_inv:rISA	−0.18	0.003	−6.75	<0.001
s_san_inv:rISA	−0.08	0.01	−9.58	<0.001
s_ind_inv:rISA	−0.21	0.02	−9.56	<0.001

impervious surface area were handled and calculated using ArcGIS 10.2 (ESRI Company, Redlands, CA, USA) and Fragstats 4.2 (McGarigal et al., 2012).

2.6. Data analysis

2.6.1. Assessing spatial distribution of project indices, ecosystem indices and the response ratio of impervious surface area

Because of the well-developed floodplain river network of Yixing district, Taihu Basin, the landform is flat, water flows slowly, and flow direction is often variable because of the influence of artificial drainage (Deng et al., 2015). Thus, we adopted a moving window approach to estimate all parameters (project, ecosystem and urbanization intensity indices) on a spatial continuum covering the whole study area. This approach is useful for summarizing local spatial trends emerging from regional dynamics (Gaüzère et al., 2016). The principle lies in calculating the metrics of interest for each cell of a squared grid (250 × 250 m, slightly less than the distance between the two nearest sampling sites to generate more windows), covering the study area, using a circular moving window centered on the centroid of each cell. In this way, the values of the different metrics attributed to each grid cell represent summaries of the neighboring restoration project sites, sampling sites and the response ratio of impervious surface area (Fig. 2).

We used a 6 km radius for the circular window (Figs. 2 and S2). The chosen window radius resulted from a compromise between incorporating the range of restoration projects and enough spatial repetition to estimate reliable linear trends in variables, and achieving an adequate coverage of the study area. This generated 4080 spatial windows, each containing at least three sampling sites and nine restoration project sites. Finally, indices (project, nutrients and biological indices) were calculated for the 4080 spatial windows based on the mean of ecosystem indices or the sum of project indices, and the response ratio of impervious surface area was then calculated for each window. This moving window approach enabled the local spatial trends of each restoration project index to be compared with the local spatial trends of aquatic ecosystem indices (Gauzere et al., 2017).

2.6.2. Statistical analysis for all indices

Visual inspection of frequency histograms showed all response ratios of ecosystem indices (ΔrNH_4^+-N, ΔrTN, ΔrTP, taxonomic richness, Shannon diversity,% Oligochaeta and Hilsenhoff FBI) followed non-normal distributions (Fig. 4). Therefore, we used Wilcoxon signed rank tests to examine whether median response ratios of ecosystem indices were significantly different from zero. Non-metric multidimensional scaling ordination (NMDS) was used to visualize invertebrate communities by site and restoration phase (before/after). Taxon density data were ordinated using Bray–Curtis similarity as the distance measure for the scaling with square-root transformation to reduce impacts of extremely high counts of individual taxa. Similarity percentage (SIMPER) analysis was used to identify which taxa contributed the most to the average Bray-Curtis dissimilarity between the two-restoration phases.

Spearman Rank correlation was used to test for significant correlations between project investment and removal quantity of NH_4^+-N, TP, TN by project category. We also used Kruskal-Wallis tests to examine whether investments differed among different restoration project categories. Finally, the relationships between restoration projects and ecosystem recovery were assessed by fitting a generalized linear mixed model (GLMM) with a Gamma distribution (log link) or Linear Mixed Model (LMM) to each nutrient (ΔrNH_4^+-N, ΔrTN, ΔrTP). Restoration project investment by category and the response ratio of impervious surface area (rISA) were added as fixed effects, while the number of years since the implementation of the restoration (DurationT) and the time since completion of the restoration (dt = 2017 - end year of the restoration) were used as random effects. GLMM with Gaussian distribution (log link) or LMM were applied to the biological indices (Δr taxonomic richness, Δr Shannon diversity, Δr% Oligochaeta, βSOR) with nutrients and investment of different restoration categories as fixed effects, rISA as covariate, Duration *T* and dt as random effects. Removal quantity of nutrient was subsequently omitted from these models because of its significant positive correlation with project investment (see Results section). To explore the interaction effect between urbanization intensity and the strength of restoration, the interaction term 'rISA*investment of different project categories' was included in the models described above.

Prior to analysis, the investment of each restoration project category was $\log_{10}$ transformed to constrain the influence of extreme values. We compared the complex model with a null model; models were simplified by removing non-significant terms and verifying the distribution through residuals analysis ((Crawley, 2002)). Akaike's Information Criterion (AIC) values were used to determine the most parsimonious fit. Model residuals were tested for spatial autocorrelation with Moran's

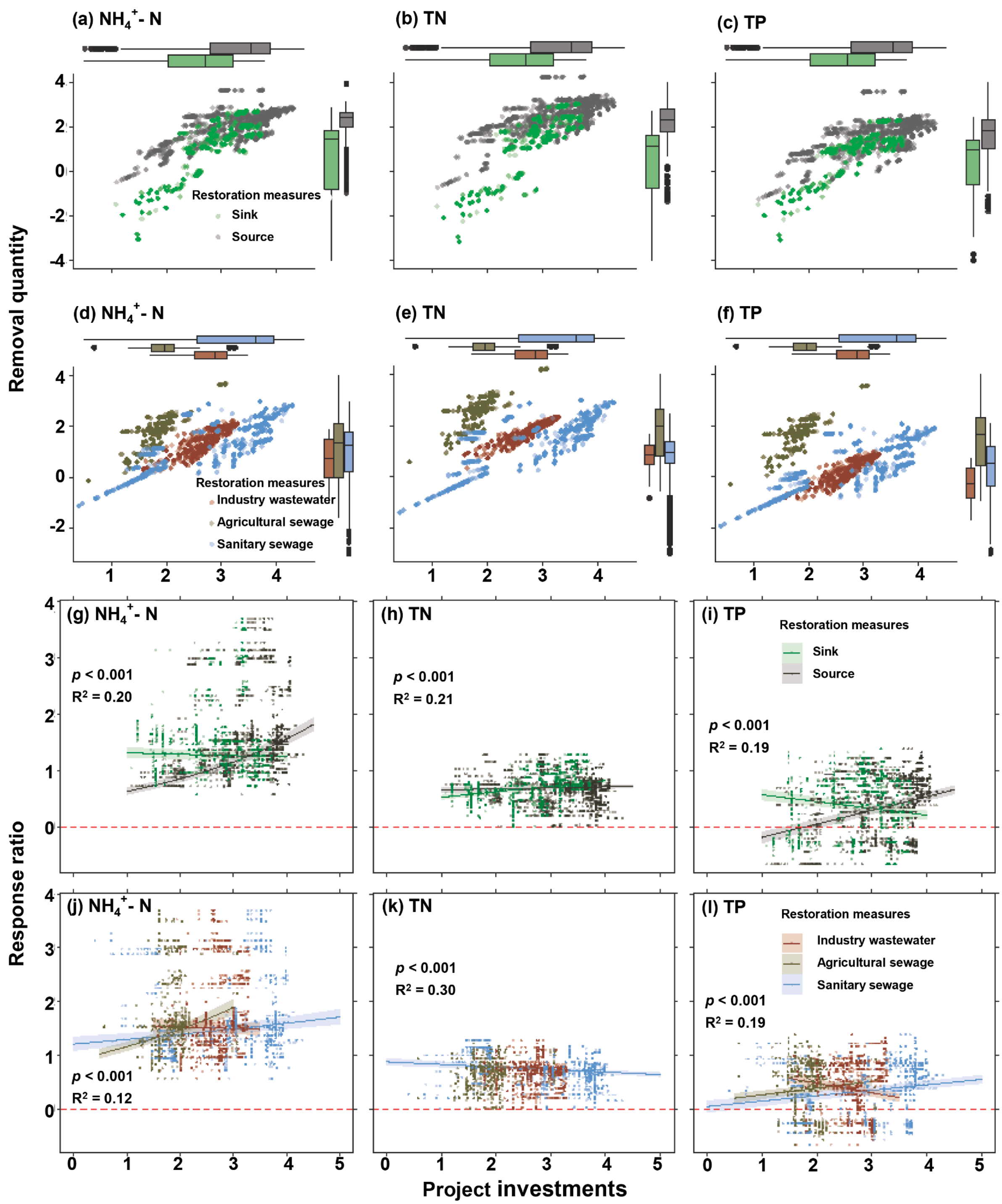

Fig. 3. Scatter plots showing the relationships between project investments of different restoration project categories and either of (a–f) removal quantities for the different nutrients (ΔrNH_4^+-N, ΔrTN, ΔrTP) (the marginal boxplots in a-f show the investment distribution on different restorations) and response ratio (g–l) of different restoration project categories, (a–c) restoration measures for pollution source (Spearman rank Rs = 0.62, 0.58, 0.55, $p < 0.001$) and pollution sink (Rs = 0.79, 0.85, 0.82, $p < 0.001$); (d–f) three main categories of restoration measures for pollution source, which include restoration measures for industry waste water (Rs = 0.92, 0.95, 0.94, $p < 0.001$), agricultural (Rs = 0.89, 0.89, 0.89, $p < 0.001$) and sanitary (Rs = 0.70, 0.68, 0.69, $p < 0.001$) sewage. Marginal effects of investment of different restoration project categories on each nutrient (ΔrNH_4^+-N, ΔrTN and ΔrTP): (g–i) restoration measures for pollution source and sink; (j–l) three main categories of restoration measures for pollution source, which include restoration measures for industry waste water, agricultural and sanitary sewage. GLMM or LMM regression lines are given where a correlation was significant ($p < 0.05$). The initial unit of investment is 10^5 RMB, and the initial unit of the removal quantity of nutrients is 10^4 t/a, both were $\log_{10}$ transformed before inclusion in models.

tests (Birk et al., 2020), which showed in all instances no autocorrelation.

All data analysis was performed in using R v 4.0.1 (R Core Team 2020, https://www.R-project.org/) using the packages: lme4 and lmerTest.

3. Results

3.1. Relationship between restoration project investments and nutrient removal

Spearman rank (*Rs*) correlations analysis showed a significant positive correlation between project investment and the removal quantity of nutrients (calculated as described in Table 1) across project categories (Fig. 3, Table S4). The amount of money invested by the government varied significantly with project category (Kruskal Wallis test, $p < 0.001$). The projects attracting larger investments were, in decreasing order of magnitude: pollution source, pollution sink, sanitary sewage, industrial wastewater, agricultural sewage (Fig. 3).

3.2. Efficiency of restoration projects on nutrients and biological status

Restoration works were found to be efficient at recovering aquatic ecosystems from their initial degraded condition as shown by their significant effect on almost all assessed ecosystem indices. The concentration of NH_4^+-N, TN and TP across the whole Yixing river network was significantly lower in restored (2017) than in degraded (2007) aquatic ecosystems, leading to overall positive response ratios (Fig. 4); Taxonomic richness and Shannon diversity of benthic macroinvertebrate were significantly higher in restored (2017) than in degraded (2007) sites (mean response ratio = 1.085, 0.415, $P < 0.001$, Fig. 4). Percent Oligochaeta was significantly lower in restored (2017, 17.53% ± 16.65%) than in degraded (2007, 40.78% ± 39.70%) sites. Hilsenhoff FBI of benthic macroinvertebrate communities showed no significant difference between degraded (2007) and restored (2017) ecosystems Fig. 5.

The composition of benthic macroinvertebrate communities differed significantly between degraded (2007) and restored (2017) periods (PERMANOVA, $p < 0.01$; final stress = 0.128, Fig. 6). SIMPER analysis identified eight species cumulatively contributing > 70% to the dissimilarity between restored (2017) and degraded (2007) invertebrate communities (Table S5). They were *Limnodrilus hoffmeisteri, Bellamya aeruginosa, Corbicula fluminea, Branchiura sowerbyi, Parafossarulus eximius, Neocaridina denticulata, Exopalaemon modestus* and *Parafossarulus striatulus* in decreasing order. Some sensitive species to anthropogenic pressures with low tolerance values recolonized after the restoration (2017). For example, river flies *Heptagenia* sp.*, Ephemera orientalis and Ceratopsyche* sp. The partitioning of the Sørensen dissimilarity index was dominated by species turnover (β_{SIM}) (mean = 0.44, SD = 0.36), implying that, in any given site, an average of 44% of the species were unique to the time (either 2007 or 2017 site assemblage). In contrast, the nestedness component (β_{SNE}) was much lower (mean = 0.34, SD = 0.33), implying that weaker patterns of species losses or gains from pre-existing communities have occurred between 2007 and 2017 (Fig. 4). The spatial distribution of total (βSOR) and nested (βSIM) dissimilarity can be seen in Figure 5.

3.3. Effects of restoration projects on aquatic ecosystem status

Examination of the marginal effect of project investment amount by category on nutrients (Fig. 3, Table 2) showed a significant correlation of decreasing river network NH_4^+-N concentrations (i.e., increasing response ratios) with increasing investment on projects targeting pollution sources but not those targeting pollution sinks (marginal = 0.20, $p < 0.001$). On the contrary, decreasing TN and TP concentrations were positively correlated with increasing investment on restoration projects targeting both pollution sources and sinks (marginal $R^2 = 0.23$, $p < 0.001$; marginal $R^2 = 0.19$, $p < 0.001$). Decreasing NH_4^+-N and TP concentrations correlated with increasing investment in both restoration projects targeting agricultural and domestic sewage, but not those targeting industry wastewater (marginal $R^2 = 0.14$, $p < 0.001$; marginal $R^2 = 0.19$, $p < 0.001$). Decreasing TN concentrations were negatively correlated with the increasing investment on restoration projects targeting sanitary sewage (marginal $R^2 = 0.32$, $p < 0.001$). A significant interaction was evident between the response ratio of impervious surface area and investments of different restoration project categories and nutrient responses. For example, poor nutrient responses were associated with the growth of impervious surface area ($p < 0.001$, Fig. 7), but these effects were overcome where restoration projects were large but impervious area increased minimally.

For the biological indices, increased Shannon diversity and taxonomic richness over time showed significant inverse relationships with NH_4^+-N concentrations, and a positive association with increasing investment on restoration projects targeting sanitary sewage (Shannon

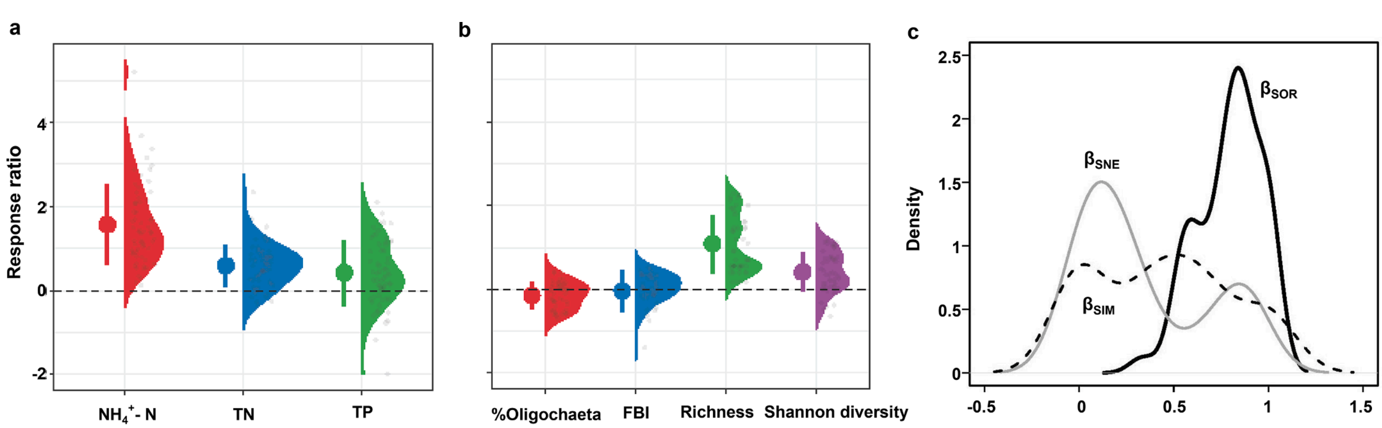

Fig. 4. Response ratios of NH_4^+-N, TN, TP and taxonomic richness (Richness), Shannon diversity, percent Oligochaeta, FBI of benthic macroinvertebrate in restored (2017) compared with degraded ecosystems (2007) (a,b). All response ratios differed significantly from zero (Wilcoxon signed rank tests, $p < 0.001$) except for Hilsenhoff FBI. The mean and SD are given alongside the overall data distribution for each metric. (c) The partition of temporal total dissimilarity (β_{SOR}-solid black line) into nested resultant dissimilarity (β_{SNE} -solid gray line) and turnover (β_{SIM}-dashed lines) for beta diversity of benthic macroinvertebrates in Yixing from 2007 to 2017.

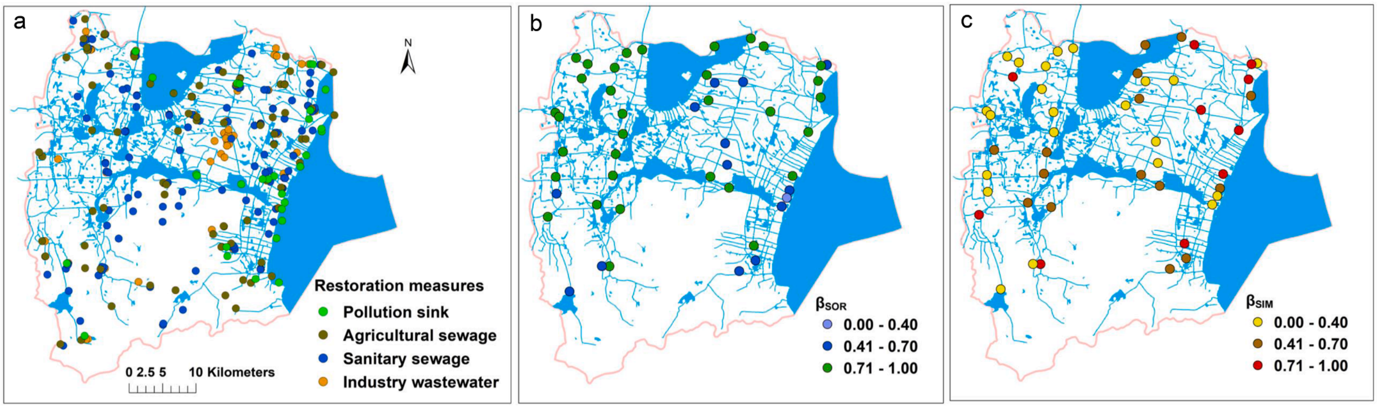

Fig. 5. (a) Location of different restoration project sites by category ($n = 420$, projects of garbage disposal and drinking water treatment were not included in the analysis) in Yixing from 2007 to 2017. (b,c) Maps showing the spatial distribution of total (β_{SOR}) and nested (β_{SIM}) dissimilarity for beta diversity of benthic macroinvertebrates.

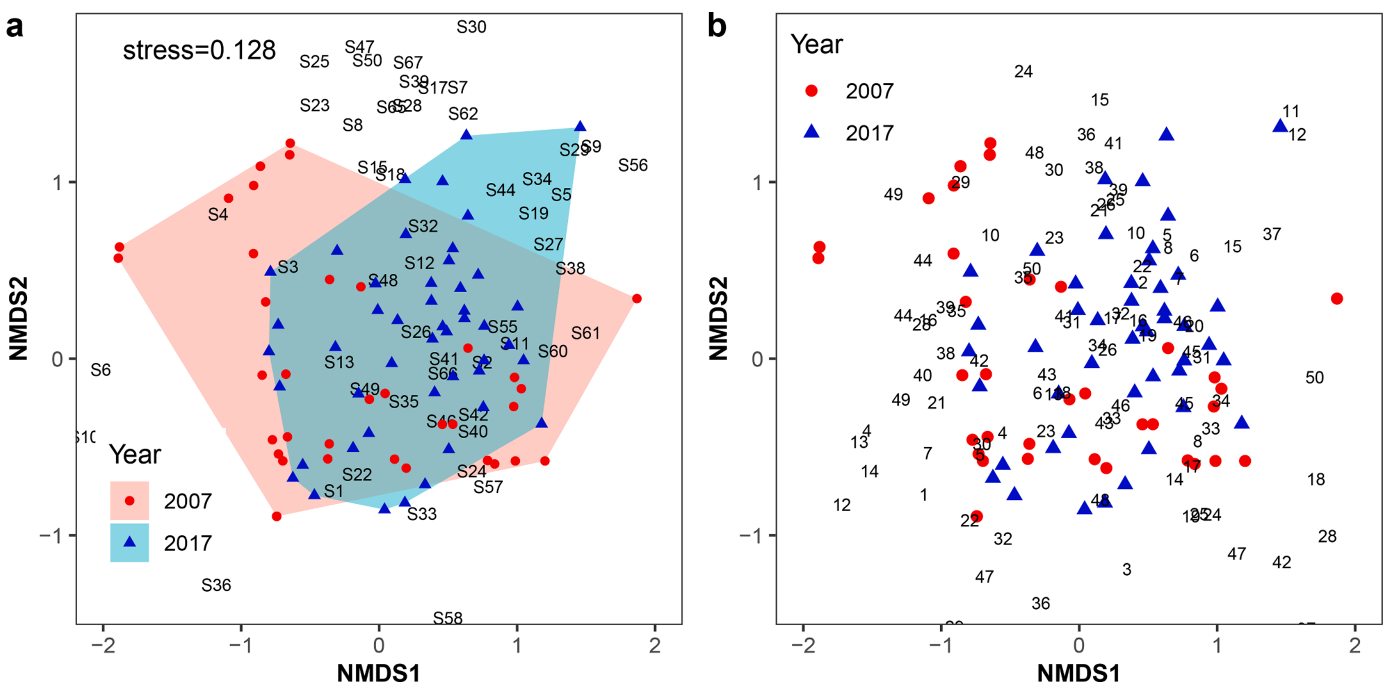

Fig. 6. NMDS biplots showing changes in community composition of benthic invertebrates among restoration projects between their initial degraded (2007) and final restored (2017) states in the Yixing river network with indication of (a) the individual taxa (denoted by S) and (b) sampling sites (denoted by the numbers). An outlier was removed from this figure because it had only one scare species in 2007 and had 16 species in 2017. S1, *Limnodrilus hoffmeisteri*; S2, *Bellamya aeruginosa*; S3, *Branchiura sowerbyi*; S4, *Corbicula fluminea*; S5, *Parafossarulus eximius*; S6, *Nephtys oligobranchia*; S7, *Parafossarulus striatulus*; S8, *Neocaridina denticulata*; S9, *Semisulcospira cancelata*; S10, *Gammarus* sp.; S11, *Exopalaemon modestus*; S12, *Alocinma longicornis*; S13, *Branchiodrilus hortensis*; S14, *Cricotopus bicinctus*; S15, *Limnoperna fortunei*; S16, *Procladius* sp.; S17, *Physa* sp.; S18, *Tanypus chinensis*; S19, *Radix swinhoei*; S20, *Ceratopsyche* sp.; S21, *Heptagenia* sp.; S22, *Chironomus plumosus*; S23, *Ploypedilum scalaenum*; S24, *Propsilocerus akamusi*; S25, *Acuticosta chinensis*; S26, *Anodonta woodiana pacifica*; S27, *Anodonta woodiana elliptica*; S28, *Glossiphonia* sp.; S29, *Dicrotendipus lobifer*; S30, *Semisulcospira libertina*; S31, *Stenothyra glabra*; S32, *Unio douglasiae*; S33, *Glossiphonia complanata*; S34, *Glyptotendipes tokunagai*; S35, *Nemertea* sp.; S36, *Microchironomus tabarui*; S37, *Aulodrilus* sp.; S38, *Nereis japonica;* S39, *Tabanus* sp.; S40, *Macrobrachium nipponense*; S41, *Stictochironomus* sp.; S42, *Cricotopus sylvestris*; S43, *Tanytarsus chinyensis*; S44, *Laccophilus* sp.; S45, *Rhaphium* sp.; S46, *Glyptotendipes pallens*; S47, *Lamelligomphus* sp.; S48, *Helobdella fusca*; S49, *Glossiphonia lata*; S50, *Aciagrion* sp.; S51, *Baetis* sp.; S52, *Harnischia fuscimana*; S53, *Ephemera orientalis*; S54, *Cricotopus vierriensis*; S55, *Erpobdella octoculata*; S56, *Hippeutis cantori*; S57, *Glyptotendipes* sp.; S58, *Polypedilum nubeculosum*; S59, *Cricotopus trifascia Edwards*; S60, *Orientogomphus* sp.; S61, *Burmagomphus* sp.; S62, *Tachaea chinensis*; S63, *Lestidae* sp.; S64, *Cryptochironomus* sp.; S65, *Cercion* sp.; S66, *Calopteryx* sp.; S67, *Holorusia* sp.; S68, *Brachythemis* sp.

marginal $R^2 = 0.63, p < 0.001$; richness marginal $R^2 = 0.73, p < 0.001$). Decreasing Oligochaeta relative abundance was associated with investment value of restoration projects both targeting agricultural and industrial wastewater (marginal $R^2 = 0.47, p < 0.001$). Increasing β_{SOR} was correlated positively with investment for all three project categories targeting pollution sources (marginal $R^2 = 0.32, p < 0.001$). There was evidence for significant interaction between the response ratio of impervious surface area and investments of different restoration project categories and biological index responses. For example, poor biological responses were associated with the growth of the impervious surface area ($p < 0.001$, Fig. 7), but these effects were overcome where restoration projects were large but impervious area increased minimally.

4. Discussion

This study has provided new insights to understand the effectiveness

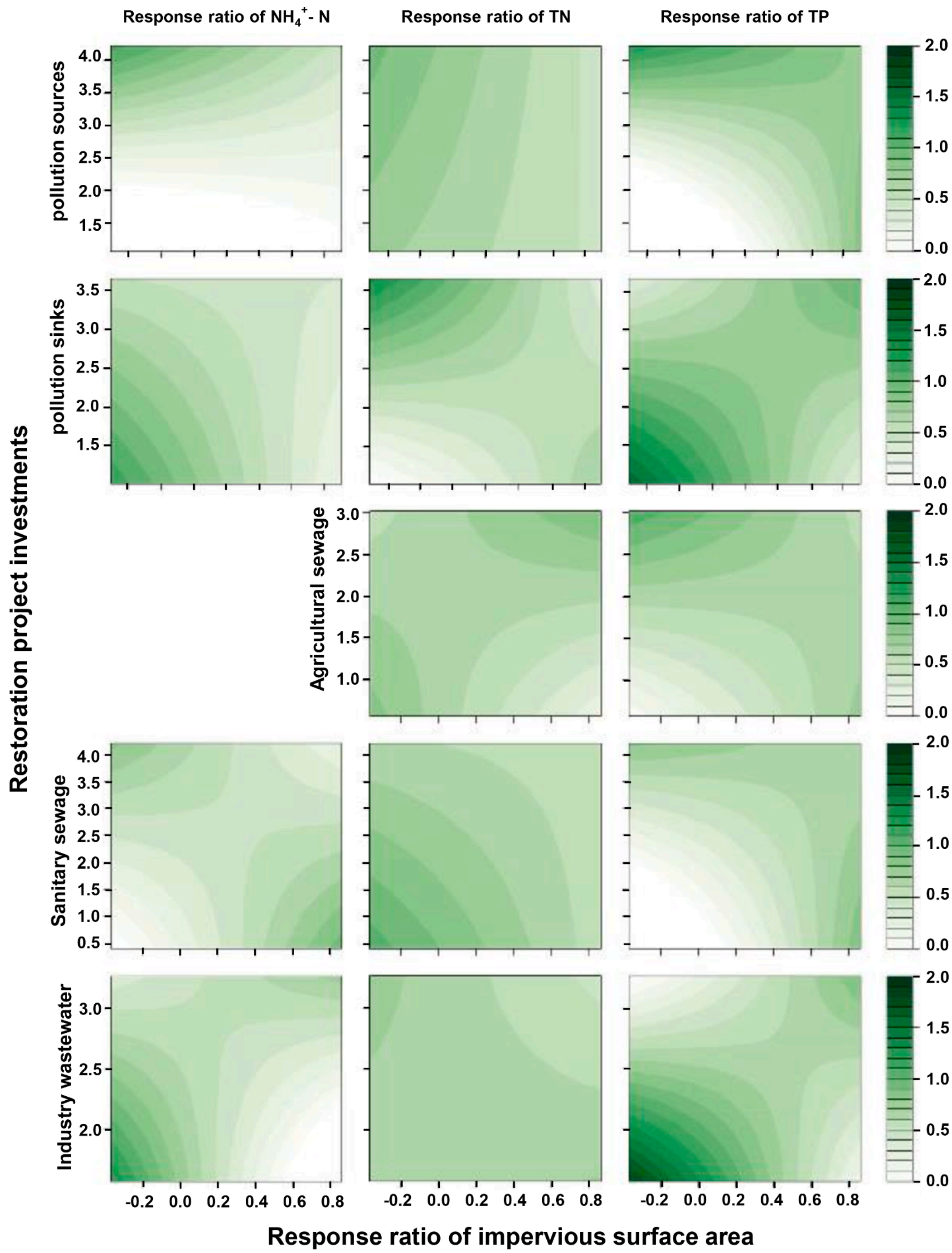

Fig. 7. Interaction effect between the response ratio of impervious surface area (rISA) and investments of different restoration project categories on the response ratio of NH_4^+-N, TN, TP in natural waterbodies in Yixing. Figures are given when the interaction effect was significant ($p < 0.05$).

of catchment-scale restoration towards increasing water quality and biodiversity in rivers of China, building on knowledge from previous studies from a variety of ecosystems in other parts of the world (Benayas et al., 2009; Crouzeilles et al., 2016). Using a large data set comprising hundreds of different aquatic ecosystem restoration projects undertaken over the last two decades in a large urban district of China, we showed that implementation of large-scale restoration projects can, to some extent, mitigate the environmental degradation as a result of economic boom. In Yixing, recovery occurred despite ongoing rapid economic growth and urbanization, although it should be noted that the impervious surface area reached only 10.15% at the bottom end of Schueler (1994) 10–25% range for significant impacts on water quality. Further urbanization may therefore negate the positive aspects of restoration observed to date.

Restoration led to decreases in indicators of stress, notably concentrations of main nutrients (NH_4^+-N, TN and TP) and Oligochaeta relative abundance, whereas taxonomic richness and Shannon-Weiner diversity of benthic macroinvertebrate were significantly higher across the Yixing river network. These general findings for macroinvertebrate community and water quality recovery are supported by studies which found a significant positive effect of restoration on the organism groups and water quality (Kong et al., 2020). External inputs of organic pollution from sewage have been reduced from the catchment, and measures such as sediment dredging, cyanobacteria salvage, etc. have been implemented to reduce the internal nutrient loading. This combination of approaches has allowed dissolved oxygen concentrations to rise, gradually improving aquatic habitat and enhancing aquatic biodiversity (Mason, 2002).

In contrast, the overall Hilsenhoff FBI showed no significant difference between degraded (2007) and restored (2017) years. Despite the enormous investment in restoration, there were 25 sites showing increases in Hilsenhoff FBI scores, due to some higher tolerance taxa still remaining, and taxonomic richness in 2007 being much lower (average: 2.16) than in 2017 (average: 8.58). During the 10 years of the study, Yixing has seen its GDP increase from 42.80 billion RMB in 2006 to 155.83 billion RMB in 2017 (Fig. 2), accompanied by 45% growth of artificial surfaces (Fig. S1 and Table S3). The effects of urbanization (hydromorphology and hydrological alteration, run off pollution) are likely to have suppressed the level of biotic recovery of freshwater macroinvertebrates that may have occurred from restoration efforts in isolation by increasing the role of other stressors (Gál et al., 2019). Despite this urban cover growth, water quality of Chinese inland waters has clearly improved generally over recent decades with restoration efforts (Zhou et al., 2017).

On the other hand, we found that the response of biotic indices to restoration projects appeared to lag behind nutrients (NH_4^+-N, TN and TP), with the standardized responses of nutrients being greater than those of biotic indices. However, compared to the degraded time-period (2007), some species that are sensitive to anthropogenic pressures (low tolerance values) recolonized after restoration (2017) including the river flies *Heptagenia* sp.and *Ephemera orientalis*. These observed increases in the *Heptageniidae* are in line with Pedersen et al. (2007) who reported they increased significantly in abundance after a short-term restoration (three years) at the Skjern River reaches, Denmark. The composition of benthic invertebrate communities differed significantly between degraded (2007) and restored (2017) periods. Eight species cumulatively contributed $>$ 70% to the dissimilarity between restored (2017) and degraded (2007) invertebrate communities (Table S5). *Limnodrilus hoffmeisteri* (turbid worms) and *Branchiura sowerbyi* (crustaceans) both decreased more in restored than in degraded rivers. These species are widely used as an indicator of organic pollution throughout China (Gorni et al., 2018), thus their decreasing abundance provides important ecological evidence for restoration success alongside the water quality improvements. However, *Limnodrilus hoffmeisteri* was still a co-dominant species in some sampling sites in Yixing river network both in degraded and restored time-periods; something not surprising as they are widely distributed throughout global freshwater ecosystems (Armendáriz and César, 2001). In contrast, snails and clams such as *Bellamya aeruginosa* and *Corbicula fluminea* increased more in restored than in degraded. Recovery of these native snail and bivalve populations can be expected to further help improve the water quality given their roles as deposit or filter feeders that remove particulates (Zhang et al., 2014). The relative abundance of snails like *Bellamya aeruginosa* increased in $>$ 20 sites over time, most likely because some native snails have been reintroduced by restoration activities in attempts to enhance algal removal. While *Bellamya aeruginosa* and *Corbicula fluminea* are common species which are widely distributed in eutrophic shallow lakes in China (Zhu et al., 2013). Although biological indices appear to lag behind abiotic indices like nutrients, sampling frequency limited our ability to elucidate more clearly the relationship between these indicators.

Even though the response ratio of taxonomic richness and Shannon diversity of benthic macroinvertebrate was significantly higher in restored (2017) than in degraded (2007) aquatic ecosystems, taxonomic richness and Shannon diversity of benthic invertebrate only showed significant positive correlation with the increasing project investment on sanitary sewage removal, the decline of NH_4^+-N concentrations in Yixing river network, and the interaction effect between the response ratio of impervious surface area and project investment on agricultural sewage removal. The muted improvements of biological indices may be due to two reasons: (i) restoration measures on pollution sink mainly include dredging of contaminated sediments, which will negatively affect the habitat of benthic macroinvertebrates; (ii) water quality in Yixing is improved but still not to a high level, and hydromorphological alterations remain throughout the catchment, limiting recovery potential. Additionally, only 14.4% of investments were targeted at pollution sinks in Yixing during 2007 to 2017. Agricultural (especially for livestock breeding and aquaculture) and domestic sewage as main sources of NH_4^+-N pollution have, however, been addressed significantly by the restoration program (Oita et al., 2016), as illustrated by the correlation between macroinvertebrate taxonomic richness, Shannon diversity and decreasing NH_4^+-N concentration, supported by the findings of Yi et al. (2018).

Although several factors can influence the outcomes of restoration, investment structure and complementarity amongst different restoration project categories appears as key factors of restoration success. Our results showed that some project categories have a disproportionate effect on nutrient recovery. Even though projects targeting both pollution sources and pollution sinks overall contributed positively to decreases in NH_4^+-N, TP, TN concentrations in Yixing river network, we have showed that:

(1) The same investment amount on restoration projects targeting pollution sources can lead to greater decreases in NH_4^+-N and TP in comparison to equivalent spending on targeting pollution sinks. This result might be driven by effective and timely actions on pollution sources, where nutrients are concentrated prior to dilution and dissipation among water and sediments in rivers and lakes. Thus, projects targeting pollution sources are the most effective way to prevent and decrease water eutrophication by NH_4^+-N and TP (Wurtsbaugh et al., 2019).
(2) The same investment amount on restoration projects targeting agricultural sewage (especially for livestock breeding and aquaculture) can lead to greater decreases in NH_4^+-N and TP (especially for NH_4^+-N) in comparison to those spent on targeting domestic sewage. This result might also be driven by frequent agricultural activities that are one of the main nitrogen sources of the Taihu basin (Liu et al., 2020), and domestic sewage which is one of main pollution sources of TP (Qin et al., 2007).
(3) Decreases in NH_4^+-*N* and TP concentrations showed slightly negative correlation with increasing investment on restoration projects targeting pollution sink and industry waste water. This could be because additional investment in restoration projects targeting pollution sinks and industry waste water could not lead to removal of more NH_4^+-N and TP in a proportionate way. Furthermore, there are many restoration projects on pollution sinks (except for dredging) that do not aim to remove nutrients in a direct way (Bai et al., 2020).
(4) Decreases in TN concentrations in the Taihu river network were correlated with the increasing investment on restoration projects targeting both pollution source and pollution sink. However, deceases in TN concentrations were correlated weakly with the increasing investment on restoration projects on domestic sewage. Additional investment in sanitary sewage treatment plants may therefore not lead to removal of more TN from the waste water in a proportionate way.

The time elapsed since restoration began was also an important ecological driver underpinning ecosystem restoration success (Crouzeilles et al., 2016). Different restoration projects start on different dates by continuous planning, and so the restoration project investments towards the end of our study period may not have had a chance to exhibit their full impact. We can explore the different timelines for abiotic and biotic indices recovery after restoration in the future, if river management agencies invest in long time-scale ater quality and bio-monitoring data.

Overall, our results demonstrate that (i) investments in environmental restoration projects improved water quality and biodiversity despite urban growth (Fig. 7); (ii) investments in source control had a stronger impact on water quality than investments in restoring sinks (Fig. 3); (iii) investments in sink water quality control improved nutrient levels, albeit not as strong as investments in source controls (Fig. 3). Stakeholders should therefore plan carefully the allocation of resources and money when restoring aquatic ecosystems. Studies such as this evaluation of river catchment restoration in SE China have an important role in building the necessary trust in restoration projects for that to happen (Metcalf et al., 2015).

5. Conclusion

Our analysis demonstrates that, despite the unstopped expansion of urbanization, nutrient concentrations and biological indices of benthic invertebrate have improved significantly across most of Yixing catchment as a result of restoration works executed over the study period. Improvements were contingent to the type of restoration project, with some restoration approach showing disproportionate effects on response rates of ecosystem indices and projects targeting pollution-sources leading to the clearest improvements compared with those remediating pollution-sinks. However, in some locations, the recovery of biotic communities appears to lag behind that of nutrients (e.g., nitrogen and phosphorus), likely reflecting the longer time required by long-distance recolonization routes for invertebrates given the level of pre-restoration degradation of the catchment. Overall, our study suggests that ecological damage caused by recent rapid economic development could potentially be mitigated by the combined effect of massive restoration investments synchronized across whole catchments, although these effects can be expected to be muted if urbanization continues apace at the same time.

Data availability statement

Data are available from the Dryad Digital Repository: ⟨https://doi.org/10.5061/dryad.547d7wm8f⟩ (Fu et al. 2021).

CRediT authorship contribution statement

Hong Fu: Methodology, Formal analysis, Resources, Writing – original draft, Writing – review & editing. **Pierre Gaüzère:** Methodology. **Jorge García Molinos:** Writing – review & editing. **Peiyu Zhang:** Writing – review & editing. **Huan Zhang:** Writing – review & editing. **Min Zhang:** Writing – review & editing, Supervision. **Yuan Niu:** Formal analysis, Resources. **Hui Yu:** Formal analysis, Resources. **Lee E. Brown:** Methodology, Writing – review & editing. **Jun Xu:** Conceptualization, Methodology, Formal analysis, Resources, Writing – review & editing, Supervision.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Supplementary materials

Supplementary material associated with this article can be found, in the online version, at doi:10.1016/j.watres.2021.117587.

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