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**Mitigation of urbanisation effects on aquatic ecosystems by synchronous
ecological restoration**

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1 **Abstract**

2 Ecosystem degradation and biodiversity loss have been caused by economic
3 booms in developing countries over recent decades. In response, ecosystem restoration
4 projects have been advanced in some countries but the effectiveness of different
5 approaches and indicators at large spatio-temporal scales (i.e. whole catchments)
6 remains poorly understood. This study assessed the effectiveness of a diverse array of
7 440 aquatic restoration projects including wastewater treatment, constructed wetlands,
8 plant/algae salvage and dredging of contaminated sediments implemented and
9 maintained from 2007 to 2017 across more than 2000km² of the northwest Taihu basin
10 (Yixing, China). Synchronized investigations of water quality and invertebrate
11 communities were conducted before and after restoration. Our analysis showed that
12 even though there was rapid urbanization at this time, nutrient concentrations (NH₄⁺-N,
13 TN, TP) and biological indices of benthic invertebrate (taxonomic richness, Shannon
14 diversity, sensitive taxon density) improved significantly across most of the study area.
15 Improvements were associated with the type of restoration project, with projects
16 targeting pollution-sources leading to the clearest ecosystem responses compared with
17 those remediating pollution-sinks. However, in some locations, the recovery of biotic
18 communities appears to lag behind nutrients (e.g. nitrogen and phosphorus), likely
19 reflecting long-distance re-colonization routes for invertebrates given the level of
20 pre-restoration degradation of the catchment. Overall, the study suggests that ecological
21 damage caused by recent rapid economic development in China could potentially be
22 mitigated by massive restoration investments synchronized across whole catchments,
23 although these effects could be expected to be enhanced if urbanization rates were
24 reduced at the same time.

25 **Keywords:** Ecosystem degradation; wastewater treatment; pollution;
26 macroinvertebrates; water quality
27

28 **1 Introduction**

29 Almost all natural ecosystems on Earth have been disturbed by human
30 development (Sévêque et al., 2020). Billions of dollars are invested annually to restore
31 degraded ecosystems (Zhang et al., 2000), but many countries continue to face a
32 dilemma between the needs of economic development and ecosystem restoration (Liu et
33 al., 2016b). Therefore, adequate assessment of the efficiency of restoration projects in
34 maintaining and restoring natural ecosystem services in line with continuous sustainable
35 development are needed.

36 There are few developing countries, like China, that have implemented so many
37 and diverse ecosystem conservation and restoration projects in recent decades (Zhao et
38 al., 2017), while maintaining rapid economic growth and urbanization. Following the
39 implementation of reform and opening-up policy, China's urban population increased
40 dramatically from 172.5 million in 1978 to 771.2 million in 2015 (Guan et al., 2018).
41 This urban population growth has resulted in severe degradation of aquatic ecosystems
42 as a consequence of land-use change, pollution and hydromorphological modification
43 (Yang et al., 2019). To mitigate the severe ecosystem degradation, the Chinese
44 government initiated major investments in eco-environmental conservation and
45 restoration projects in 2000. The investment in total environmental restoration across
46 the China mainland has increased from almost nothing in 1994 to 1 trillion RMB yuan
47 in 2014 (Zhou et al., 2017). Whilst these factors have made China one of the world's
48 leading investors in ecosystem restoration, there is also a general perception that the
49 national restoration policies and actions have contributed a lot to improve the status of
50 water quality across China (Zhou et al., 2017). However, no study has yet attempted to
51 describe the quantitative relationship between the indices of different restoration

52 projects targeting either pollution sources (the place when pollution was generated) or
53 pollution sinks (natural aquatic ecosystems like rivers) and ecosystem indices (nutrients,
54 biological communities, etc.) across large space and time scale.

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

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

77 expected to result in different effects on the ecosystem restoration, with for example,
78 approaches such as dredging modifying physical habitat and potentially exacerbating
79 stress and delaying recovery.

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

89 **2 Material and methods**

90 **2.1 Study region**

91 The Taihu basin, located at the plain river network region in the downstream area
92 of the Yangtze River, covers an area of approximately 36,895 km² (Fig. 2). The basin,
93 representing 0.4% of China's land area, is heavily populated (40 million residents) and
94 highly industrialized supporting 11% of Gross Domestic Product (GDP) (Yi et al.,
95 2017).

96 This study focused specifically on the upstream areas of the northwest Taihu basin,
97 covering the whole area of Yixing city (Fig. 2). The district covers a total area of 1996.6
98 km² (including 242.29 km² of Lake Taihu), 16.8% of which is occupied by water bodies.
99 The catchment, has a northern subtropical monsoon climate with an average annual
100 temperature of 16.0 °C, and abundant rainfall (1177 mm a year on average,). The urban

101 area of 66.3 km² includes rivers with a density of approximately 2.27 km/km² (Wang et
102 al., 2017).

103 Yixing provides an ideal case study to test our hypotheses for two main reasons:
104 (1) it represents the typical characteristics of the wider Taihu basin (Pan and Zhao,
105 2007), and includes nine of the 13 main tributaries to Lake Taihu, which together
106 account for around 60% of the total flow into the lake Taihu. (2) Between 2007 to 2017,
107 Yixing has spent 8.21 billion RMB (\$1.2B USD) on 440 different aquatic environment
108 restoration projects throughout the catchment.

109 **2.2 Restoration project data**

110 We collected data corresponding to restoration projects which were implemented
111 and maintained between 2007 and 2017. The database contained >440 water
112 environmental restoration projects from the Development and Reform Commission of
113 Jiangsu province (Jiangsu Development and Reform Commission, 2008; 2014). One
114 hundred projects that did not provide information on specific restoration measures, the
115 project scale, or for which the measures taken could not be quantified and converted
116 into removal quantity of nitrogen and phosphorus, were discarded (see below for an
117 explanation of how we calculated these parameters). Similarly, projects that did not
118 have direct impacts on the aquatic environment (garbage disposal, drinking water
119 treatment) were removed from further analysis. We collected data on the location of the
120 restoration works (latitude and longitude) (Fig. 2), type of restoration projects (Fig. 5),
121 starting and completion year, specific restoration measures, project scale, and total
122 investments (Table S1). To eliminate the effects of inflation on the project investment
123 costs, we take 2007 as the base year and make price adjustments to that baseline for
124 other years' investments (Table S2) (Imai, 2018). Projects were classified according to

125 targeted pollution paths: (i) restoration projects targeting pollution sources (e.g.
126 treatment of industrial and agricultural (farming, aquaculture and livestock breeding etc.)
127 wastewater or sanitary sewage), and (ii) restoration projects targeting pollution sinks
128 (e.g. dredging of contaminated sediment, water hyacinth cultivation for removal of
129 pollutants, harvesting of harmful blue-green algae, etc.). For the restoration projects
130 which aimed to control wastewater pollution at source, we further divided those
131 restoration projects into three different categories: (i) industry-focused, (ii) agricultural
132 wastewater (mainly include livestock breeding and aquaculture in this study) and (iii)
133 domestic sewage.

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

140 **2.3 Field sampling**

141 To assess relationships between aquatic ecosystems and nutrient removal
142 efficiency, we monitored the recovery of 63 locations by sampling each site both before
143 (2007) and after (2017) the implementation of the restoration works. Sampling sites
144 were located in the limnetic zone of the lakes, or the rivers of Yixing and collected
145 between July and September during both sampling campaigns (Fig. 2).

146 Benthic macroinvertebrate samples were collected within a 100 m reach for each
147 site using a 0.05 m^2 modified Peterson grab (three grabs per reach), and sieved in situ
148 through a 250- μm mesh. The resulting sieved materials were stored in a cooler box and

149 transported to the laboratory on the same day. In the laboratory, the samples were sorted
150 on a white tray, and all specimens picked out and preserved in 7% formalin solution.
151 Specimens were identified to the lowest feasible taxonomic level under a dissection
152 microscope (Olympus® SZX10) according to several taxonomic keys (Morse et al.,
153 1994; Wang, 2002).

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

164 **2.4 Quantification of restoration effects**

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

$$173 \Delta r = -\ln(\textit{After Restoration}/\textit{Degraded}) \quad (1)$$

174 **Biological indices:** By referring to the applications of biological indices in the
175 Yangtze River Basin, China (Huang et al., 2015), taxonomic richness, Shannon–Wiener
176 index (Simpson, 1949) and % Oligochaeta were selected as representative indices to
177 describe the variation of benthic macroinvertebrate assemblages. A function of species
178 richness and density (Nzengya and Wishitemi, 2000) was used to determine the
179 Shannon diversity.

180 The Hilsenhoff Family Biotic Index (FBI) (Hilsenhoff, 1988) was applied to
181 assess the ecological conditions of each site. FBI score are assigned a tolerance number
182 from 0 (very intolerant) to 10 (highly tolerant), and calculated by equation: $FBI =$
183 $\sum[(TV_i)(n_i)]/N$, where TV_i is the tolerance value of the i^{th} taxon, n_i is the number of
184 individuals in i^{th} taxon, and N is the total number of individuals in the sample. The
185 tolerance value of each family was obtained from Qin et al. (2014) and Wang and Yang
186 (2004). Low FBI values reflect a higher abundance of sensitive invertebrate groups,
187 thus a lower level of organic pollution.

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

195 Biological response ratios were based on a slightly modified formula:

$$196 \quad \Delta r = \ln[(After Restoration + 1)/(Degraded + 1)] \quad (2)$$

197 where, in this case, the degraded and restored conditions were calculated using
198 the biological indices of benthic macroinvertebrate (taxonomic richness, Shannon

199 diversity, percent Oligochaeta and Hilsenhoff FBI). The addition of a unit (+1) to each
200 term in the formula was needed because some sites registered zero values.

201 **2.5 Land use data and urbanization metric**

202 Land use data for Yixing district was derived from 30-m resolution land use
203 maps for 2007 and 2017 (taken as surrogates for existing conditions before and after
204 implementation of restoration) provided by the Resource and Environmental Science
205 Data Center of the Chinese Academy of Sciences (<http://www.resdc.cn>) (Fig. S1). The
206 26 original land use categories were simplified into six categories according to the land
207 resource classification system of China's land use/land cover change (CNLUCC),
208 namely farm land, building land (artificial surfaces), forest land, grassland, water body
209 and barren land (Song and Deng, 2017). The land use transformation matrix for the
210 Yixing district across the six land use categories between 2007 and 2017 is provided in
211 Table S3.

212 The impervious surface area (ISA) of Yixing has increased from 4.36% in 2007
213 to 10.15% in 2017. Prior research has noted that when the ISA increases to a range
214 between 10 and 25%, the impact on aquatic environments is significant (Schueler,
215 1994). However, the water environment in relation to the ISA may vary depending on
216 regional conditions (Luo et al., 2018). Thus, we used the response ratio of impervious
217 surface area ($rISA = \ln(ISA_{2017}/ISA_{2007})$) as a co-variable in subsequent analyses to
218 assess confounding effects of land use change (urbanization) acting in opposition to
219 restoration effects. Land use data and the impervious surface area were handled and
220 calculated using ArcGIS 10.2 (ESRI Company, Redlands, CA, USA) and Fragstats 4.2
221 (McGarigal et al., 2012).

222 **2.6 Data analysis**

223 *2.6.1 Assessing spatial distribution of project indices, ecosystem indices and the* 224 *response ratio of impervious surface area*

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

238 We used a 6-km radius for the circular window (Fig. 2 and S2). The chosen
239 window radius resulted from a compromise between incorporating the range of
240 restoration projects and enough spatial repetition to estimate reliable linear
241 trends in variables, and achieving an adequate coverage of the study area. This
242 generated 4080 spatial windows, each containing at least three sampling sites and nine
243 restoration project sites. Finally, indices (project, nutrients and biological indices) were
244 calculated for the 4080 spatial windows based on the mean of ecosystem indices or the
245 sum of project indices, and calculated the response ratio of impervious surface area for
246 each window. This moving window approach enabled the local spatial trends of each

247 restoration project index to be compared with the local spatial trends of aquatic
248 ecosystem indices (Gauzere et al., 2017).

249

250 *2.6.2 Statistical analysis for all indices*

251 Visual inspection of frequency histograms showed all response ratios of
252 ecosystem indices ($\Delta r\text{NH}_4^+\text{-N}$, $\Delta r\text{TN}$, $\Delta r\text{TP}$, taxonomic richness, Shannon diversity, %
253 Oligochaeta and Hilsenhoff FBI) followed non-normal distributions (Fig. 4). Therefore,
254 we used Wilcoxon signed rank tests to examine whether median response ratios of
255 ecosystem indices were significantly different from zero. Non-metric multidimensional
256 scaling ordination (NMDS) was used to visualize invertebrate communities by site and
257 restoration phase (before/after). Taxon density data were ordinated using Bray–Curtis
258 similarity as the distance measure for the scaling, with square-root transformation to
259 reduce impacts of extremely high counts of individual taxa. Similarity percentage
260 (SIMPER) analysis was used to identify which taxa contributed the most to the average
261 Bray-Curtis dissimilarity between the two-restoration phase.

262 Spearman Rank correlation analyses were performed to test for significant
263 correlations between project investment and removal quantity of $\text{NH}_4^+\text{-N}$, TP, TN by
264 project category. We also used Kruskal-Wallis tests to examine whether investments
265 differed among different restoration project categories. Finally, the relationships
266 between restoration projects and ecosystem recovery were assessed by fitting: a
267 generalized linear mixed model (GLMM) with a Gamma distribution (log link) or
268 Linear Mixed Model (LMM) to each nutrient ($\Delta r\text{NH}_4^+\text{-N}$, $\Delta r\text{TN}$, $\Delta r\text{TP}$). Restoration
269 project investment by category and the response ratio of impervious surface area (rISA)
270 were added as fixed effects, while the number of years since the implementation of the
271 restoration (DurationT) and the time since completion of the restoration (dt = 2017 - end

272 year of the restoration) as random effects. GLMM with Gaussian distribution (log link)
273 or LMM were applied to the biological indices (Δr taxonomic richness, Δr Shannon
274 diversity, Δr % Oligochaeta, β SOR) with nutrients and investment of different
275 restoration categories as fixed effects, rISA as covariate, DurationT and dt as random
276 effects. Removal quantity of nutrient was subsequently omitted from these models
277 because of its significant positive correlation with project investment (see Results
278 section). To explore the interaction effect between urbanization intensity and the
279 strength of restoration, 'rISA*investment of different project categories' was included
280 in the models above.

281 Prior to analysis, the investment of each restoration project category was \log_{10}
282 transformed to constrain the influence of extreme values. We compared the complex
283 model with a null model; models were simplified by removing non-significant terms
284 and verifying the distribution through residuals analysis (Crawley, 2002). Akaike's
285 Information Criterion (AIC) values were used to determine the most parsimonious fit.
286 Model residuals were tested for spatial autocorrelation with Moran's tests (Birk et al.,
287 2020), which showed in all instances no autocorrelation.

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

290

291 **3 Results**

292 **3.1 Relationship between restoration project investments and nutrient** 293 **removal**

294 Spearman rank (Rs) correlations analysis showed a significant positive
295 correlation between project investment and the removal quantity of nutrients (calculated

296 as described in table 1) across project categories (Fig. 3, Table S4). The amount of
297 money invested by the government varied significantly with project category (Kruskal
298 Wallis test, $p < 0.001$). The projects attracting larger investments were, in descending
299 order of magnitude: pollution source, pollution sink, sanitary sewage, industrial
300 wastewater, agricultural sewage (Fig. 3).

301 **3.2 Efficiency of restoration projects on nutrients and biological status**

302 Restoration works were found to be efficient at recovering aquatic ecosystems
303 from their initial degraded condition as shown by their significant effect on almost all
304 assessed ecosystem indices. The concentration of NH_4^+ -N, TN and TP across the whole
305 Yixing river network was significantly lower in restored (2017) than in degraded (2007)
306 aquatic ecosystems, leading to overall positive response ratios (Fig. 4); Taxonomic
307 richness and Shannon diversity of benthic macroinvertebrate were significantly higher
308 in restored (2017) than in degraded (2007) sites (mean response ratio = 1.085, 0.415,
309 $P < 0.001$, Fig. 4). Percent Oligochaeta was significantly lower in restored (2017, 17.53%
310 $\pm 16.65\%$) than in degraded (2007, 40.78% $\pm 39.70\%$) sites. Hilsenhoff FBI of benthic
311 macroinvertebrate communities showed no significant difference between degraded
312 (2007) and restored (2017) ecosystems.

313 The composition of benthic macroinvertebrate communities differed
314 significantly between degraded (2007) and restored (2017) time periods
315 (PERMANOVA, $p < 0.01$; final stress=0.128, Fig. 6). SIMPER analysis identified eight
316 species cumulatively contributing $> 70\%$ to the dissimilarity between restored (2017)
317 and degraded (2007) invertebrate communities (Table S5). They were *Limnodrilus*
318 *hoffmeisteri*, *Bellamyia aeruginosa*, *Corbicula fluminea*, *Branchiura sowerbyi*,
319 *Parafossarulus eximius*, *Neocaridina denticulata*, *Exopalaemon modestus* and
320 *Parafossarulus striatulus* in decreasing order. Some sensitive species to anthropogenic

321 pressures with low tolerance values recolonized after restoration (2017). For example,
322 river flies *Heptagenia* sp., *Ephemera orientalis* and *Ceratopsyche* sp.. The partitioning
323 of the Sørensen dissimilarity index was dominated by species turnover (β_{SIM}) (mean =
324 0.44, SD = 0.36), implying that in any given site an average of 44% of the species were
325 unique to the time (either 2007 or 2017 site assemblage). In contrast, the nestedness
326 component (β_{SNE}) was much lower (mean = 0.34, SD = 0.33), implying that weaker
327 patterns of species losses or gains from pre-existing communities have occurred
328 between 2007 and 2017 (Fig. 4).

329 **3.3 Effects of restoration projects on aquatic ecosystem status**

330 Examination of the marginal effect of project investment amount by category on
331 nutrients (Fig. 3) showed a significant correlation of decreasing river network NH_4^+ -N
332 concentrations (i.e., increasing response ratios) with increasing investment on projects
333 targeting pollution sources but not those targeting pollution sinks (marginal $R^2 = 0.20$, p
334 < 0.001). On the contrary, decreasing TN and TP concentrations were positively
335 correlated with increasing investment on restoration projects targeting both pollution
336 sources and sinks (marginal $R^2 = 0.23$, $p < 0.001$; marginal $R^2 = 0.19$, $p < 0.001$).
337 Decreasing NH_4^+ -N and TP concentrations correlated with increasing investment in
338 both restoration projects targeting agricultural and domestic sewage, but not those
339 targeting industry wastewater (marginal $R^2 = 0.14$, $p < 0.001$; marginal $R^2 = 0.19$, $p <$
340 0.001). Decreasing TN concentrations were negatively correlated with the increasing
341 investment on restoration projects targeting sanitary sewage (marginal $R^2 = 0.32$, $p <$
342 0.001). A significant interaction was evident between the response ratio of impervious
343 surface area and investments of different restoration project categories and nutrient
344 responses. For example, poor nutrient responses were associated with the growth of

345 impervious surface area ($p < 0.001$, Fig. 7), but these effects were overcome where
346 restoration projects were large but impervious area increased minimally.

347 For the biological indices, increased Shannon diversity and taxonomic richness
348 over time showed significant inverse relationships with NH_4^+ -N concentrations, and a
349 positive association with increasing investment on restoration projects targeting sanitary
350 sewage (Shannon marginal $R^2 = 0.63$, $p < 0.001$; richness marginal $R^2 = 0.73$, $p <$
351 0.001). Decreasing Oligochaeta relative abundance was associated with investment
352 value of restoration projects both targeting agricultural and industrial wastewater
353 (marginal $R^2 = 0.47$, $p < 0.001$). Increasing β_{SOR} was correlated positively with
354 investment for all three project categories targeting pollution sources (marginal $R^2 =$
355 0.32 , $p < 0.001$). There was evidence for significant interaction between the response
356 ratio of impervious surface area and investments of different restoration project
357 categories and biological index responses. For example, poor biological responses were
358 associated with the growth of the impervious surface area ($p < 0.001$, Fig. 7), but these
359 effects were overcome where restoration projects were large but impervious area
360 increased minimally.

361 **4 Discussion**

362 This study has provided new insights to understand the effectiveness of
363 catchment-scale restoration towards increasing water quality and biodiversity in rivers
364 of China, building on knowledge from previous studies from a variety of ecosystems in
365 other parts of the world (Benayas et al., 2009; Crouzeilles et al., 2016). A large data set
366 comprising hundreds of different aquatic ecosystem restoration projects undertaken over
367 the last two decades in a large urban district of China show that implementation of
368 large-scale restoration projects can, to some extent, mitigate the environmental

369 degradation as a result of economic boom. In Yixing, recovery occurred despite
370 ongoing rapid economic growth and urbanization, although it should be noted that the
371 impervious surface area reached only 10.15% at the bottom end of Schueler's (1994)
372 10-25% range for significant impacts on water quality. Further urbanization may
373 therefore negate the positive aspects of restoration observed to date.

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

385 In contrast, the overall Hilsenhoff FBI showed no significant difference between
386 degraded (2007) and restored (2017) years of sampling. Despite the enormous
387 investment in restoration, there were 25 sites showing increases in Hilsenhoff FBI
388 scores, due to some higher tolerance taxa still remaining, and taxonomic richness in
389 2007 being much lower (average: 2.16) than in 2017 (average: 8.58). During the 10
390 years of the study, Yixing has seen its GDP increase from 42.80 billion RMB in 2006 to
391 155.83 billion RMB in 2017 (Fig. 2), accompanied by 45% growth of artificial surfaces
392 (Fig. S1 and Table S3). The effects of urbanization (hydromorphology and hydrological
393 alteration, run off pollution) are likely to have suppressed the level of biotic recovery of

394 freshwater macroinvertebrates that may have occurred from restoration efforts in
395 isolation by increasing the role of other stressors (Gál et al., 2019). Despite this urban
396 cover growth, water quality of Chinese inland waters has clearly improved generally
397 over recent decades with restoration efforts (Zhou et al. (2017).

398 On the other hand, we found that the response of biotic indices to restoration
399 projects appeared to lag behind nutrients (NH_4^+ -N, TN and TP), with the standardized
400 responses of nutrients being greater than those of biotic indices. However, compared to
401 the degraded time-period (2007), some species that are sensitive to anthropogenic
402 pressures (low tolerance values) recolonized after restoration (2017) including the river
403 flies *Heptagenia* sp. and *Ephemera orientalis*. Increases in the *Heptageniidae* are in line
404 with Pedersen et al. (2007) who reported they increased significantly in abundance after
405 a short-term restoration (three years) at the Skjern River reaches, Denmark. The
406 composition of benthic invertebrate communities differed significantly between
407 degraded (2007) and restored (2017) periods. Eight species cumulatively contributed >
408 70% to the dissimilarity between restored (2017) and degraded (2007) invertebrate
409 communities (Table S5), *Limnodrilus hoffmeisteri* (turbid worms) and *Branchiura*
410 *sowerbyi* (crustaceans) both decreased more in restored than in degraded rivers. These
411 species are widely used as an indicator of organic pollution throughout China (Gorni et
412 al., 2018), thus their decreasing abundance provides important ecological evidence for
413 restoration success alongside the water quality improvements. However, *Limnodrilus*
414 *hoffmeisteri* was still the co-dominant species in some sampling sites in Yixing river
415 network both in degraded and restored time-periods; this is not surprising as they are
416 widely distributed throughout global freshwater ecosystems (Armendáriz and César,
417 2001). In contrast, snails and clams such as *Bellamya aeruginosa* and *Corbicula*
418 *fluminea* increased more in restored than in degraded. Recovery of these native snail

419 and bivalve populations can be expected to further help improve the water quality given
420 their roles as deposit or filter feeders that remove particulates (Zhang et al., 2014). The
421 relative abundance of snails like *Bellamya aeruginosa* increased in > 20 sites over time,
422 most likely because some native snails have been reintroduced by restoration activities
423 in attempts to enhance algal removal. While *Bellamya aeruginosa* and *Corbicula*
424 *fluminea* are common species which are widely distributed in eutrophic shallow lakes in
425 China (Zhu et al., 2013). Although biological indices appear to lag behind abiotic
426 indices like nutrients, sampling frequency limited our ability to elucidate more clearly
427 the relationship between these indicators.

428 Even though the response ratio of taxonomic richness and Shannon diversity of
429 benthic macroinvertebrate was significantly higher in restored (2017) than in degraded
430 (2007) aquatic ecosystems, taxonomic richness and Shannon diversity of benthic
431 invertebrate only showed significant positive correlation with the increasing project
432 investment on sanitary sewage removal, the decline of NH_4^+ -N concentrations in Yixing
433 river network, and the interaction effect between the response ratio of impervious
434 surface area and project investment on agricultural sewage removal. The muted
435 improvements of biological indices may be due to two reasons: (i) restoration measures
436 on pollution sink mainly include dredging of contaminated sediments, which will
437 negatively affect the habitat of benthic macroinvertebrates; (ii) water quality in Yixing
438 is improved but still not to a high level, and hydromorphological alterations remain
439 throughout the catchment, limiting recovery potential. Additionally, only 14.4% of
440 investments were targeted at pollution sinks in Yixing during 2007 to 2017. Agricultural
441 (especially for livestock breeding and aquaculture) and domestic sewage as main
442 sources of NH_4^+ -N pollution have, however, been addressed significantly by the
443 restoration program (Oita et al., 2016), as illustrated by the correlation between

444 macroinvertebrate taxonomic richness, Shannon diversity and decreasing NH_4^+ -N
445 concentration, supported by the findings of Yi et al. (2018).

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

452 (1) The same investment amount on restoration projects targeting pollution sources
453 can lead to greater decreases in NH_4^+ -N and TP in comparison to equivalent spending
454 on targeting pollution sinks. This result might be driven by effective and timely actions
455 on pollution sources, where nutrients are concentrated prior to dilution and dissipation
456 among water and sediments in rivers and lakes. Thus, projects targeting pollution
457 sources are the most effective way to prevent and decrease water eutrophication by
458 NH_4^+ -N and TP (Wurtsbaugh et al., 2019).

459 (2) The same investment amount on restoration projects targeting agricultural
460 sewage (especially for livestock breeding and aquaculture) can lead to greater decreases
461 in NH_4^+ -N and TP (especially for NH_4^+ -N) in comparison to those spent on targeting
462 domestic sewage. This result might also be driven by frequent agricultural activities that
463 are one of the main nitrogen sources of the Taihu basin (Liu et al., 2020), and domestic
464 sewage which is one of main pollution sources of TP (Qin et al., 2007).

465 (3) Decreases in NH_4^+ -N and TP concentrations showed slightly negative
466 correlation with increasing investment on restoration projects targeting pollution sink

467 and industry waste water. This could be because additional investment in restoration
468 projects targeting pollution sinks and industry waste water could not lead to removal of
469 more NH_4^+ -N and TP in a proportionate way. Furthermore, there are many restoration
470 projects on pollution sinks (except for dredging) that do not aim to remove nutrients in a
471 direct way (Bai et al., 2020).

472 (4) Decreases in TN concentrations in the Taihu river network were correlated with
473 the increasing investment on restoration projects targeting both pollution source and
474 pollution sink. However, decreases in TN concentrations were correlated weakly with
475 the increasing investment on restoration projects on domestic sewage. Additional
476 investment in sanitary sewage treatment plants may therefore not lead to removal of
477 more TN from the waste water in a proportionate way.

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

485 Overall, our results demonstrate that (i) investments in environmental restoration
486 projects improved water quality and biodiversity despite urban growth (Fig. 7); (ii)
487 investments in source control had a stronger impact on water quality than investments in
488 restoring sinks (Fig 3); (iii) investments in sink water quality control improved nutrient
489 levels, albeit not as strong as investments in source controls (Fig 3). Stakeholders
490 should therefore plan carefully the allocation of resources and money when restoring
491 aquatic ecosystems. Studies such as this evaluation of river catchment restoration in SE

492 China have an important role in building the necessary trust in restoration projects for
493 that to happen (Metcalf et al., 2015).

494 **5 Conclusion**

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

509

510 **Data availability statement**

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

513

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522

523 **Author contributions**

524 Conceptualization, J.X.; methodology, J.X., H.F., P.Z. and L.E.B; formal analysis, H.F.
525 Y.N., H.Y. and J.X.; resources, H.F., Y.N., H.Y. and J.X.; writing—original draft
526 preparation, H.F.; writing-review and editing, H.F., J.G.M., J.X., P.Z., H.Z., M.Z.,
527 L.E.B. and J.X.; supervision, J.X. and M.Z.

528

529 **Conflict of interest**

530 The authors declare no conflict of interest.

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725

726

727 **Figure legends**

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

733

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

741

742 **Fig. 3.** Scatter plots showing the relationships between project investments of different
743 restoration project categories and either of (a-f) removal quantities for the different
744 nutrients ($\Delta r\text{NH}_4^+\text{-N}$, $\Delta r\text{TN}$, $\Delta r\text{TP}$) (the marginal boxplots in a-f show the investment
745 distribution on different restorations) and response ratio (g-l) of different restoration
746 project categories,: (a-c) restoration measures for pollution source (Spearman rank R_s =
747 0.62, 0.58, 0.55, $p < 0.001$) and pollution sink (R_s = 0.79, 0.85, 0.82, $p < 0.001$); (d-f)
748 three main categories of restoration measures for pollution source, which include
749 restoration measures for industry waste water (R_s = 0.92, 0.95, 0.94, $p < 0.001$),
750 agricultural (R_s = 0.89, 0.89, 0.89, $p < 0.001$) and sanitary (R_s = 0.70, 0.68, 0.69, $p <$

751 0.001) sewage. Marginal effects of investment of different restoration project categories
752 on each nutrient ($\Delta r\text{NH}_4^+\text{-N}$, $\Delta r\text{TN}$ and $\Delta r\text{TP}$): (g-i) restoration measures for pollution
753 source and sink; (j-l) three main categories of restoration measures for pollution source,
754 which include restoration measures for industry waste water, agricultural and sanitary
755 sewage. GLMM or LMM regression lines are given where a correlation was significant
756 ($p < 0.05$). The initial unit of investment is 10^5 RMB, and was \log_{10} transformed before
757 inclusion in models.

758

759 **Fig. 4.** Response ratios of $\text{NH}_4^+\text{-N}$, TN, TP and taxonomic richness (Richness),
760 Shannon diversity, percent Oligochaeta, FBI of benthic macroinvertebrate in restored
761 (2017) compared with degraded ecosystems (2007) (a, b). All response ratios differed
762 significantly from zero (Wilcoxon signed rank tests, $p < 0.001$) except for Hilsenhoff
763 FBI. The mean and SD are given alongside the overall data distribution for each metric.
764 (c) The partition of temporal total dissimilarity (β_{SOR} —solid black line) into nested
765 resultant dissimilarity (β_{SNE} —solid grey line) and turnover (β_{SIM} —dashed lines) for
766 beta diversity of benthic macroinvertebrates in Yixing from 2007 to 2017.

767

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

772

773 **Fig. 6.** NMDS biplots showing changes in community composition of benthic
774 invertebrates among restoration projects between their initial degraded (2007) and final
775 restored (2017) states in the Yixing river network with indication of (a) the individual
776 taxa (denoted by S) and (b) sampling sites (denoted by the numbers). An outlier was
777 removed from this figure because it had only one scarce species in 2007 and had 16

778 species in 2017. S1, *Limnodrilus hoffmeisteri*; S2, *Bellamyia aeruginosa*; S3,
779 *Branchiura sowerbyi*; S4, *Corbicula fluminea*; S5, *Parafossarulus eximius*; S6, *Nephtys*
780 *oligobranchia*; S7, *Parafossarulus striatulus*; S8, *Neocaridina denticulata denticulata*;
781 S9, *Semisulcospira cancelata*; S10, *Gammarus* sp.; S11, *Exopalaemon modestus*; S12,
782 *Alocinma longicornis*; S13, *Branchiodrilus hortensis*; S14, *Cricotopus bicinctus*; S15,
783 *Limnoperna fortunei*; S16, *Procladius* sp.; S17, *Physa* sp.; S18, *Tanytus chinensis*; S19,
784 *Radix swinhoei*; S20, *Ceratopsyche* sp.; S21, *Heptagenia* sp.; S22, *Chironomus*
785 *plumosus*; S23, *Ploypedilum scalaenum*; S24, *Prosilocerus akamusi*; S25, *Acuticosta*
786 *chinensis*; S26, *Anodonta woodiana pacifica*; S27, *Anodonta woodiana elliptica*; S28,
787 *Glossiphonia* sp.; S29, *Dicrotendipus lobifer*; S30, *Semisulcospira libertina*; S31,
788 *Stenothyra glabra*; S32, *Unio douglasiae*; S33, *Glossiphonia complanata*; S34,
789 *Glyptotendipes tokunagai*; S35, *Nemertea* sp.; S36, *Microchironomus tabarui*; S37,
790 *Aulodrilus* sp.; S38, *Nereis japonica*; S39, *Tabanus* sp.; S40, *Macrobrachium*
791 *nipponense*; S41, *Stictochironomus* sp.; S42, *Cricotopus sylvestris*; S43, *Tanytarsus*

792 *chinyensis*; S44, *Laccophilus* sp.; S45, *Rhaphium* sp.; S46, *Glyptotendipes pallens*; S47,
793 *Lamelligomphus* sp.; S48, *Helobdella fusca*; S49, *Glossiphonia lata*; S50, *Aciagrion* sp.;
794 S51, *Baetis* sp.; S52, *Harnischia fuscimana*; S53, *Ephemera orientalis*; S54, *Cricotopus*
795 *vierriensis*; S55, *Erpobdella octoculata*; S56, *Hippeutis cantori*; S57, *Glyptotendipes*
796 sp.; S58, *Polypedilum nubeculosum*; S59, *Cricotopus trifascia Edwards*; S60,
797 *Orientogomphus* sp.; S61, *Burmagomphus* sp.; S62, *Tachaea chinensis*; S63, *Lestidae*
798 sp.; S64, *Cryptochironomus* sp.; S65, *Cercion* sp.; S66, *Calopteryx* sp.; S67, *Holorusia*
799 sp.; S68, *Brachythemis* sp..

800

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

805

806 **Table legends**

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

808

809 **Table 2.** Results of GLMM and LMM for abiotic environmental indices (NH_4^+ -N, TN,

810 TP), the investments of different restoration project categories and the intensity of

811 urbanization (the response ratio of impervious surface area (rISA), covariate) on

812 biological parameters (taxonomic richness, Shannon diversity, % Oligochaeta, β_{SOR}).

813 Variables are given when a correlation was significant ($P < 0.05$). Data in grey show

814 positive correlations. s_Agric_inv, investment targeting agricultural sewage; s_san_inv,

815 investment targeting sanitary sewage; s_ind_inv, investment targeting industry waste

816 water; sinkPinvstm, investment targeting pollution sink.

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820