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Catastrophic effects of sand mining on macroinvertebrates in a large shallow lake with implications for management

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Abstract： Sand mining is a human activity that is increasing in inland waters and has profound effects on entire aquatic ecosystems. However, current knowledge of the effects of sand mining on freshwater lake ecosystems remains limited, especially for biotic communities. Here, we investigated the responses of macroinvertebrates to indiscriminate sand mining in a large shallow lake of China. Our results indicated that sand mining significantly increased the content of suspended particulate matter, total nitrogen, total phosphorus and chlorophyll *a* in the water column both in the sand mining area and the area adjacent to the dredging activities. While there was significantly lower total nitrogen and the total phosphorus content of the sediment were observed in the sand mining area. In terms of benthic animals, there were reductions of the macroinvertebrate density and biomass of 89.80% and 99.54%, respectively, and there was a considerable decline of the majority of macroinvertebrate taxonomic taxa as well as biological traits observed in the sand mining area due to direct dredging-induced substrate deterioration and high turbidity water. Moreover, in the area adjacent to the dredging activities, dredging-induced high turbidity water also resulted in 28% and 79% decreases in macroinvertebrate density and biomass, respectively, with a significant decrease in the densities of Bivalvia and Polychaeta but an increase in the density of Crustacea. In terms of biological traits, species (e.g., *Grandidierella* sp. and *Sphaerium lacustre*) characterized by a small body size, short life cycle and dietary sources mainly from sediment were typically associated with the ecological condition of the indirect effects of the dredging activities. Taxa (e.g., *Corbicula fluminea*) with a larger body size and longer life cycle that are filter feeders should be favored by the ecological conditions of the reference sites. For biomonitoring of sand mining perturbations, a number of taxonomic and biological trait indicators were proposed in our study based on indicator value analysis, and the general applicability of trait-based indicators was

highlighted. We also suggest that the biodiversity indices may be less suitable indicators of sand mining effects. Given the limited understanding of the responses of macroinvertebrates to sand mining in inland freshwaters, we believe that our results may provide important information for biomonitoring of sand mining activities and provide scientific management support to governments.

Key words: sand dredging, macroinvertebrates, biomonitoring, biological traits

1. Introduction

Sand is an indispensable material for many buildings and infrastructures. Hence, the rapid development of the construction industry has increased the demand for sand, especially for emerging developing countries. Sand mining has been widely observed in inland and coastal waters with profound effects on entire aquatic ecosystems (Newell et al., 1998; Wu et al., 2007; Mensah, 2010). It has been argued that sand mining should be considered as an aspect of global environmental change (Mayekar, 2006) in the view of its worldwide extent and the magnitude of its impacts (Bendixen et al., 2019). Sand mining in aquatic systems was first reported in developed countries, accompanied by serious coastal erosion, dramatic channel incision (Kondolf, 1994; Mayekar, 2006; Thornton et al., 2006), and profound effects on biotic communities (Boyd et al., 2005; Fraser et al., 2017). With its globalization, sand mining activities have increased in developing countries (Padmalal et al., 2008; Mensah, 2010), particularly in China, which is undergoing rapid industrialization (Lu et al., 2007; Li et al., 2014; Chen, 2017). Most alarmingly, indiscriminate sand mining is rampant in many inland water bodies of China (Bendixen et al., 2019), almost in all great lakes (e.g., Lake Poyang, Lake Dongting and Lake Hongze) and rivers (e.g., Yangtze River and Yellow River) (Lu et al., 2007; Leeuw et al., 2010; Duan et al., 2018). As a result, exhaustive exploitation seriously degrades freshwater ecosystems, causing high turbidity water (Wu et al., 2007; Cao et al., 2017) and serious substrate deterioration (Wu et al., 2007), changing the lacustrine discharge ability (Lai et al., 2014) and lake level (Mei et al., 2016) as well as even causing the release of toxic substances (e.g., fluoride) from sediments (Carr, 1954; Wang et al., 2015).

Macroinvertebrates are essential for the functioning of lake ecosystems. They can change the physicochemical environment of the sediment-water interface and promote nutrient cycling and energy flows (Zhang et al., 2010); they also link the benthic and pelagic food-webs and are of great significance to fishery production (Covich et al., 1999). Many macroinvertebrates dwell in the sediment and often stay immobile and therefore have a weak ability to escape from the negative impacts of sand mining. Moreover, as demonstrated by several studies, sand mining always causes a substantial increase in water turbidity (Wu et al., 2007; Cao et al., 2017; Duan et al., 2018), which may be particularly harmful to filter feeders (e.g., Bivalvia) by restricting their feeding behaviors (Österling et al., 2007; Bucci et al., 2008; Österling et al., 2010). Therefore, dredging-induced habitat degradation maybe also raises biodiversity concerns. However, knowledge of the influence of sand mining on freshwater benthic fauna is currently limited. As far as we know, only one earlier study has assessed the effects of sand mining on macroinvertebrates in a freshwater lake (Meng et al., 2018). These authors observed a substantial decline of benthic animals in the area directly affected by commercial sand mining after one year duration of these activities. In contrast, increased taxa richness, abundance and biomass were observed in the area adjacent to dredging activities (Meng et al., 2018). However, the wider applicability of their results needs to be validated, and bioindicators of sand mining disturbances need to be identified.

Lake Hongze, the fourth largest freshwater lake in China, was subjected to severe indiscriminate sand mining from 2012 to March 2017. Sand mining has caused a significant increase of water turbidity in almost the entire offshore area of this lake (Cao et al., 2017). In contrast, no other significant human-induced changes or natural conditions were observed during this period (Cao et al., 2017). Therefore, Lake Hongze may be an optimal area for a case study to elaborate upon the effect of indiscriminate sand mining on lake conditions and communities of benthic macroinvertebrates.

We studied the environmental conditions and macroinvertebrate communities before (in 2008-2009) and after (in 2016) the onset of large-scale sand mining activities (in 2012) in Lake Hongze. We sampled the offshore area of the lake from sites directly and indirectly affected by sand mining activities. Our main objective was to investigate the responses of macroinvertebrates to large-scale indiscriminate sand mining in a large freshwater lake in China. Moreover, we give recommendations for prevention of harmful sand mining effects and evaluate the implication for lake management. We expected that the substantial alteration of habitat quality and sediment disturbance in the sand mining area may result in the collapse of macroinvertebrate assemblages. We also expected that, in the area indirectly subjected to dredging effects, increased water turbidity would also cause changes in benthic communities.

2. Materials and Methods

2.1 Study area

Lake Hongze (33°06′-33°40′N, 118°10′–118°52′E), with a surface area of 1597 km^2, is the fourth largest freshwater lake in China. Lake Hongze is located downstream of the Huai River; the mean depth and water retention time of this lake are approximately 1.9 m and 35 days, respectively, with a total water volume of 3.04×10^9 m^3 (Wang and Dou, 1998). Lake Hongze supplies multiple ecosystem services with significant socio-economic benefits to its local people. These services include aquaculture, agricultural irrigations, flood control, tourism and drinking water supply. More importantly, this lake is an important channel of the East Route of the South-to-North Water Diversion Transfer Project (Yin, 2013). However, Lake Hongze has undergone rapid eutrophication with the input of nutrients from agricultural non-point sources during the past three decades (Cai et al., 2016).

In addition, over the past years, sand mining operations have been rampant in Lake Hongze (Cao et al., 2017). Indiscriminate sand mining was first observed in spring 2007, with slight annual growth from 2007 to 2011 (Zuo and Zhu, 2011). However, those small-scale practices in 2007-2011 were conducted at the edge of the lake, thus exerting a limited effect on the environmental conditions in the offshore area (Cao et al., 2017). The large-scale sand mining operations started in 2012, and by May 2015, the number of sand mining vessels sharply increased to more than 600 (Table S1), with the much bigger steel dredgers replacing previous concrete ships for improving yields. The dredging depths also increased from 20-30 m to 40-50 m in the sediment (Yan, 2015). Consequently, the increasing severity of sand mining in Lake Hongze drew high attention from the Chinese central government, and the unprecedented control efforts were implemented. Due to these joint efforts, indiscriminate sand mining was finally stopped in Lake Hongze by March 2017 (Duan et al., 2018).

2.2 Collection of environmental variables and macroinvertebrates

In November 2008 and June 2009, the environmental variables and benthic macroinvertebrates were sampled from fourteen sites located in the offshore area of Lake Hongze (Figure S1). Similar sampling was conducted monthly in 2016. All fourteen sites sampled in 2008-2009 shared similar habitat characteristics. The substrata of these sites were a combination of silt and clay, and no macrophytes were found. Moreover, all of these sites were characterized by very shallow water depth (<3 m), suffering strong wind-induced resuspension of sediment (Zhu and Dou, 1993).

All eight sites sampled in 2016 were also located in the offshore area (Figure 1) without growth of macrophytes. Among these sites, HZ8 was the sand mining site used for our field observations, and this site was subjected to direct extraction of substrate and dredging-induced high turbidity water, thus it was considered as a directly affected site (DAS). Other study sites (HZ1-HZ7) were subjected to dredging-induced increases in water turbidity but were without direct extraction of substrate. These sites were considered to be indirectly affected sites (IAS). Given that the large-scale sand mining activities were conducted from 2012 to 2017, they caused significant changes on environmental conditions in the offshore areas (e.g., the substantial increase of water turbidity) (Duan et al., 2018). Thus, the environmental conditions and benthic macroinvertebrates of the fourteen sites in 2008-2009 could serve as type-specific reference sites (RS) to the indiscriminate sand mining perturbations (DAS and IAS).

During both sampling periods, 2008-2009 and 2016, water depth (WD) and Secchi-depth (SD) were measured in the field with a Speedtech SM-5 Portable Depth Sounder and Secchi Disk, respectively. Electrical conductivity (EC), dissolved oxygen (DO), pH, and water temperature (WT) of surface water were acquired with a YSI 6600 V2 Multi-Parameter Water Quality Sonde. Moreover, depth-integrated water samples were collected and kept in cool and shaded conditions before being transported to the laboratory. Water chemistry parameters, including total nitrogen (TN), total phosphorus (TP), permanganate index (COD_{Mn}) and chlorophyll *a* (Chl-*a*) were measured in the laboratory based on standard methods (APHA, 2012). In addition, surface sediment samples were collected, and total nitrogen (TNs) and total phosphorus (TPs) of the sediment were determined by subsampling approximately 30 mg of the dried sediment from each station. The samples were ground with a mortar and pestle and weighed. Then, 25 mL of distilled water was added, and the samples were analyzed after thawing, using combined persulfate digestion followed by spectrophotometric analysis for phosphate and nitrate (Ebina et al., 1983). Benthic macroinvertebrates were sampled with a modified Peterson grab (0.05 m^2) with three replicates for each sampling occasion along with the collection of environmental variables. Samples were sieved using a 250 μm aperture mesh. The sieved materials were preserved in low temperature conditions and transported to the laboratory, where the samples were sorted on a white tray, and the macroinvertebrates were preserved with 8% buffered formalin. Specimens were identified to the lowest taxonomic level possible using available taxonomic guides (Liu, 1979; Morse et al., 1994; Wang and Wang 2011), and then they were counted and blotted dry to determine the wet weight with an electronic balance (Sartorius BS-124, Precision: 0.1 mg).

In addition, for quantifying long-term water turbidity dynamics, suspended particulate matter (SPM) was acquired based on the optical remote-sensing methods from 2002 to 2016 of the eight sites in Lake Hongze (Figure 1). The algorithms for SPM concentrations were composed of empirical algorithms established based on the relationship between either a single band or combinations of several bands and SPM concentrations (Ondrusek et al., 2012). In addition, the MODIS data at 250-m and 500-m resolutions were used to study the SPM variability. The MODIS/Aqua Level-1A data of Lake Hongze from July 2006 and December 2016 were downloaded from NASA's archive (https://oceandata.sci.gsfc.nasa.gov/), and 1602 cloud-free images were selected after visual examination. Level-1A data were preprocessed with software SeaDAS 7.3 to generate Level-1B. Then, a partial atmospheric calibration was used to the Level-1B data to correct for the gaseous absorption (mainly by ozone) and Rayleigh scattering (molecular) effects. The SPM concentration of Lake Hongze was calculated as follows:

$$SPM=exp\ (15.4\times[R_{rc}(645)-R_{rc}(1240)])+1.994.$$

Detailed descriptions of this equation are provided in Cao et al., (2017).

2.3 Data analysis

2.3.1 Index selection

The Shannon-Wiener index and Pielou's evenness index were chosen to investigate the response of the macroinvertebrate community to indiscriminate sand mining (Shannon, 1948; Smith, 1977). As for the response of biological traits, four trait groups were selected and each were divided into 2-7 categories. Of the studied traits, body size was classified into five categories based on the body length of species: 0.5-1 cm, 1-2 cm, 2-4 cm, 4-8 cm and >8 cm. Life cycle duration was classified into two categories: ≤1 year and >1 year. Based on their locomotion and substrate relation, species were classified into skaters, divers, crawlers, burrowers, interstitial and temporarily attached; species were also classified into absorbers, deposit feeders, shredders, scrapers, filter feeders, predators and parasites based on their feeding habits. The trait information was acquired mainly from the literature (Morse, 1994; Tomanova and Usseglio-Polatera, 2007; Statzner, 2010) and online resource www.freshwaterecology.info (Schmidt-Kloiber and Hering, 2015). We used the fuzzy coding procedure in biological trait classification for our data. In the fuzzy coding procedure, the taxa intrinsically representing several categories of a trait are assigned respective weights for these trait categories (Francois, 2010). In our biological trait classification, we used the weights given in the literature for the trait categories (see, e.g., Schmidt-Kloiber and Hering, 2015).

The indicator value index (IndVal) was used for the data related to both the macroinvertebrate taxonomic taxa and biological traits among the RS, DAS and IAS to discover the indicators that were typically associated with the three different ecological conditions. This method combines measures of fidelity (relative frequency) and specificity (relative abundance) to generate IndVals of each taxon expressed as percentages (Dufrêne and Legendre 1997). A random reallocation procedure with 5,000 permutations was used to test for the significance of IndVals. The calculation of IndVals was performed with the "indval" function implemented in the R package "labdsv".

2.3.2 Spatiotemporal comparisons and statistical analysis

Environmental variables and the benthic macroinvertebrate community that were considered as the reference condition were collected in November 2008 and June 2009. Therefore, to examine the effects of sand mining on the environmental condition and macroinvertebrates, the data collected in 2016 in the same months (June and November) and the adjacent months (May and October 2016) with similar temperature levels (Table 1) as reference data were used for comparison among the RS, DAS and IAS.

Multivariate analyses were performed to investigate differences both in the macroinvertebrate taxonomic composition and biological trait group among the RS, DAS and IAS using the analysis of similarity (ANOSIM) (Clarke, 1993; Hammer et al., 2001). The ANOSIM was performed with PAST (Paleontological STatistics v1.9) using Bray-Curtis similarity matrices obtained from log(x+1) transformed density data. The ANOSIM with 999 permutations was used to test the significance of differences in the benthic taxonomic composition and biological trait group, and Global R values represented the degrees of separation that were used to compare relative discriminatory powers of the various datasets, with a score value of 1 indicating complete separation and 0 indicating no separation. Differences in the environmental variables, macroinvertebrate community parameters and biodiversity indices among the RS, DAS and IAS were tested using a one-way ANOVA followed by a post hoc Tukey test or a Kruskal-Wallis followed by a Mann-Whitney U test depending on the normality and homoscedasticity of the data.

3. Results

3.1 Environmental effects

Compared with the RS, significantly (P<0.05) higher values of EC, TN, TP, COD_{Mn}, Chl-*a*, and lower values of SD were observed in both the DAS and IAS, indicating that the sand dredging process may release nutrients from the sediment and promote eutrophication. In addition, the WD difference of HZ8 minus HZ7 increased from -0.63 m in 2008-2009 to 1.74 m in 2016, implying that sand mining deepened the lake bed approximately 2.4 m in the DAS. For sediment, large-scale sand mining has significantly decreased TNs and TPs in the DAS (P<0.05) (Table 1).

The average SPM concentrations did not differ significantly neither in DAS nor IAS (P>0.05) between the period from 2002 to 2006 (total absence of sand mining of Lake Hongze) and the period from 2007 to 2011 (with small-scale sand mining at the edge of the lake). This result implies that the small-scale sand mining may have a rather limited influence on the water environment in the offshore areas. In addition, the average SPM values in the DAS and IAS from 2002 to 2011 were 31.25±9.70 mg/L and 30.45±8.10 mg/L, respectively (before large-scale sand mining). A pronounced (P<0.05) increase of SPM concentrations was observed in the DAS and IAS from 2012 to 2016 (after large-scale sand mining), with average values of 53.68±24.87 mg/L and 48.64±17.56 mg/L, respectively (Figure 2).

3.2 Macroinvertebrate taxonomic composition and biological traits

The ANOSIM Global R values showed that both the taxonomic communities and biological trait groups differed significantly among the RS, DAS and IAS (P<0.01). The pairwise comparison of the ANOSIM indicated that when comparing both the taxonomic communities and biological trait groups, the sand mining affected sites (DAS and IAS) separated significantly from the RS (P<0.01). In addition, taxonomic communities differed significantly between the IAS and DAS (P<0.05) (Table 2).

The average density and biomass of macroinvertebrates in the RS were 212.50±179.51 ind.m^{-2} and 180.39±206.87 g.m^{-2}, respectively. After sand mining perturbation, the reductions of the two community parameters in the DAS were 89.80% and 99.54%, respectively (P<0.01). Moreover, an obvious decline of almost all of the sampled taxonomic taxa except *Novaculina chinensis* and *Sphaerium lacustre* was observed. As for the IAS, the reductions of the macroinvertebrate density and biomass were 78.82% and 27.79%, respectively, although only the reduction in the biomass was statistically significant (P<0.01). In addition, obvious lower densities of *Nephtys oligobranchia* and *Corbicula fluminea* were found compared with those in the RS. In contrast, *Grandidierella* sp. and *S. lacustre* exhibited obvious increased abundances. In terms of biodiversity, insignificant differences (P>0.05) of Pielou's evenness index were tested between the RS, and sand mining affected sites (DAS and IAS). However, considerably lower values of the Shannon-Wiener index were observed in the DAS compared with those in the RS (P<0.01) (Table 3 and Table 4).

With respect to biological traits, much lower values of almost all of the selected biological traits were observed in the DAS compared with those in the RS. In terms of the IAS, compared with counterparts in the RS, significantly higher densities (P<0.05) of small-sized (0.5-2 cm), short-lived

(≤1 year) macroinvertebrates and shredders were observed. In contrast, significantly lower densities (P<0.05) of macroinvertebrates with larger body size (2-8 cm), longer life cycle duration, burrowers and filter feeders were found (Table 5).

3.3 Indicators of sand mining perturbation in Lake Hongze

C. fluminea and Bivalvia were found to be significant indicators of ecological conditions of the RS (P<0.05). In contrast, *Grandidierella* sp. and Crustacea were typically associated with the ecological conditions of the IAS (P<0.01). Correspondingly, filter feeders and shredders were indicators of the RS and IAS, respectively (P<0.01). In addition, small-sized macroinvertebrate with body size between 0.5-1 cm and short-lived macroinvertebrate (≤1 year) preferred the conditions of the IAS (P<0.05). However, large-sized (2-8 cm) and long-lived (>1 year) species were found to be significant indicators of the RS (P<0.01). In terms of locomotion and substrate relation, the individuals with burrower behavior favored the habitat of the RS (P<0.01) (Table 6).

4. Discussion

4.1 Response of macroinvertebrates to indiscriminate sand mining

The benthic macroinvertebrate assemblages in the offshore area of Lake Hongze were not abundant or diverse prior to indiscriminate sand mining activities due to the homogeneous habitat and strong wind-wave disturbance (Cai et al., 2016). However, indiscriminate sand mining activities resulted in a nearly complete loss of macroinvertebrate assemblages and all of their associated biological traits in the sand mining area. For one, this may be because the direct sand extraction can cause a massive mortality of original benthic animals. For another, dredging-induced habitat degradation in sand mining areas may hamper the recolonization of macroinvertebrates. As we know, the dredging-induced increase in water turbidity and water depth can seriously decrease the light availability of benthic algae, thus inhibiting the biomass of these microphytes (Newell et al., 2005). On the other hand, the intensive sedimentation during the sand mining process may form new surface sediment with a significantly lower content of nutrients and organic detritus. These results indicate that macroinvertebrates could hardly assimilate energy from the sediment. On the other hand, dredging-induced high turbidity conditions may also significantly reduce the survival of filter feeders (e.g., Bivalvia) by restricting their feeding behaviors (Österling et al., 2007, Bucci et al., 2008, Österling et al., 2010), implying that energy assimilation of benthic animals from the water column was also hindered. It is possible that the dredging-related increase in water depth may be associated with the decreased dissolved oxygen in the sediment-water interface, inhibiting the survival of oxygen-sensitive organisms, such as *C. fluminea*, which are generally sensitive to oxygen depletion in lakes (Ilarri et al., 2011).

In terms of the area indirectly affected by sand mining activities, some obvious changes in both the taxonomic assemblages and biological traits were also observed. Our results indicated that the ecological conditions of indirectly affected areas may promote the dominance of *Grandidierella* sp. To our knowledge, the energy assimilation of this species is mainly derived from sediments, presenting feeding habits of shredders and deposit feeders (Schmidt-Kloiber and Hering, 2015). Thus, the dredging-induced high turbidity in the water column may exert limited effects on their foraging activity. On the other hand, the study of marine macroinvertebrates implied that dredging activities significantly promote the sedimentation process and that mobile species are generally less vulnerable than sessile taxa to the sedimentation, as they are able to move to areas with less sediment accumulation or be more efficient in removing particles (Fraser et al., 2017). In fact, the increased densities of *Paranthura* sp. and *Corophium* sp., which shared the same biological traits with

Grandidierella sp. were also observed in the IAS. As a result, the increased abundance of Crustacea and its associated biological traits (e.g., crawlers and shredders) were found in the IAS.

In contrast, our results indicate that dredging-induced high turbidity probably inhibited the survival of *C. fluminea*. As we know, *C. fluminea* has been observed to withstand turbid conditions (Sousa et al., 2008; Foe and Knight, 1985). However, the field observations also demonstrated that this species experienced rapid die-offs triggered by the high silt loads during spring floods (Ilarri et al., 2011). Therefore, the dredging-induced high turbidity conditions in our study may exceed the tolerance range of *C. fluminea*, resulting in its strong decline. Coincidentally, the total absence of *C. fluminea* was observed in Lake Luoma, a medium-sized lake near Lake Hongze, after several indiscriminate sand mining perturbations (Zou et al., 2017). However, the increased abundance of fingernail-sized bivalves (e.g., *S. lacustre*) was observed in the IAS. Compared with *C. fluminea*, *S. lacustre* exhibits a shorter life cycle duration and much smaller body size, which suggests that *S. lacustre* may be relatively an r-strategist in the continuum of strategies r-k (Barbosa, 1977; Schmidt-Kloiber and Hering, 2015). These traits may allow rapid population recovery of *S. lacustre* after reductions in dredging-induced disturbances. The increase of density of *S. lacustre* was altogether much lower than the decrease in the density of *C. fluminea*; therefore, the Bivalvia and its associated biological trait (i.e., filter feeder) were typically associated with the ecological conditions in the RS. Meanwhile, the appearance of small-sized and short-lived species (e.g., *Grandidierella* sp. *and S. lacustre*) resulted in the increased abundance of individuals with a 0.5-2 cm body length and short life cycle duration (≤1 year) in the IAS. The reason behind the significantly lower densities of *N. oligobranchia* observed in the IAS may also be related to food restrictions and feeding habits. This species exhibited a 25% affinity for filter feeding, which implies that the high water turbidity may cause negative effects on their survival (Österling et al., 2007, Österling et al. 2010). In regard to substrate relation, *N. oligobranchia* is dominantly a burrower. Generally, the low ability to escape sediment disturbance of burrowers may explain the decline of *N. oligobranchia* in the sites directly affected by sand dredging.

4.2 Implications for management

Macroinvertebrates are an important food source for organisms at higher trophic levels (Covich et al., 1999), and some species can also be used as human food. With the estimation of secondary production based on the earlier P/B-ratios of macroinvertebrates (Yan et al., 1987), we evaluated that the macroinvertebrate secondary productivity dropped by 99.5% (from 360.17 $g.(m^2.a)^{-1}$ to 1.66 $g.(m^2.a)^{-1}$) in the DAS and by 78.9% (from 360.17 $g.(m^2.a)^{-1}$ to 76.12 $g.(m^2.a)^{-1}$) in the IAS of Lake Hongze. Therefore, indiscriminate sand mining may cause considerable adverse impacts on fisheries production and more general lake ecosystem service. The changes observed, including a decrease of macroinvertebrate body size, a shortening of life cycle duration, a decreased density and a secondary production of macroinvertebrates, indicated that indiscriminate sand mining impaired the ecosystem health of Lake Hongze (Odum, 1985; Rapport and Whitford 1999). However, commercial sand mining is very popular in inland waters of China (Leeuw et al., 2010; Meng et al., 2018) and other countries (Mayekar, 2006) because sand is an indispensable resource for urbanized societies. The following suggestions are proposed for preventing the irrational utilization of sand resources and its associated adverse effect on ecosystem health.

First, we suggest that extremely strict management regulations seem to be the premise of preventing such indiscriminate operations and associated ecological damage. In Lake Hongze, a series of policies were launched by the local government to restrict such activities starting in 2014, such as levying huge fines to offenders and dismantling their sand dredging equipment. However, indiscriminate sand mining always resurged (Duan et al., 2018). The reason is that the large profits (in Lake Hongze, approximately 20,000–30,000 ￥of the net profits for one multi-ton sand dredger in 10 h) of indiscriminate sand mining make offenders less concerned about the financial punishment. The final stop of these indiscriminate activities was followed by harsh punishment (i.e., the arrest and sentence to some offenders). Therefore, for the same reasons, we believe that the unsustainable utilization of sand resources may also occur in areas where no strict regulations are issued.

Contrary to our observations, the opposite response of macroinvertebrates (e.g., increased density and biomass of macroinvertebrates after sand mining) to sand mining perturbations in an indirectly affected area of another large freshwater lake (Lake Dongting) was observed (Meng et al., 2018). We assume that the effect of dredging-induced high turbidity on benthic invertebrates may take a longer time to be fulfilled, whereas the effects of sand mining in Lake Dongting were reported only after one year (Meng et al., 2018). However, the longer temporal duration (from 2012 to 2016) of sand mining in our study may explain the stronger responses of the benthic macroinvertebrates. Furthermore, the higher flushing rate, larger surface area and larger volume of Lake Dongting can provide a better self-purification capacity compared to Lake Hongze (Cai et al., 2017), and the larvae of macroinvertebrates (e.g., *C. fluminea*) can also be recolonized more frequently by upstream runoff. Therefore, in the area adjacent to sand mining activities, the responses of macroinvertebrates to sand mining activities and other dredging-induced effects may vary greatly with the severity and duration of the dredging operation, as well as the self-purification capacity of the water body. Therefore, these features should be taken into consideration in the management of sand mining activities, such as the assessment of ecologically sustainable maximum sand yields.

In addition, we have identified a number of indicators of sand mining perturbations in Lake Hongze. We suggest the general application of these indicator may need to be further validated, and the trait-based indicators are likely more useful at broader geographic scales. However, our results indicated that traditional biodiversity indices, which has been substantially used in the ecological monitoring and assessment of aquatic systems, may be less suitable for biomonitoring of sand mining perturbations. In Lake Hongze, sand mining activities significantly decreased the densities of the original predominant species (i.e., *C. fluminea* and *N. oligobranchia)*, resulting in the increase of evenness in the macroinvertebrate assemblages of IAS. Moreover, the sand mining perturbations seem to have given rise to the appearance of some opportunistic species (e.g., Chironomidae larvae), leading to the increase of species richness in IAS. As a result, higher values of Pielou's evenness index and Shannon-Wiener index were observed. In the DAS, only one species was found in 50% of the sampling occasions, implying that the Pielou's evenness index of these sampling occasions was equal to 1, whereas the value of this index was 0.77 in the RS with higher species richness. These results highlight the importance of a bioindicator-based approach to biomonitoring of sand mining disturbance, rather than the use of a simple diversity indices approach in which species identity plays a limited part (Cousins, 1991).

5. Conclusion

In our study, we observed the collapse of macroinvertebrates due to the considerable habitat degradation in sand mining areas. For the areas adjacent to sand mining, dredging activities had also dramatically altered natural macroinvertebrate assemblages. These results indicate that the benthic population is very sensitive to sand mining perturbations; thus, this fauna may serve to help develop metrics for the assessment of dredging-related ecological effects in freshwater lakes. In Lake Hongze, a number of both taxonomic-based and trait-based macroinvertebrate indicators of sand mining perturbations were identified. Given that the responses of macroinvertebrates to sand mining perturbation may vary greatly with the scale of dredging activities and self-purification capacity of the water body, the usage of our recommended indicators at broader geographic scales may need to be further validated. However, these findings may have important implications in future biomonitoring of sand mining perturbations. We suggest that the dietary preference from the water column, the capacity of recolonization and the mobility of the macroinvertebrate should be taken into consideration when obtaining the more general and robust indicators of the sand mining perturbations. Moreover, further research of sand mining effects, such as the response of other aquatic organism groups, the impact on ecosystem functions and the ecosystem recovery capacity, will be required to achieve a sustainable development of sand mining.

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Figure 1

The location of Lake Hongze in China, and the spatial distribution of the eight sampling sites of Lake Hongze in 2016. HZ8 was the sand mining site, and it was directly affected by sand mining. For HZ1-HZ7, no sand mining activities were observed, but they were indirectly affected by dredging-induced high turbidity water.

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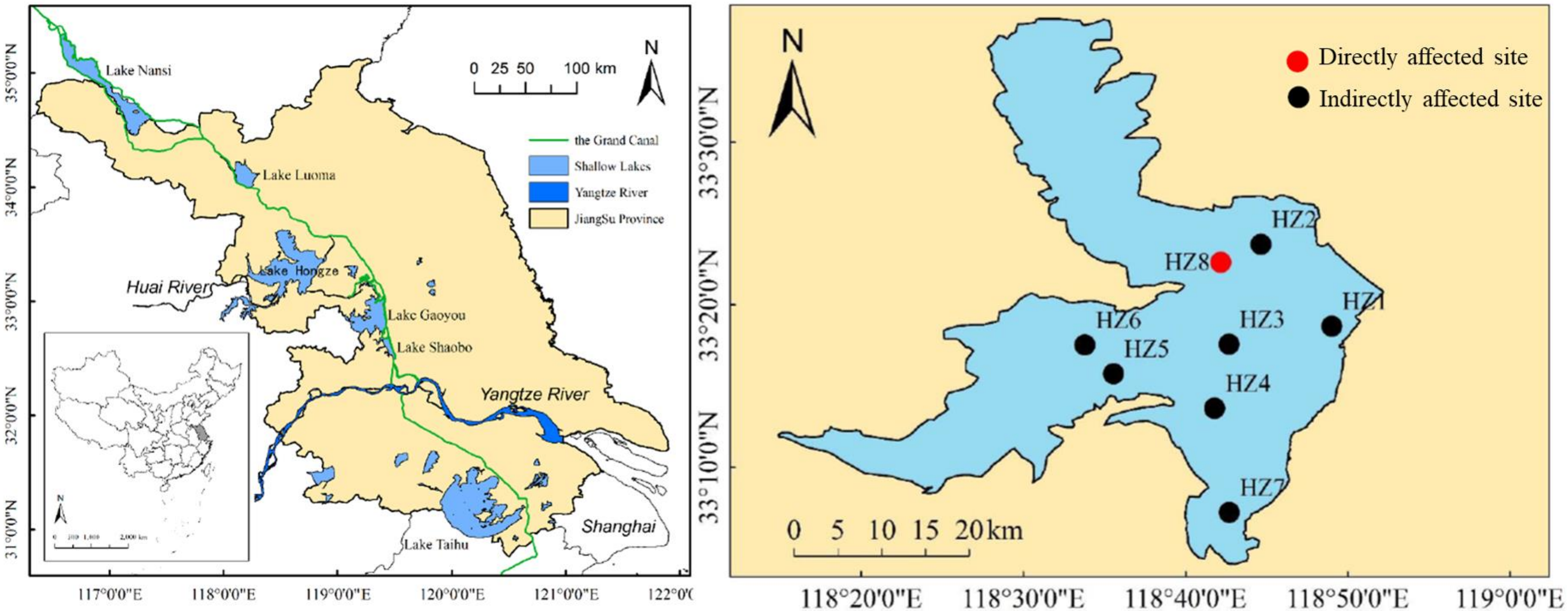

Figure 2

Monthly average concentration of suspended particulate matter (SPM) based on MODIS/Aqua data in the directly affected sites (DAS) and indirectly affected sites (IAS) in Lake Hongze from 2002 to 2016.

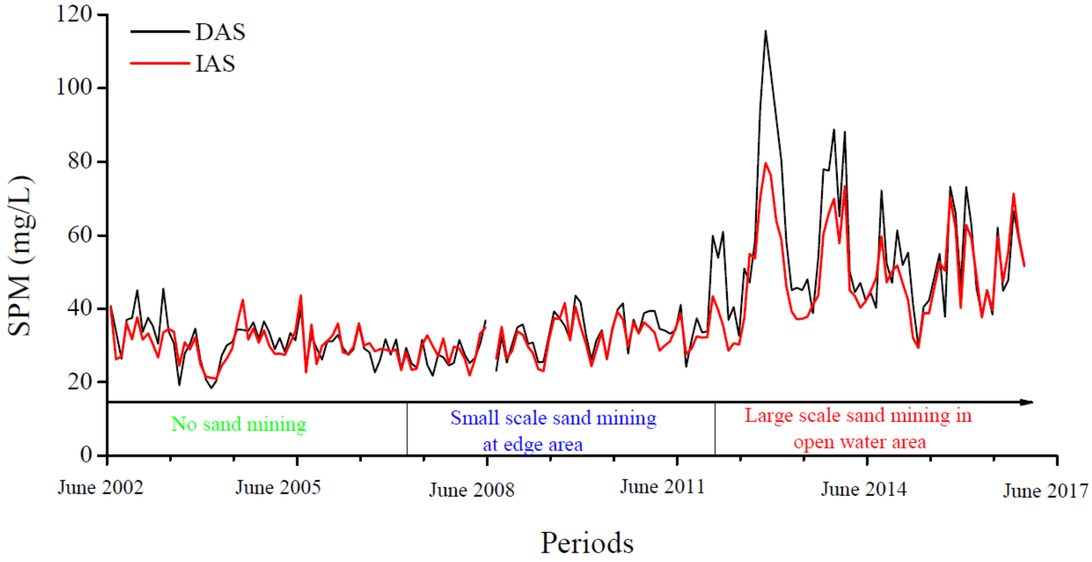

Table 1

Environmental variables of the reference sites (RS) in 2008-2009, directly affected site (DAS) and indirectly affected sites (IAS) in 2016 of Lake Hongze. Different letters indicate significant differences among the groups (one-way ANOVA followed by a post hoc Tukey test or Kruskal-Wallis followed by a Mann-Whitney U test, P<0.05).

Variables	2008-2009 (RS)	2016 (DAS)	2016 (IAS)	*P*
Water temperature (℃)	22.32±5.87^{a}	22.05±6.55^{a}	21.96±4.69^{a}	0.323
Dissolved oxygen (mg/L)	7.97±1.24^{b}	9.26±1.70^{a}	8.86±1.34^{a}	0.024
Electrical conductivity (μs/cm)	450.47±72.83^{b}	525.25±163.58^{a}	550.50±106.53^{a}	0.002
pH	8.77±0.38^{a}	8.99±0.28^{a}	8.72±0.22^{a}	0.279
Secchi depth (cm)	0.25±0.08^{a}	0.18±0.10^{a}	0.13±0.07^{b}	<0.001
Chlorophyll *a* (μg/L)	4.69±3.24^{b}	7.05±3.77^{a}	8.51±6.61^{a}	0.030
Total nitrogen (mg/L)	1.62±0.47^{b}	2.27±0.61^{a}	2.01±0.71^{a}	0.023
Total phosphorus (mg/L)	0.09±0.02^{b}	0.16±0.08^{a}	0.13±0.05^{a}	0.002
Permanganate index (mg/L)	3.98±0.76^{b}	4.99±1.04^{a}	5.12±1.43^{a}	0.002
Total nitrogen of surface sediment (mg/kg)	1.19±0.45^{a}	0.92±0.08^{b}	1.22±0.19^{a}	0.038
Total phosphorus surface sediment (mg/kg)	0.40±0.05^{a}	0.22±0.02^{b}	0.37±0.08^{a}	0.001

Table 2

Results of one-way analysis of similarities (ANOSIM) of the macroinvertebrate taxonomic communities and biological traits groups among the reference sites (RS) in 2008-2009 and the indirectly affected sites (IAS) and directly affected site (DAS) in 2016 of Lake Hongze.

	Comparison	Dissimilarity %	R	*P*
	RS, IAS and DAS	73.52	0.3320	0.0001
Taxonomic	RS and IAS	71.31	0.3798	0.0001
communities	RS and DAS	80.42	0.5485	0.0005
	IAS and DAS	82.12	0.3798	0.0173
Biological	RS, IAS and DAS	61.59	0.1694	0.0003
traits	RS and IAS	58.86	0.1117	0.0036
groups	RS and DAS	74.10	0.5389	0.0024
	IAS and DAS	68.20	0.2180	0.0981

Table 3

Macroinvertebrate total density (ind.m^{-2}), total biomass (g.m^{-2}), Shannon-Wiener and Pielou's evenness indices of the reference sites (RS) in 2008-2009 and the directly affected site (DAS) and indirectly affected sites (IAS) in 2016 of Lake Hongze. Different letters indicate significant differences among the groups (one-way ANOVA followed by a post hoc Tukey test or Kruskal-Wallis followed by a Mann-Whitney U test, P<0.05).

Indices	RS (n=28)	IAS (n=28）	DAS (n=4)	*P*
Total density	212.50±179.51^{a}	153.45±177.16^{ab}	21.67±16.67^{b}	0.003
Total biomass	180.39±206.87^{a}	38.21±90.39^{b}	0.83±1.14^{c}	<0.001
Shannon-Wiener index	0.63±0.41^{a}	0.95±0.45^{a}	0.14±0.28^{b}	0.001
Pielou's evenness index	0.77±0.16^{a}	0.86±0.13^{a}	0.72±0.48^{a}	0.12

Table 4

Mean density (ind.m^{-2}) and biomass (g.m^{-2}) for the 18 species collected in the reference sites (RS) in 2008-2009 and the directly affected site (DAS) and indirectly affected sites (IAS) in 2016 of Lake Hongze. Occurrence (Occ.) is the number of stations at which each species was collected. The number of total sampling occasions for the RS, DAS and IAS were 28, 28 and 4, respectively.

	Density (Occ.)			Biomass		
Taxa	RS	IAS	DAS	RS	IAS	DAS
Oligochaeta						
Branchiura sowerbyi Beddard (1892)	9.32 ±13.62(12)	7.85 ±19.88(5)	0.00 ±0.00(0)	0.20 ±0.50	0.19 ±0.62	0.00 ±0.00
Polychaeta						
Nephtys oligobranchia Southern (1921)	111.29 ±126.49(23)	66.67 ±132.73(21)	15.00 ±19.15(2)	0.59 ±0.58	0.35 ±0.90	0.10 ±0.12
Nereis japonica Izuka (1908)	0.00 ±0.00(0)	1.66 ±7.62(2)	0.00 ±0.00(0)	0.00 ±0.00	0.01 ±0.09	0.00 ±0.00
Notomastus latericeus Sars (1851)	10.86 ±24.27(8)	25.00 ±35.35(18)	0.00 ±0.00(0)	0.16 ±0.48	0.41 ±0.60	0.00 ±0.00
Crustacea						
Paranthura sp.	0.00 ±0.00(0)	6.90 ±12.10(9)	0.00 ±0.00(0)	0.00 ±0.00	0.08 ±0.15	0.00 ±0.00
Corophium sp.	0.57 ±3.02(1)	3.57 ±12.37(3)	0.00 ±0.00(0)	0.00 ±0.00	0.01 ±0.06	0.00 ±0.00
Grandidierella sp.	1.14 ±4.19(2)	17.74 ±40.81(11)	0.00 ±0.00(0)	0.00 ±0.00	0.06 ±0.14	0.00 ±0.00
Chironomidae						
Chironomus plumosus Linnaenus (1758)	0.57 ±3.02(1)	0.23 ±1.26(1)	0.00 ±0.00(0)	0.00 ±0.00	0.00 ±0.00	0.00 ±0.00
Glyptotendipes pallens Meigen (1804)	0.00 ±0.00(0)	0.71 ±3.78(1)	0.00 ±0.00(0)	0.00 ±0.00	0.00 ±0.00	0.00 ±0.00
Dicrotendipes pelochloris Kieffer (1912)	0.00 ±0.00(0)	0.71 ±3.78(1)	0.00 ±0.00(0)	0.00 ±0.00	0.00 ±0.00	0.00 ±0.00
Bivalvia						
Corbicula fluminea Müller (1774)	72.07 ±96.99(26)	5.00 ±11.71(5)	0.00 ±0.00(0)	160.50 ±195.80	20.77 ±52.00	0.00 ±0.10
Sphaerium lacustre Müller (1774)	1.14 ±6.04(1)	10.48 ±14.78(11)	6.66 ±9.42(1)	0.01 ±0.08	2.33 ±7.01	0.63 ±1.11
Novaculina chinensis Liu et Zhang (1979)	0.57 ±3.02(1)	1.19 ±3.17(4)	5.00 ±10.00(1)	0.01 ±0.01	0.23 ±0.87	0.14 ±0.28

Limnoperna fortunei Dunker (1857)	6.28±21.48(5)	3.57±15.44(2)	0.00±0.00(0)	1.09±4.48	1.37±7.22	0.00±0.10
Unio douglasiae Gray (1833)	0.00±0.00(0)	0.71±3.78(1)	0.00±0.00(0)	0.00±0.00	15.60±82.556	0.00±0.00
Lamprotula laleci Gray (1835)	0.57±3.02(1)	0.00±0.00(0)	0.00±0.00(0)	17.15±90.776	0.00±0.00	0.00±0.00
Gastropoda						
Stenothyra glabra Adams (1850)	0.00±0.00(0)	1.42±7.56(1)	0.00±0.00(0)	0.00±0.00	0.29±1.55	0.00±0.00
Semisulcospira cancellata Benson (1833)	1.14±4.19(2)	0.00±0.00(0)	0.00±0.00(0)	0.60±2.23	0.00±0.00	0.00±0.00

Table 5

Density (ind.m^{-2}) of macroinvertebrates belonging to the four traits in the reference sites (RS) in 2008-2009 and the directly affected site (DAS) and indirectly affected sites (IAS) in 2016 of Lake Hongze. Different letters indicate significant differences among the groups (one-way ANOVA followed by a post hoc Tukey test or Kruskal-Wallis followed by a Mann-Whitney U test, P<0.05).

		RS (n=28)	IAS (n=28)	DAS (n=28)	*P*
Body size	0.5-1	1.71±5.04^{b}	22.55±42.03^{a}	0.00±0.00^{b}	0.002
(cm)	1-2	2.94±7.73^{b}	17.17±19.57^{a}	5.00±7.07^{ab}	0.007
	2-4	113.54±104.81^{a}	71.09±109.18^{b}	16.66±14.33^{b}	0.003
	4-8	94.43±85.45^{a}	39.94±49.57^{b}	5.00±4.08^{b}	<0.001
	>8	2.90±4.24^{a}	2.68±6.01^{a}	0.00±0.00^{a}	0.134
Life cycle duration	≤1	4.60±11.95^{b}	27.55±44.90^{a}	1.67±2.36^{b}	0.002
(year)	>1	210.94±173.15^{a}	125.8±167.61^{ab}	25.00±22.73^{b}	0.003
Locomotion and	skaters	0.16±0.60^{a}	0.44±1.9^{a}	0.00±0.00^{a}	0.86
substrate relation	divers	37.98±32.39^{a}	24.89±32.61^{ab}	3.83±3.28^{b}	0.002
	crawlers	15.02±19.74^{a}	21.59±27.59^{a}	1.79±1.57^{a}	0.136
	burrowers	136.94±112.92^{a}	81.44±98.65^{b}	16.14±14.61^{b}	0.003
	interstitial	24.65±25.86^{a}	21.69±35.25^{a}	3.00±3.83^{b}	0.033
	temporarily attached	6.79±1.99^{a}	3.38±4.32^{ab}	1.90±2.69^{b}	0.031
Feeding habits	absorbers	2.33±3.40^{a}	1.96±4.97^{a}	0.00±0.00^{a}	0.121
	deposit feeders	99.93±98.29^{a}	85.72±119.01^{a}	11.25±14.36^{b}	0.031
	shredders	1.35±3.45^{b}	20.06±26.90^{a}	0.00±0.00^{b}	<0.001
	scrapers	0.63±2.11^{a}	1.04±4.54^{a}	0.00±0.00^{a}	0.704
	filter feeders	107.84±110.30^{a}	42.69±45.54^{b}	15.41±12.72^{b}	0.004
	predators	0.06±0.34^{a}	0.19±0.59^{a}	0.00±0.00^{a}	0.481
	parasites	3.39±10.93^{a}	1.79±7.72^{a}	0.00±0.00^{a}	0.378

Table 6

Indicator values (IndVals) index of the ecological condition indicators based on taxonomic taxa and biological traits. The ecological conditions referred to are the reference sites (RS) sampled in 2008-2009 and the indirectly affected sites (IAS) sampled in 2016 of Lake Hongze. In our study, the fidelity (relative occurrence) is the ratio of the number of sites where the indicators are present and the total number of sites in the preferred ecological conditions. Specificity (relative abundance) is the ratio of the mean number of individuals of the indicators across sites of preferred ecological conditions and the sum of the mean numbers of individuals of the indicators in each condition.

	Indicators	Preferred conditions	Fidelity	Specificity	IndVal	Probability
Taxonomic taxa	*Grandidierella* sp.	IAS	0.39	0.94	37 %	0.004
	C. fluminea	RS	0.71	0.94	67 %	0.005
	Crustacea	IAS	0.68	0.94	64 %	0.009
	Bivalvia	RS	0.75	0.74	56 %	0.039
Biological traits	≤1 year	IAS	0.71	0.81	58 %	0.026
	>1 year	RS	0.93	0.63	58 %	0.009
	0.5-1 cm	IAS	0.50	0.93	46 %	0.026
	2-4 cm	RS	0.96	0.58	56 %	0.039
	4-8 cm	RS	0.93	0.70	65 %	0.003
	filter feeders	RS	0.96	0.65	63 %	0.010
	shredders	IAS	0.68	0.94	64 %	0.007
	burrowers	RS	1.00	0.58	58 %	0.008