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The joint effect of tidal barrier construction and freshwater releases on the macrobenthos community in the northern Yellow River Delta (China)

Wei Yang*, Ming Li, Tao Sun, Yuwan Jin

State Key Laboratory of Water Environment Simulation, School of Environment, Beijing Normal University, Beijing 100875, China

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ABSTRACT

Tidal barriers isolate intertidal areas from tides, creating distinct conditions on either side of the barrier, and freshwater releases change the receiving area's hydrology and salinity. However, the combined effect of these human actions is unknown. Using the macrobenthos community as a bioindicator, we sampled part of the northern Yellow River Delta that has been managed using tidal barriers and freshwater releases, in the spring and autumn of 2014, before and after the summer freshwater release. The macrobenthos communities differed greatly on opposite sides of the barrier. During the spring, 7 to 8 species were found inside the barrier (mainly Insecta and Crustacea), versus 22 in the intertidal area (mainly Polychaetes, Mollusca, Crustacea, and Insecta). During the autumn, 10 to 13 species were found inside the barrier and 16 in the intertidal area. Non-metric multidimensional scaling and hierarchical clustering divided the macrobenthos communities into groups that mostly agreed with the spatial distribution of the investigated areas. The characteristics of the modified ecosystems determined the similarity of the macrobenthos communities. The effects of the tidal barriers and the freshwater releases interacted: the barriers decreased sediment salinity compared with that in the intertidal area, and freshwater releases increased this differentiation. To restore the delta's original freshwater ecosystems, the tidal barriers are required to contain the freshwater releases. In addition, the effects of the freshwater releases were generally positive. Therefore, it is reasonable to retain the barriers and to continue or increase these releases.

1. Introduction

Intensive human activities, such as the creation of embankments (e.g., tidal barriers, dikes), construction, and land reclamation can lead to the degradation and loss of coastal wetlands (Bi et al., 2014). Numerous researchers have focused on the resulting changes in the shoreline and associated habitats (Aubanel et al., 1999; Reise, 2005; Vos and van Kesteren, 2000), variations of soil physicochemical properties (Mora and Burdick, 2013; Tang et al., 2013), vegetation succession (Almeida et al., 2014; Piesschaert et al., 2005), and hydrological conditions (Carol et al., 2014). In addition to these characteristics of the coastal wetlands, turnover of the species that live in coastal watersheds and salt marshes is common, as these species are often highly vulnerable to human activities (Braeckman et al., 2014; Gordon, 1994). The macrobenthos is a particularly important part of the biocenosis in coastal wetlands because these organisms play a key role in transferring energy and materials within the food web (Austen et al., 2002; Covich et al., 1999; Dauvin and Desroy, 2005). Due to their restricted habitat, long life cycle, and direct contact with sediments, the macrobenthos community also provides a sensitive bioindicator of environmental characteristics (Bongers and Ferris, 1999; Covich et al., 2004; Wardle et al., 1995).

In recent years, many researchers have studied the effect of physical barriers (e.g., tidal barriers) or the prevention of reclamation activities on the macrobenthos community in intertidal areas (Koo et al., 2008; Li et al., 2016; Ma et al., 2012; Meire et al., 1994). Meire et al. (1994) suggested that the impact of barrier construction on the macrobenthos community was relatively small, but did not claim that there would be no long-term effect. In contrast, other studies indicated that the structure of the macrobenthos community on the land side of such barriers would be

* Corresponding author. No. 9 Xinjiekouwai St., Haidian District, Beijing, China.
E-mail address: yangwei@bnu.edu.cn (W. Yang).

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severely affected by low salinity after embankment construction (Ge et al., 2005; Huang et al., 2011; Koo et al., 2008). Koo et al. (2008) found that the sedimentary environment changed drastically due to the reduction of tidal currents after closing of a dyke, and noted that this increased the differences between the macrobenthos assemblages in the habitats on opposite sides of the barrier. Huang et al. (2011) found that human activities (such as embankment construction and reclamation) changed the habitat characteristics, resulting in a simpler community structure and a low community similarity index for communities on opposite sides of a barrier.

Based on the abovementioned research, wetland managers have become increasingly concerned about the impacts of ecological restoration projects (e.g., freshwater releases, vegetation replanting, and embankment removal) in degraded coastal wetlands (e.g., Cui et al., 2009). In particular, the effects of freshwater releases on the macrobenthos community have raised extensive concerns (Dukowska et al., 2007; Hose et al., 2007; Rolston and Dittmann, 2009). Freshwater is an essential resource for maintaining coastal wetlands, as it both alters the salinity and carries an important load of sediment, which contains important nutrients such as organic matter (Naiman and Dudgeon, 2011; Rolls et al., 2012; Shafroth et al., 2010). Cui et al. (2009) have monitored the ecological response to wetland restoration by freshwater releases in China's Yellow River Delta and found that the restoration project has had positive effects on the wetland ecosystem over the past 7 years. Hernández-Arana and Ameneyro-Angeles (2011) found a significant increase in species richness at locations adjacent to an artificial channel that carried freshwater and suspended sediments into the adjacent wetland.

However, it remains unclear how these restoration projects influenced recovery of the typical original biocenosis. Habitat change typically results from a combination of the freshwater releases and related hydraulic engineering activities (Cañedo and Rieradevall, 2010; Cui et al., 2009; Valipour, 2013). The combination of freshwater releases with tidal barriers will also cause hydrological and salinity conditions to fluctuate in a manner that differs from natural fluctuations (Valipour, 2012). For example, the managed hydrologic cycle often has a much longer period (1 year rather than 1 season; Valipour, 2015), and rather than the original two cycles per day caused by semi-diurnal tides, sediment inputs and salinity would vary more drastically.

In this study, we used field surveys in China's Yellow River Delta to fill gaps in our knowledge of the ecological responses of the macrobenthos community to the joint effect of tidal barriers and freshwater release, and to clarify how the restoration project influenced recovery of the delta's typical freshwater biocenosis. These responses include spatial variation of the macrobenthos community structure and related environmental characteristics. The results will help wetland managers to assess the combined effects of the tidal barriers and freshwater releases and improve their management of such wetlands.

2. Materials and methods

2.1. Study area

The Yellow River Delta Wetlands (37°40′N to 38°10′N, 118°41′E to 119°16′E), one of China's key national nature reserves, is located on the western coast of the Bohai Sea (Fig. 1). The Yiqianer Management Station is located in the northern part of the Yellow River Delta National Nature Reserve, in the eastern part of the old Yellow River estuary. This region has a temperate, semi-humid, continental monsoon climate. Its average annual temperature is 12.1 °C, with the highest monthly mean temperature (27.3 °C) in July. The annual precipitation averages 552 mm, of which 70% falls during the summer, from May to July (Yang et al., 2013). The annual mean pan evaporation averages 1962 mm. Because of excessive withdrawals of water in upstream regions, the study region has no freshwater inflows and its original freshwater ecosystems have been adversely affected by seawater ingress and erosion of the coast, and the problem was exacerbated by relocation of the Yellow River's mouth in 1976 (Li and Wang, 2003). To prevent further wetland damage and restore the area's original freshwater wetlands, managers constructed tidal barriers in 2001 that also provided more favorable conditions for the implementation of environmental flow releases. Since 2010, freshwater releases have been carried out.

The tidal barriers in the study area run from east to west, roughly parallel to the coastline, and divide the intertidal zone into two separate areas: an area affected by the freshwater releases, and the intertidal area between the last tidal barrier and the sea (Fig. 1). These areas reflect different stages of the freshwater release project. From land to sea, the management area is further subdivided into areas I, II, III, and IV, each of which is completely separated from adjacent areas by the barriers. Areas I and II have received freshwater releases from the Yellow River since July 2010. Each release lasted for 19–25 days, and involved the transfer of 0.575×10^7–3.17×10^7 m^3 of water into these areas. The boundary between areas I and II is near the position that the high tides originally reached before construction of the tidal barriers. The second barrier is between areas II and III, and is located ca. 1.5 km outside the intertidal area. The third barrier is between areas III and IV, and is located ca. 0.5 km inside the intertidal area. Freshwater releases have been carried out in area III since 2012. Area IV is an undisturbed intertidal area. Thus, areas I to III represent the ecological restoration zone and area IV represents the reference area.

2.2. Field sample sites and study design

To account for the annual freshwater releases, which are conducted in July, the field study was carried out in the spring (from April to May, before the releases) and the autumn (from September to October, after the releases) in 2014. Macrobenthos and sediments were sampled along south-to-north transects at 18 sites (as shown in Fig. 1).

At each sample site, we obtained three surface sediment cores (to a depth of 30 cm below the surface, which equaled the depth of the macrobenthos sampling) using a core sampler with a diameter of 5.0 cm and a depth of 2.54 cm for the measurement of sediment salinity and pH, and pooled the samples to create a single composite sample for each site.

In the laboratory, the samples were weighed, then the water content was determined by oven-drying part of the sample at 105 ± 5 °C for 2 h. Then a mortar and pestle was used to grind the sediment sample to a grain size of less than 0.83 mm. Next, the sediment samples were weighed and mixed with sterile distilled water at a ratio of 1:5 w/w and the resulting solution was then shaken on a rotary shaker (HZQ-F160, Wanhua, Jintan, China) for more than 30 min. Then the salinity and pH were measured using an HQ 30D portable multi-parameter water quality monitor (Hach, Loveland, Colorado, USA). We also weighed about 2 g of the air-dried sediments from each sample and put them in a muffle furnace (SG-XL1100, SIOM, Shanghai, China) at 550 °C for 5 h to calculate the loss of mass on ignition, which represented the total organic carbon content. We measured the sediment grain-size distribution by using a laser particle analyzer (HELOS-CUVETTE, Sympatec GmbH, Clausthal-Zellerfeld, Germany); from this data, the median grain size was calculated.

We also obtained three macrobenthos samples at each site on

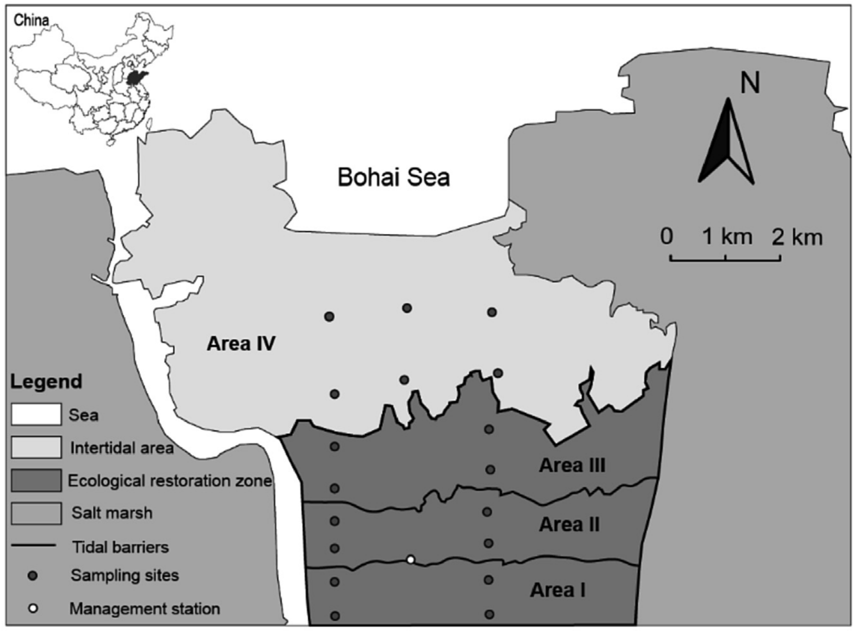

Fig. 1. Map of the four parts of the Yiqianer Management Station Area and the associated sample sites in the northern Yellow River Delta. The original high tide line is near the barrier between areas I and II.

each sampling date, and each site in a given area was separated from the closest other site in that area by at least 500 m. The samples were collected using a 0.1 $m^2 \times 0.3$ m dredge (an opening 0.33 m wide by 0.3 m tall, and a depth of 0.3 m; Ma et al., 2012), and were pooled to create a single composite sample. No macrobenthos occurred in the spring samples at sites I-3 and I-4 and in area III because the sediments had largely dried out. In the field, macrobenthos samples were preserved in a solution of brackish water plus formalin (CSOA, 2007). They were then washed in the laboratory with sea water, and passed through a 0.5-mm sieve to remove the macrobenthos. All collected organisms (except one specimen from phylum Chordata) were identified to at least the genus or class level and weighed to provide the fresh weight of the organism.

2.3. Data analysis

2.3.1. Dominance and biodiversity of the macrobenthos

The biomass and density of the macrobenthos at each site were calculated by dividing the total mass and the total number of the macrobenthos (respectively) by the total sampling area at the site.

The relative dominance of each species (Y) was calculated as follows:

$$Y = (n_j/N)f_j \tag{1}$$

where n_j is the number of individuals of species j; N is the total number of macrobenthos individuals; and f_j is the percentage of all sampling sites where species j was found. When $Y \geq 0.02$, the species is regarded as being one of the dominant species (Shen et al., 2010).

The biodiversity was measured using four indices: the Shannon-Weiner index (H'; Magurran, 1988), Pielou's evenness index (J; Pielou, 1975), Simpson's diversity index (D; Magurran, 1988), and Margalef's richness index (d; Margalef, 1968).

2.3.2. Statistical tests

One-way analysis of variance (ANOVA) was used to judge whether the sediment characteristics differed significantly among the four study areas. Where the data revealed significant heteroskedasticity, the Welch test was instead used. When these test results were significant, *post hoc* multiple-comparison tests (Tukey's HSD) were used to detect sediment characteristics that differed significantly between pairs of areas. Independent-sample t-tests were used to detect significant differences in sediment physicochemical properties between the spring and autumn samples. These analyses were performed using version 18.0 of the SPSS statistical software (http://www.spss.com.cn/).

2.3.3. Non-metric multidimensional scaling and cluster analysis

Non-metric multidimensional scaling (NMDS) was used to describe the degree of similarity of the macrobenthos communities in a two-dimensional graph (Clarke and Green, 1988). In this graph, the similarity of two macrobenthos communities at different sites depends on the distance between the points that represent the macrobenthos communities at these sites (i.e., shorter distances represent greater similarity). We performed NMDS using version 5.0 of the Canoco statistical software (http://www.canoco5.com/). To confirm the results of the NMDS, we also performed hierarchical cluster analysis to produce a dendrogram showing the degree of similarity among the macrobenthos communities at the 18 sites. This analysis was performed using the SPSS software. By comparing the results of the cluster analysis and the NMDS, it is possible to understand the differences among the macrobenthos communities between the parts of the ecological restoration zone (areas I to III) and the intertidal zone (area IV).

3. Results

3.1. Spatial variation of sediment properties from areas I to IV

As the environmental factors that influenced the survival of macrobenthos community, we selected salinity, sediment grain size, total organic carbon (TOC), water content, and pH of the sediments as indicators because they reflected the combined effects of the tidal barrier and the freshwater releases on key ecosystem chemical and physical properties (Barendregt and Swarth, 2013). Fig. 2 summarizes the changes in physical and chemical properties of the sediments along the transect from land to sea during the spring and autumn of 2014. (Details of the measurements at the 18 sampling sites are provided in Supplemental Tables S1–S5.)

The sediment pH did not differ significantly between the areas in the spring and autumn (Fig. 2i,ii). In the spring, salinity differed significantly among the four areas ($p < 0.05$; Fig. 2iii,iv and Table S6); sediment salinity was significantly higher in area III and significantly lower in area II than in the other areas. In the autumn, area IV had a significantly higher salinity than the other areas. In the spring, the sediment TOC contents in areas I, II, and III were significantly higher than those in area IV, but there were no significant differences among the areas in the autumn (Fig. 2v,vi). The median sediment grain size did not differ significantly among the four areas in either season (Fig. 2vii,viii). The sediment water content was significantly lower in area III during the spring and in area IV during the autumn (Fig. 2ix,x).

On the one hand, these differences appear to be directly related to the freshwater releases, since the areas that received this water for the longest duration (areas I and II) had significantly lower salinity and higher water content than the area with the shortest duration (area III). In addition, the TOC contents in the areas that received freshwater releases (I, II, and III) were significantly higher than those in the reference area (IV) during the spring; they also had higher TOC in the autumn, but the difference was not significant. On the other hand, the presence of the tidal barrier and the low spring precipitation in the study area would also affect these results.

As a result of the water releases, freshwater wetlands reappeared in area I. In contrast, area III only received freshwater releases starting in 2012, and the area remained dry during the spring. After a 22-day freshwater release (3.17×10^7 m^3) from June to July 2013, areas I, II, and III received no additional freshwater releases for more than 9 months before our spring 2014 sampling.

3.2. Spatial variation in the macrobenthos community biomass and density from areas I to IV

Fig. 3 describes the differences in the biomass and density of the macrobenthos community among the four parts of the study area and between the spring and autumn samples. The biomass was higher in area IV than in the other areas during the spring, but the difference was not significant because of the high variation; however, during the autumn, it was significantly higher in area IV than in the other areas (which did not differ significantly). The macrobenthos density was significantly higher in area II than in the other areas in the spring, but there were no significant differences among the areas in the autumn.

For biomass, the only significant difference between spring and autumn was for area II, which showed a significant decrease from spring to autumn. For density, area II showed a significant decrease from spring to autumn, whereas areas III and IV showed significant increases. However, no organisms were found in spring sampling in area III, and this would have influenced the results.

Fig. 3 shows that the greatest macrobenthos biomass values (14.18 g/m^2 in spring and 10.53 g/m^2 in autumn) were both in area IV. However, the decrease in the macrobenthos biomass was largest (5.67 g/m^2) in area II, and the increase in area III was from 0 in the spring to 0.32 g/m^2 in the autumn. For density, the greatest spring value (1424 ind/m^2) was in area II; in the autumn, the greatest density (624 ind/m^2) occurred in area IV, which also showed the greatest increase (by 523 ind/m^2) from spring to autumn. In area III, the density increased from 0 in the spring to 124 ind/m^2 in the autumn. Some macrobenthos species, such as *Notomastus latericeus* (Polychaetes) and *Cladotanytarsus* sp. (Chironomidae), can survive from the spring to the autumn. Although the density of the macrobenthos community in areas I and II decreased from spring to autumn, the decrease was much larger in area II (from 1424 ind/m^2 to 355 ind/m^2), versus a decrease from 101 ind/m^2 to 55 ind/m^2 in area I. These changes appear to have resulted from the annual freshwater release in July.

3.3. Spatial variation of the macrobenthos community structure from areas I to IV

3.3.1. Dominant species in different areas

Table 1 lists the dominant species in the four areas in the spring and autumn of 2014. In this table, only dominant species (those with $Y \geq 0.02$) are reported. The taxon with the largest number of dominant species in areas I to III was the Insecta; however, the Crustacea had a stronger dominance (Y) and higher proportion of the total biomass in area II. The dominant species with the largest Y was *Chironomus* sp. (Insecta) in area I, *Corophium sinense* (Crustacea) in area II, and *Notomastus latericeus* (Polychaetes) in area III. The longer the ecological restoration engineering had been performed, the higher the proportion of Insecta. In contrast, the intertidal area (area IV) had no dominant species in the Insecta; instead, the dominant species belonged to the Mollusca, Polychaetes, and Crustacea.

As a result of the freshwater releases, freshwater species were relatively stable in area I, and all of the dominant species belonged to the Insecta. In area II, *Corophium sinense* (Crustacea), a euryhaline species, was the dominant species in both seasons, with the largest number of individuals, the highest proportion of total biomass, and the highest Y. Some freshwater species became dominant species in area II, which had originally been part of the more saline intertidal zone. For example, *Cladotanytarsus mancus* (Chironomidae, Insecta) and *Polypedilum* sp. (Chironomidae, Insecta) became dominant species in the autumn, after the freshwater releases. In addition, Polychaetes and Mollusca such as *Neanthes succinea* and *Philine* sp., which are marine or brackish species found in area IV, almost disappeared in the restoration areas. (They were occasionally present, but not often enough to achieve $Y \geq 0.02$; they are therefore not shown in Table 1.) Five dominant species (one in the Polychaetes, one in the Crustacea, and three in the Insecta) were found in area III in the autumn, after the releases. In area IV, species diversity was high in the spring (with five dominant species that had $Y \geq 0.02$), including species from the Polychaetes, Mollusca, and Crustacea; however, only one dominant species (*Corophium sinense*, in the Crustacea) was present in the autumn.

We also found that several species could live in areas with different habitat (sediment) conditions, including *Corophium sinense*, *Chironomus* sp., and *Cladotanytarsus mancus*. *Chironomus* sp. and *Cladotanytarsus mancus* (freshwater species) were most likely to co-occur in areas I and II, although they were dominant in different seasons, and *Corophium sinense* (a euryhaline species) was found in areas II, III, and IV. This suggests a transition from salt marsh to freshwater wetlands in the ecological restoration zone.

3.3.2. Changes in the macrobenthos community structure from areas I to IV

Table 2 shows the differences in the number of species in each class among the areas and the changes from spring to autumn. The total number of species increased from spring to autumn in areas I to III, but decreased in area IV. However, some taxa showed different trends. For example, the number of species in the Insecta decreased between spring and autumn in areas I and IV.

Fig. 4 illustrates the community composition (based on the % of

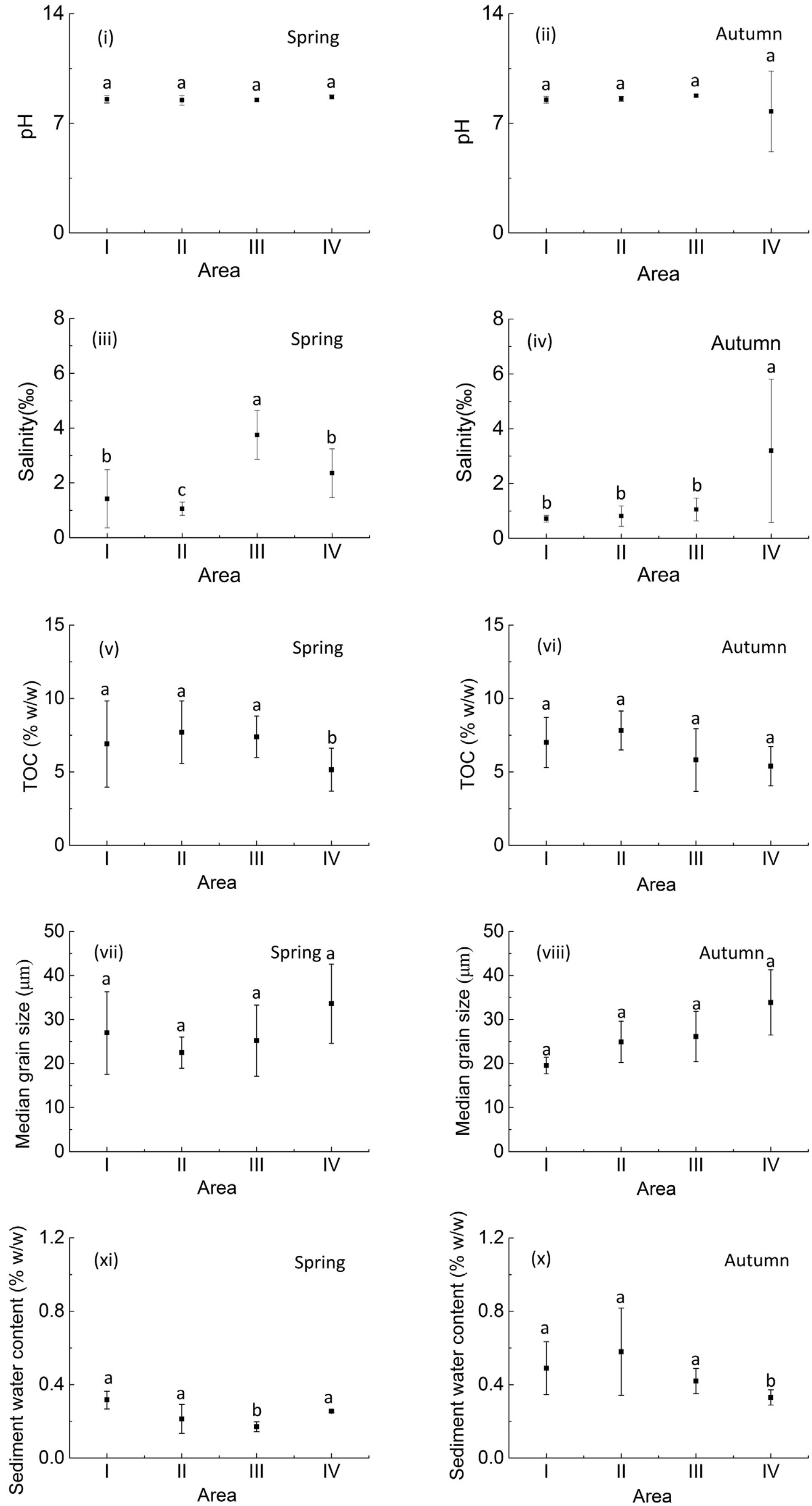

Fig. 2. Variations in physicochemical properties of the sediments (pH, salinity, total organic carbon (TOC) content, median sediment grain size, and sediment water content) along the transect from land (areas I, II, and III) to the intertidal zone (area IV) during the spring and the autumn in 2014. Values are means ± SD; for pH, some SD values are smaller than the symbol used to represent the mean. Values labeled with different letters differed significantly in a given season (ANOVA followed by a *post hoc* multiple-comparison test (Tukey's HSD), $p < 0.05$).

the total biomass) and the macrobenthos structure in the four areas, and reveals the changes along the transect from land to sea. In area I, the macrobenthos community composition changed greatly from spring to autumn: the proportion of Insecta decreased by 69 percentage points, and these species were replaced by Crustacea. In area II, the composition was more stable, but Crustacea increased by about 4 percentage points from spring to autumn and about half of the Insecta were replaced by Polychaetes. No macrobenthos community was observed in area III during the spring, but in the autumn, many Polychaetes appeared (61% of the total), along with Crustacea, Insecta, and Phylum Chordata, seemingly benefited by the inputs of freshwater. In area IV, the macrobenthos community changed slightly; Crustacea remained dominant, at more than 64% of the total, but Mollusca decreased greatly (by nearly 18 percentage points), and were replaced mostly by Polychaetes.

In areas I to III (inside the tidal barriers), Mollusca and Nemertinea disappeared and the number of Polychaetes also decreased greatly compared with their levels in area IV. In their place, Crustacea and Insecta became the main components of the macrobenthos community.

NMDS ordination for the macrobenthos communities at the 18 sample sites during the spring and autumn revealed the similarities and differences among the macrobenthos communities. Fig. 5 shows that both in the spring (Fig. 5i) and in the autumn (Fig. 5ii), the sampling sites were generally divided into groups that coincided with the spatial distribution of the sampling areas. The dendrogram produced by hierarchical cluster analysis generally confirmed this classification (Fig. 6). Some differences between Figs. 5 and 6 are likely to have been caused by the different approaches used in the two methods (i.e., different rules for classification). Figs. 5 and 6 show that sample points that were physically close together also had high similarities during the spring. However, this relationship was less evident in the autumn. For example, the macrobenthos community at some sites in area III (III-3 and III-4) showed distinct differences compared with adjacent areas (areas II and IV).

3.4. Spatial variation of the macrobenthos community biodiversity along the transect from areas I to IV

Fig. 7 summarizes the values of the four biodiversity indices in the four areas in the spring and autumn of 2014. In the intertidal zone (area IV), all four biodiversity indices decreased from spring to autumn. In contrast, all four indices increased from spring to autumn in the other three areas, inside the tidal barriers. The increases were particularly large in area II, where the increase from spring to autumn was more than 150%, and in area III, where H', J, D, and d increased from 0 to 2.8, 0.8, 0.8, and 2.0, respectively. In the ecological restoration zone, the four biodiversity indicators gradually decreased along the transect from land to sea during the spring, but in the autumn, the values of the indicators in area III increased to values close to those for area I. However, the greatest macrobenthos H', J, D, and d values (1.96, 0.71, 0.81, and 2.63, respectively) occurred in the intertidal area (area IV) in the spring, and the lowest values (0.49, 0.13, 0.13, and 1.71, respectively) occurred in area IV in the autumn. The areas inside the tidal barriers clearly benefited from the freshwater releases, leading to the appearance of several species in response to improved habitat characteristics.

4. Discussion

4.1. The rationale for constructing tidal barriers in intertidal areas

The construction of tidal barriers can prevent erosion of coastal wetlands by tides. However, by isolating some parts of the high-tide zone from tides to some degree, tidal barriers change the habitat and alter the structure and composition of the macrobenthos community in these areas. Considering that areas I, II, and III had received no freshwater releases since the previous July (i.e., almost 1 year), the effect of the freshwater releases appears to have weakened as much as possible by the time of the spring sampling in 2014. Thus, the change in the macrobenthos structure from sea to land shows the ability of the barriers to promote changes in the associated ecosystems. Without the ingress of saline seawater, the areas that had been intertidal areas changed into coastal brackish or freshwater marshes. In this study, one key factor (salinity) increased along the transect from land to sea during the spring, but the differences between areas I, II, and III disappeared by the autumn as a result of the freshwater releases; however, salinities in these areas decreased as a result of these releases to levels far below that in area IV. In contrast, the intertidal area showed a relatively stable salinity throughout the year. The areas inside the tidal barriers that received insufficient water (e.g., parts of area I and all of area III in the spring) had significantly higher spring salinity than in the intertidal area (area IV) as a result of evaporation that brought salts to the surface of the sediments. This change made it possible for many oligohaline species, such as *Chironomus* sp. and *Cladotanytarsus mancus* (Lu, 1997), to live in the areas with suitable salinity inside the tidal barrier, which would not have been possible without the changes created by the tidal barrier. The obstruction of the tide by the tidal barriers also created a more stable sediment environment for the areas located inside the barrier. This provided good habitat for many freshwater insect species, such as *Cladotanytarsus mancus*, *Dicrotendipes* sp., and *Polypedilum* sp. (Oliver, 1971), and allowed them to survive in these areas.

Another effect of the change in hydrodynamic conditions inside the tidal barrier is that it created a barrier to the reproduction of many Polychaetes (such as Nereididae). These species are typically widespread in brackish water and seawater areas (Lobo et al., 2016). They tend to emerge from burrows and swim to the surface of the water during their reproductive period (Sun and Yang, 2004). The construction of a tidal barrier eliminates the stable water level inside the tidal barrier that is required for reproduction of these species. This may explain why Polychaetes disappeared in area I inside the tidal barriers. Under the effects of the tidal barrier, the observed changes in the macrobenthos communities resembled the results of Lv et al. (2012) and Ma et al. (2012).

On the other hand, the construction of tidal barriers is necessary to allow the management of freshwater releases. As the freshwater releases increased the sediment water content and decreased the salinity of the zones that received these releases, the ecological restoration zones showed higher autumn biodiversity than the intertidal area. A typical freshwater species structure and composition of the macrobenthos community was established in these areas as a result of the freshwater releases. Therefore, the construction of tidal barriers may promote the ecological restoration of coastal freshwater wetlands in the long term.

4.2. Insights to guide freshwater releases

Freshwater replenishment and seasonal precipitation patterns could obviously change the environmental characteristics in the affected areas. In 2013, a total of 3.17×10^7 m^3 of freshwater was released into areas I, II, and III, versus total annual precipitation of only 0.74×10^7 m^3 in these areas (SBDC, 2014). The seasonal dynamics of water, precipitation, and tides strongly determine species distributions in natural intertidal areas (Bao et al., 2008; Dauvin et al., 2006). However, in the study area, natural precipitation could not counteract the strong spring evaporative demand

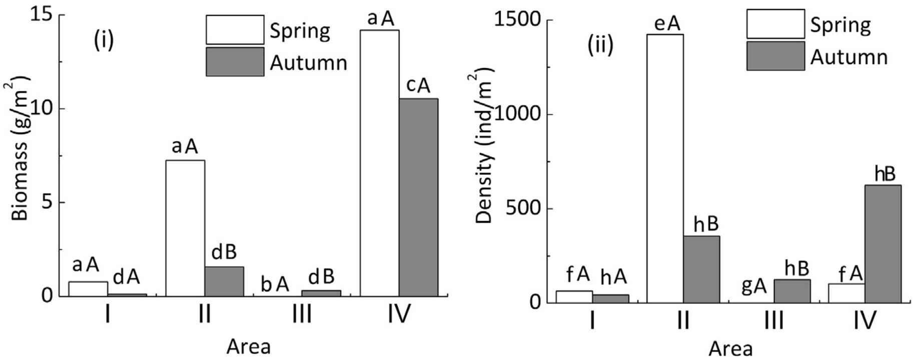

Fig. 3. Variations in the (i) biomass and (ii) density of the macrobenthos community. Bars labeled with different lower-case letters (a, b, c, d) differed significantly between areas in a given season; bars labeled with different capital letters (A, B) differed significantly between seasons at a given site (ANOVA followed by a *post hoc* multiple-comparison test (Tukey's HSD), $p < 0.05$).

Table 1
Dominant species (those with $Y \geq 0.02$) in the four areas along the transect from land (areas I to III) to sea (intertidal area IV) in spring and autumn. Area locations are shown in Fig. 1.

Area	Season	Species name	Class	Number of individuals	Proportion of total biomass (%)	Y
I	Spring	*Chironomus* sp.	Insecta	**215**	**8.45**	**0.72**
		Dicrotendipes	Insecta	22	0.86	0.17
	Autumn	*Cladotanytarsus mancus*	Insecta	**18**	**34.62**	0.17
		Einfeldia sp.	Insecta	14	26.92	**0.27**
		Polypedilum sp.	Insecta	7	13.46	0.07
II	Spring	*Corophium sinense*	Crustacea	**1980**	**77.83**	**0.93**
		Chironomus sp.	Insecta	215	8.45	0.06
	Autumn	*Corophium sinense*	Crustacea	**294**	**0.69**	**0.69**
		Ceratopogonidae	Insecta	33	0.08	0.06
		Cladotanytarsus mancus	Insecta	17	0.04	0.03
		Polypedilum sp.	Insecta	41	0.10	0.07
III	Autumn	*Notomastus latericeus*	Polychaetes	**37**	**0.25**	**0.12**
		Corophium sinense	Crustacea	23	0.15	0.08
		Cladotanytarsus sp.	Insecta	35	0.23	0.06
		Dicrotendipes tritomus	Insecta	25	0.17	0.04
		Sigara substriata	Insecta	13	0.09	0.07
IV	Spring	*Neanthes succinea*	Polychaetes	33	1.30	0.08
		Bivalvia	Mollusca	**87**	**3.42**	0.07
		Moerella jedoensis	Mollusca	42	1.65	0.08
		Philine sp.	Mollusca	61	2.40	**0.12**
		Macrophthalmus japonicus	Crustacea	13	0.51	0.03
	Autumn	*Corophium sinense*	Crustacea	**1050**	**0.93**	**0.62**

Note: Boldfaced numbers represent the largest value in a given season for a given area.

Table 2
Number of species in the main taxa in the four areas along the transect from land (areas I to III) to sea (intertidal area IV) in spring and autumn. Area locations are shown in Fig. 1.

Area	Season	Nemertinea	Polychaetes	Mollusca	Crustacea	Insecta	Phylum Chordata	Total no. of species
I	Spring	0	0	0	0	8	0	8
	Autumn	0	0	0	4	6	0	10
II	Spring	0	1	0	2	4	0	7
	Autumn	0	1	0	3	9	0	13
III	Spring	0	0	0	0	0	0	0
	Autumn	0	2	0	2	6	1	11
IV	Spring	1	10	5	4	2	0	22
	Autumn	1	6	4	5	0	0	16

without the help of plants such as reeds to create a boundary layer that reduced transpiration. Due to the existence of many reeds in area I, some ponds survived until the spring sampling. In contrast, strong evaporation of water from the unprotected surface in area III caused the sediments to mostly dry out and brought salt to the surface, leading to a higher salinity. In contrast, because some freshwater remained in areas I and II, the salinity decreased below the level in the intertidal zone (area IV). After freshwater was released into area III during the spring, the water content of the sediments became significantly higher. However, there was no significant difference in salinity among the three areas that received the freshwater releases. In addition, the sediment salinity in areas I to III was lower in the autumn than in the spring, in contrast with the higher value in area IV. This can be explained by a dilution effect due to the freshwater releases and the higher precipitation before and during the autumn. Although the mean

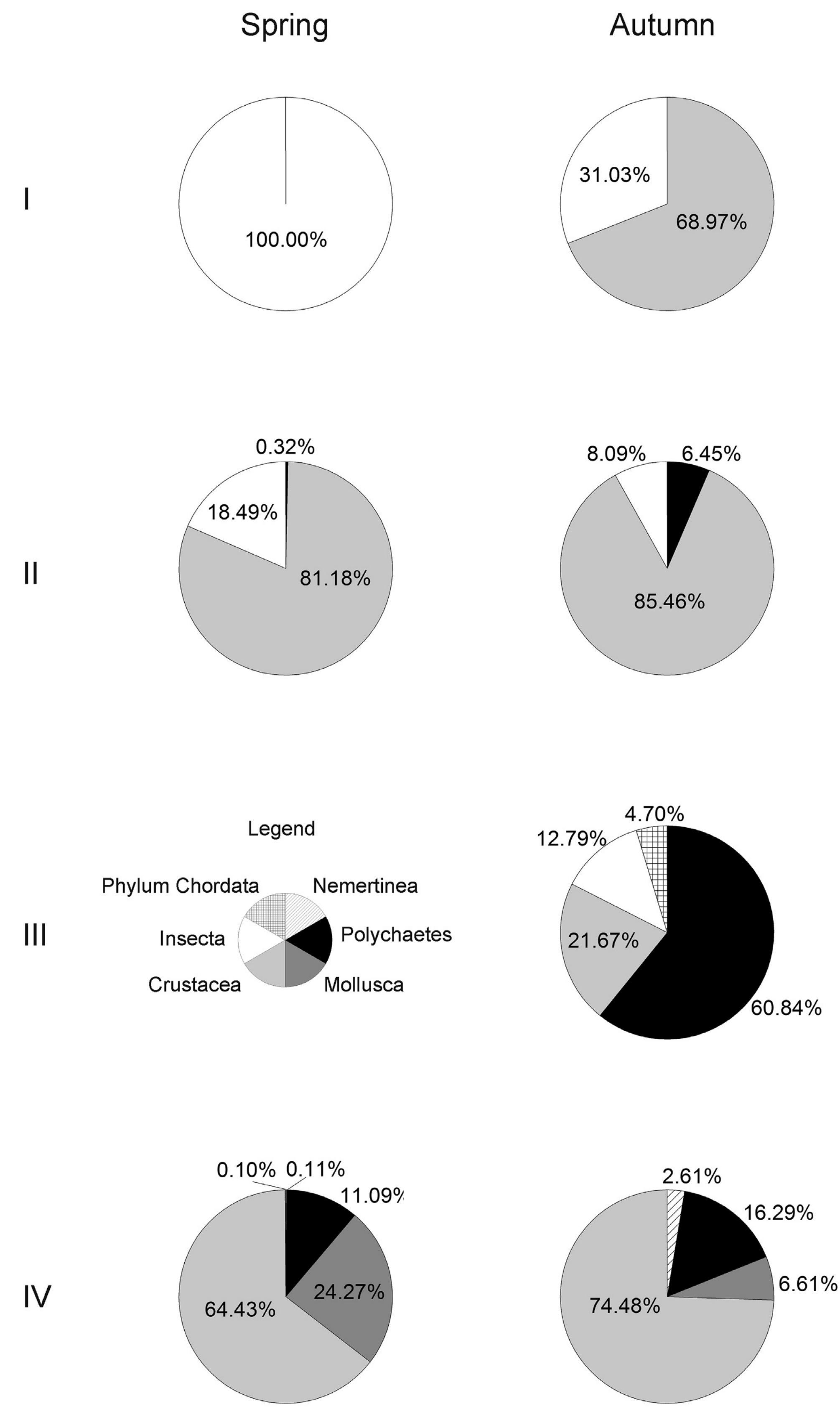

Fig. 4. Proportions of the macrobenthos community structure (% of the total biomass) in the four areas in the spring and autumn (No macrobenthos samples were collected in the spring at sites I-3 and I-4, and in area III, which had dried out completely at that time.).

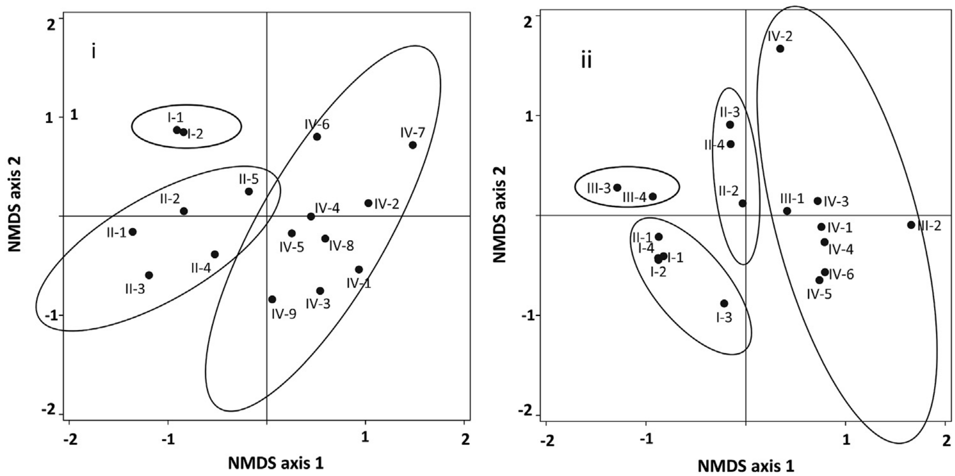

Fig. 5. Non-metric multidimensional scaling (NMDS) ordination for the macrobenthos communities in the four sampling areas during (i) the spring and (ii) the autumn of 2014 (No macrobenthos samples were collected in the spring at sites I-3 and I-4 and in area III, which had dried out completely at that time.).

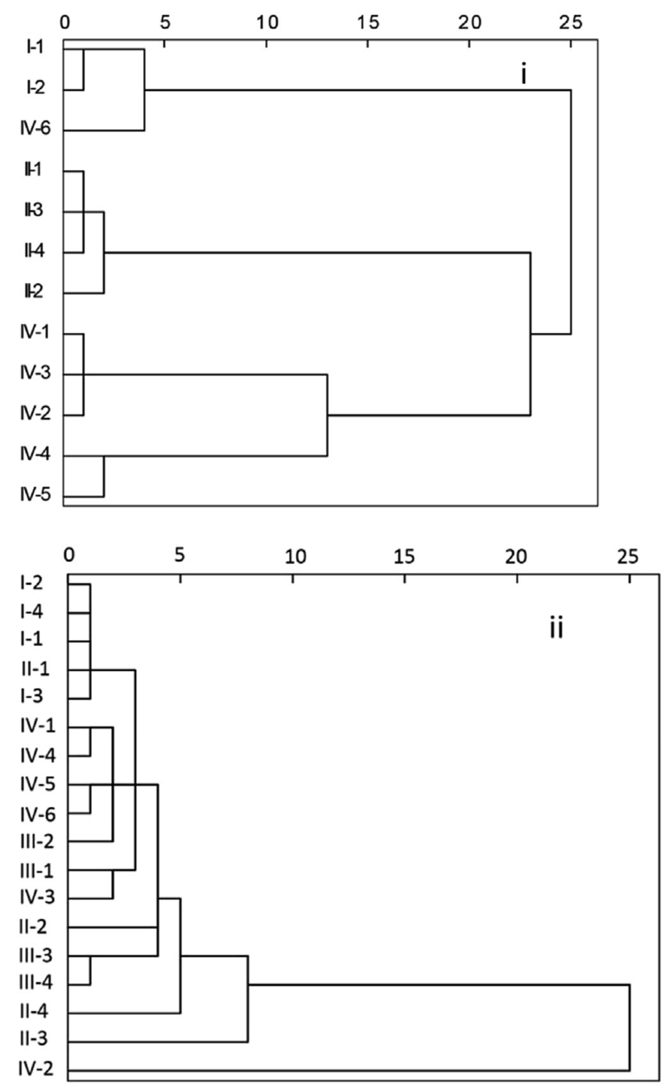

Fig. 6. Hierarchical cluster analysis for the macrobenthos communities in the four sampling areas in the (i) spring and (ii) autumn of 2014. No macrobenthos samples were collected in the spring at sites I-3 and I-4 and in area III, which had dried out completely at that time.

sediment grain size increased and the TOC content decreased along the transect from land to sea, the change was not statistically significant. On the other hand, the sediment water content decreased significantly moving from the areas that received freshwater releases (I, II, III) to the reference area (IV).

By decreasing the salinity of the sediments, improving hydrological connectivity among habitats, and providing a relatively stable environment, the freshwater releases played an important role in the recovery of ecosystems in the study area. Because the sediment salinity changed in response to the freshwater releases, some freshwater species such as *Cladotanytarsus mancus* and *Polypedilum* sp. became the dominant species in areas that had originally been part of the more saline intertidal zone, which provides evidence of the reappearance of a freshwater ecosystem. Our data on changes in the structure of the macrobenthos community in a given area between the spring and autumn clearly showed greater changes in the community structure in the ecological restoration zones than in the intertidal area. Because of the presence of a greater area of open water, biological connectivity among the habitats in areas I to III also increased, whereas Area IV was largely separated from the other areas by the tidal barriers. The large number of *Corophium sinense* individuals in area II in both seasons and in part of area III in the autumn provides some evidence for this. The dominant species were similar in adjacent areas. For example, *Corophium sinense* was the dominant species in area II in the spring, and became a dominant species from area II to area IV in the autumn.

Thus, freshwater releases appear to strongly alter the driving factors responsible for the seasonal changes in the macrobenthos community structure and composition. On the one hand, freshwater replenishment decreased sediment salinity, and oligohaline species appeared in the ecological restoration zones. On the other hand, large-scale freshwater releases may have allowed many species to become established in area III, leading to a higher biodiversity in autumn than in most of the other ecological restoration zones. In autumn, some *Cladotanytarsus* species, including *Cladotanytarsus mancus*, were found in areas I and II, where they had not been found in the spring. One explanation for this is that these species were promoted by the freshwater releases. Another reason may be that some of the organisms were carried into the three areas or among the areas by the flowing water, or in the reverse direction. Given the good performance of area III in terms of the increase in its biodiversity from spring to autumn, reaching

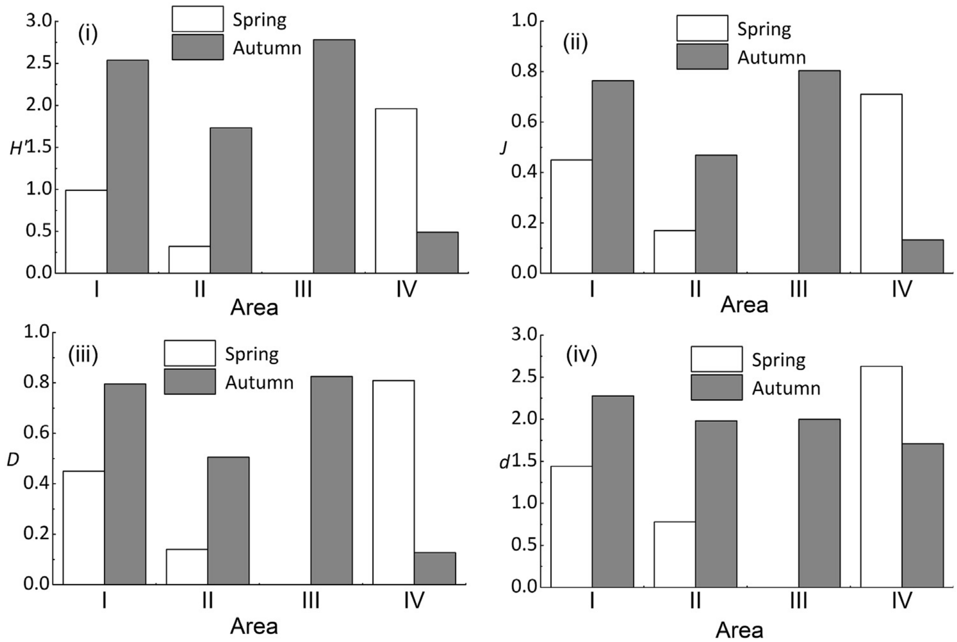

Fig. 7. Changes in the four biodiversity indicators along the transect from land (areas I to III) to sea (area IV) during the spring and autumn of 2014: (i) the Shannon-Weiner index (H'), (ii) the Pielou evenness index (J), (iii) Simpson's diversity index (D), and (iv) Margalef's richness index (d).

levels comparable to those in the other areas, this suggests a need to increase the frequency or magnitude of freshwater replenishment in this zone.

The seasonal changes of the macrobenthos community in areas I to III and their relationship with the timing of the start of the freshwater releases (earlier in areas I and II than in area III) suggest that long-term freshwater replenishment has a potential additive effect: areas I and II showed a more stable structure of their macrobenthos community. Therefore, the effect of freshwater releases appears to be positive, and to increase over time. To avoid drastic seasonal fluctuations in the macrobenthos community, it may be necessary to increase the frequency of the freshwater releases.

4.3. Future research

After the freshwater releases, areas isolated by the tidal barriers turned into coastal freshwater wetlands. The areas that had experienced 4 years of freshwater releases showed characteristics more typical of a freshwater macrobenthos community. A more stable macrobenthos community and higher biodiversity appeared to develop in the ecological restoration zone.

The main source of error in the present research is sampling error, which resulted from the small sample size. The choice of an appropriate sampling intensity is vital to reduce the sampling error, including the choice of representative sample sites (Tagliapietra and Sigovini, 2010), of a sufficient number of replicates (Filippova et al., 2015; Rumohr et al., 2001), and of the reliable and accurate biometric identification (Grizzle, 1984; Tagliapietra and Sigovini, 2010). In future research, we will increase the number of replicates to reduce the sampling error and increase the likelihood of statistically significant results. It will also be necessary to expand the monitoring of water and sediment characteristics to account for other factors (e.g., biological and chemical oxygen demand in the released freshwater) that may affect biodiversity and recovery of the ecosystem.

Construction of the tidal barriers is only the first step in a larger project to restore disturbed coastal wetlands. It provides a good foundation for planning future freshwater releases. The present results focused on the spatial variation of the macrobenthos community along the transect from land to sea, which served as a proxy for time since the releases began. However, the results must be verified over a longer term using a chronological sequence at the same sites. A comparison over time will provide stronger information about the cumulative effect of the freshwater releases. This is particularly true given the different environments throughout the study area. In addition, it will be necessary to examine the effects on other organisms (e.g., plants, microbenthos, birds) to provide a more complete picture. For example, it is possible that the number of predators of macrobenthos species in the study area, such as *Larus saundersi*, will increase (Ge, 2012). If this increase occurs, it will become an important factor for the survival and development of the macrobenthos community. In addition, it will be necessary to study populations of algae in the new habitats, since they act as a main food source for the macrobenthos. We therefore plan to study these aspects of the ecosystem during our future research.

Another interesting possibility will be to examine the possibility of using remote-sensing to monitor the entire study area with greater frequency. To do so, the data collected in the present study and in our future research could be used to calibrate models of sediment properties based on satellite images.

5. Conclusions

In this study, we compared the sediment characteristics and macrobenthos communities along a transect from land (inside the tidal barriers) to sea (outside the barriers), and found that the barriers significantly affected water and sediment quality, leading to the development of distinctly different macrobenthos communities on opposite sides of the barriers. In addition, the macrobenthos communities in the spring and autumn showed distinct spatial differences in our hierarchical clustering and NMDS analyses; most sample sites in a given area were grouped together in both analyses. However, the distinctions among the groups weakened in the autumn, probably due to increased hydrological connectivity among habitats caused by the large inputs of freshwater. In summary, the effects of construction of the tidal barriers and those of the freshwater releases interacted: the barriers caused differentiation of sediment salinity between the intertidal area and the areas inside the barriers, and the freshwater releases increased this difference by the autumn.

Despite the limitations of this study, the effect of freshwater releases on the macrobenthos community appears to have been generally positive. Therefore, it appears reasonable to recommend continuation of the current release program or an increase in the frequency of freshwater replenishment in the Yellow River Delta, although monitoring will be necessary in the long term to confirm the present results.

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Appendix A. Supplementary data

Supplementary data related to this article can be found at http://dx.doi.org/10.1016/j.ocecoaman.2016.11.024.

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