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A solution for restoration of critical wetlands and waterbird habitats in coastal deltaic systems

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A R T I C L E I N F O

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A B S T R A C T

Loss of coastal wetland habitats has been directly linked to a decline in waterbird populations including migratory species, leading to calls to reverse this trend in part by restoring these habitats. However, distinct "sediment scarcity" has hindered coastal habitat restoration. Here, taking the Yangtze River Delta, China as an example, we put forward a feasible solution to solve the sediment shortage in habitat restoration so necessary to restore migratory waterbird numbers. Four biological indices including total wetland area, wetland vegetation area and waterbird species richness and abundance, were used to compare and assess the restorative efforts. Three solutions were adopted for the rehabilitation sites, including promoting sediment deposition and settlement through engineering intervention in Chongming Dongtan (CD) and Eastern Nanhui (EN), and using dredged sediments to nourish and create new habitats in Hengsha Eastern Shoal (HES). The mean wetland area increased 19.66 km^2/yr in EN, 8.78 km^2/yr in HES and 3.83 km^2/yr in CD after rehabilitation. Along with the increase of wetlands and habitats, the abundance of waterbirds increased 1.3 times, 121 times and 1.5 times in EN, HES and CD, respectively. In contrast, in the site of Fengxian and Jinshan (FJ) where no any rehabilitation measure was taken after reclamation, the habitats were lost almost completely and the waterbird abundance dropped drastically. The comparison and assessment results demonstrate that proper coastal silting structures and ecological utilization of nearby dredged sediments are the feasible and effective solutions to retain sediments, restore coastal habitats and increase waterbird diversity and abundance.

1. Introduction

From a global perspective, approximately 23% of human population live within the narrow coastal zone (Small and Nicholls, 2003), while the proportion in China is nearly 44% of the nation's population that contributes about 61% of the national gross domestic product (Wang et al., 2014). High population density and rapid economic development have caused huge demands on marine space and resources in the coastal zones, and consequently, accelerated natural habitat losses and increased ecological risks (Bamford et al., 2008; Ma et al., 2014; Murray et al., 2014). For example, the global tidal flat area shrank by about 16% between 1984 and 2016, while the loss in Chinese coastline exceeded 50% (Murray et al., 2018; Li et al., 2018). This loss of habitats has caused shoreline erosion, the well documented and extraordinary decline of migratory waterbirds (Bamford et al., 2008; Larson, 2015; Studds et al., 2017) and the loss of a wide range of ecosystem services (Murray et al., 2015). Meanwhile, widespread establishment of the invasive cordgrass *Spartina alterniflora* at the expense of the native species has reduced the quality of remaining nesting and feeding habitat for waterbirds (Jackson et al., 2021). Therefore, habitat restoration has become an urgent and important task in coastal zones.

Many ecological solutions have been developed for coastal habitat

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restoration, e.g., engineering restoration, intertidal marshes and mudflat creation, restoration or enhancement and nature-based management (Yozzo et al., 2004; Reynolds et al., 2017; Li et al., 2018). In these solutions, a sufficient supply of sediments plays an essential role in providing substratum, nourishing habitat and supporting coastal plant restoration at the coast (Temmerman and Kirwan, 2015; De Vriend et al., 2015). River damming has reduced 20% of the global sediment inputs to the coasts and estuaries over the last several decades (Syvitski et al., 2005). For example, the sediment inputs in the Nile Delta, the Indus Delta and the Mississippi River Delta have declined by 98%, 94% and 69% respectively, since their first dam was built in the catchment (Syvitski et al., 2009; Giosan et al., 2014). The sediment load of the nine major rivers in China combined has also dropped by 85% over the past 60 years (Wu et al., 2020). "Sediment scarcity" in coasts and estuaries can not only lead to coastal erosion, but also challenge the effectiveness of coastal habitat restoration (Syvitski et al., 2009; Kondolf et al., 2014). To reduce reservoir siltation at river basins and downstream sediment starvation, some countries have adopted a sediment flushing strategy (Juracek, 2015; Hauer et al., 2018; Ren et al., 2021), which can increase the sediment loading from the dam to river but might lead to a suite of negative ecological effects in the downstream river reaches (Espa et al., 2016, 2019, 2019). At the coasts and estuaries, coastal engineering works have been constructed to retain or utilize the available and scarce sediment resources (Sumer et al., 2001; Alvarez-Guerra et al., 2008; van Rijn, 2011), however the impacts of these different types of coastal engineering on the ecosystems, especially to the coastal habitat restoration and waterbird populations, are not understood.

Yangtze River Delta (YRD) in China is one of the largest coastal habitats in the world, it is also an important stopover site for energy replenishment of migratory birds along the East Asian-Australasian Flyway (EAAF) (Ma et al., 2009). The YRD is generated by tremendous sediments from the Yangtze River downstream and extends from 31°42′25.44″ N in the south to 30°52′35.82″ N in the north (Fig. 1a), with a coastline of 504 km in length and a total of 4500 km^2 of coastal wetlands (Maff and Larsen, 2000). Over the last five decades, approximately 1100 km^2 coastal wetlands have been lost in the YRD due to various kinds of land reclamation and coastal engineering construction (Du et al., 2016). Meanwhile, Yangtze River damming, particularly the operation of Three Gorges Dam after 2003, has reduced approximately 70% of sediment discharge to the YRD and sediment discharge decreased to 1.4×10^8 t/yr in the recent years (Dai et al., 2018). This substantial reduction of sediments has triggered the erosion of the delta front (Yang et al., 2006; Du et al., 2016) and further threatened natural habitat stability and function, e.g., habitat and food supply for waterbirds (Tian et al., 2008; Zhang et al., 2018).

Although damming decreases sediment inputs from river to the coastal deltas, there is still a certain amount of sediments entering the estuarine region (Syvitski et al., 2005, 2009; Yang et al., 2017). Majority of sediments in the estuary could be washed away because of inefficient sediment traps on delta plains (Giosan et al., 2014). Meanwhile, the

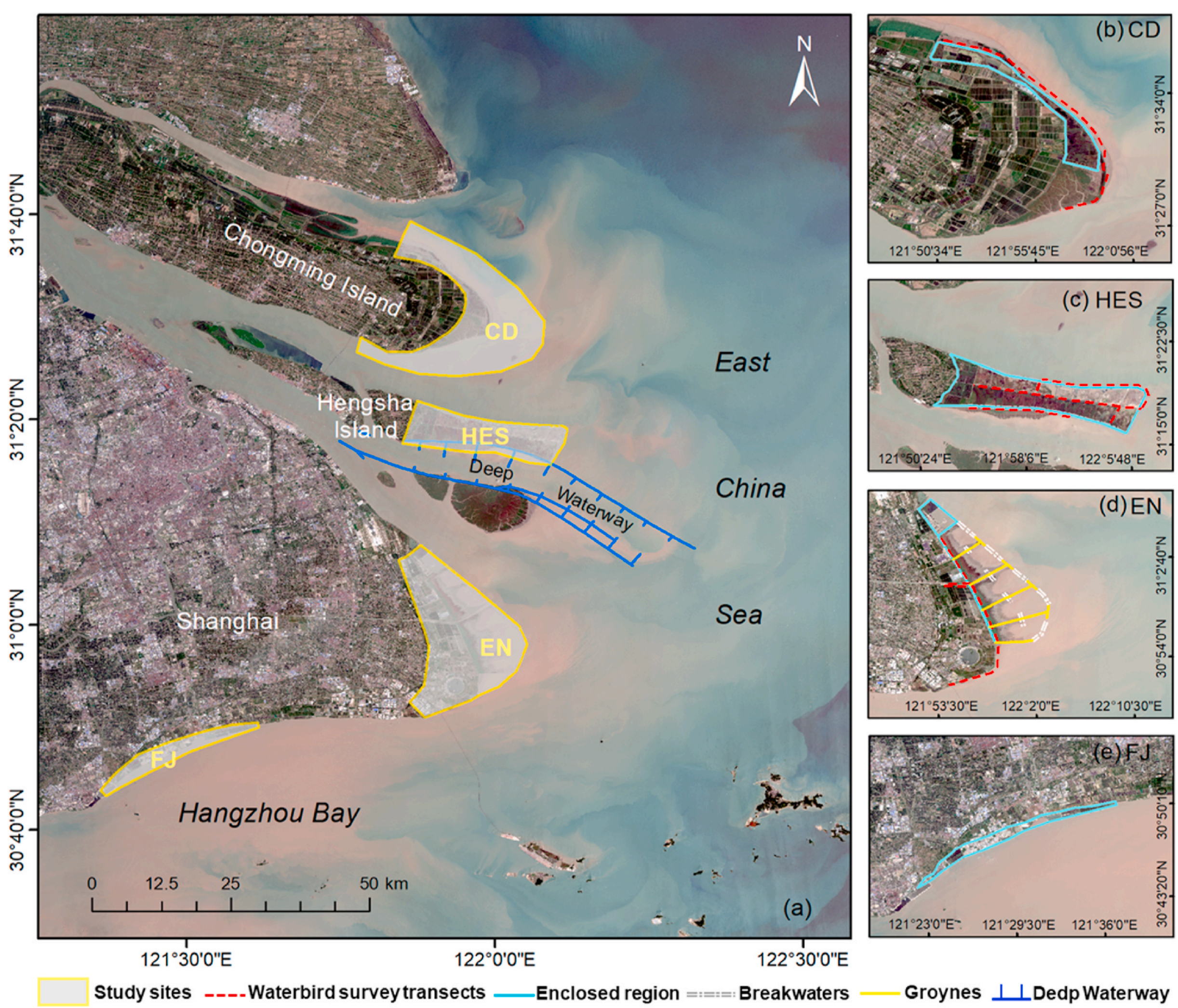

Fig. 1. Location map of the four study sites and the transects for waterbird survey in the study area around the Yangtze Estuary mouth.

sediments deposited at the estuary channel need to be dredged for shipping safety. About 300 × 10^6 m^3 sediments in shipping channel of the Yangtze estuary were dredged from 1998 to 2010 to maintain waterway safety (Du et al., 2016). How to effectively retain and utilize the limited sediment resources is a prerequisite for coastal habitat restoration. In this study, we compared and assessed the restorative effects of the solutions to retain and utilize sediments which were applied at three rehabilitated sites in YRD and a contrasting site without any restoration measure, by using four indices of ecological importance, i.e. variations in total wetland area, wetland vegetation area, waterbird abundance and species richness during 1984–2018. Based on these, we aim to put forward a feasible and effective solution to retain and utilize sediments in the waterbird habitat restoration under the condition of “sediment scarcity".

2. Materials and methods

2.1. Study sites description

Three rehabilitated sites (CD: Chongming Dongtan; HES: Hengsha Eastern Shoal; EN: Eastern Nanhui) and one contrasting unrestored site (FJ: Fengxian and Jinshan) were chosen for comparison in this study (Fig. 1a–e and Table 1). CD and EN utilized construction of coastal engineering to restore coastal habitats through effective sediment retention and aggradation, while HES utilized nearby dredged sediments to nourish and create new habitats (Table 1).

The CD is located on the eastern fringe of Chongming Island (Fig. 1a and b). It is an important Ramsar site for EAAF migratory waterbirds, approximately 288 waterbird species have been recorded there (Ma et al., 2002). Although sediment accretion is still positive on the tidal flat, the progradation rate has decreased from 4.3 km^2/yr in 1977–1983 to 0.3 km^2/yr in 2000–2011 and has almost reached the balance between accretion and erosion due to sediments reduction (Du et al., 2016; Yang et al., 2017). From 2013 to 2016, about 24 km^2 ecological engineering works were undertaken in CD as part of the “*Spartina alterniflora control and waterbird habitat optimization*” project to control the expansion of this invasive species and restore the degraded waterbird habitats (Fig. 1b) (Yuan et al., 2014). The control engineering involved first enclosing the area invaded by *S. alterniflora* with a 27 km length concrete dyke (Fig. 1b and Table 1). The above-ground stems of *S. alterniflora* on its flowering period were cut by a harvester and then waterlogged one month by maintaining a 40–50 cm water level. After that, a variety of habitat types (ponds, native saltmarshes, mud flat and creeks) were created and maintained by a control system of sluices and pumps (Zhao et al., 2020), and native plant *Scirpus mariqueter*, the most important food for many waterbirds, was restored by transplanting in the enclosed ecological engineering area and on the natural coastal intertidal flat outside the dyke (Yuan et al., 2020).

The EN is located on the southern Yangtze River Estuary and recognized as a national important waterbird area by Birdlife International in 2008 (www.birdlife.org/datazone/site) (Fig. 1a and d). Due to sediment discharge reduction and reclamation, the growth rate of the subtidal zone was reduced from 6.0 km^2/yr in 1983 to 1.5 km^2/yr in 2011 (Du et al., 2016). In addition, EN is also one of the intensive reclamation sites at Yangtze River Estuary. To promote natural accretion and tidal flat development after reclamation, an engineering project was conducted in EN from 2013. The concrete T-head groynes and permeable rock breakwaters were constructed to trap sediments and promote sedimentation and accretion. About 14.9 km^2 of intertidal flat was half-enclosed in the north of the engineering area, and the remainder of the area was kept as natural coastal wetlands experiencing a diurnal tidal cycle (Fig. 1d and Table 1). The sedimentation in the engineering area has promoted a fast development of saltmarshes since 2013.

The HES is located on the east of Hengsha Island (Fig. 1a and c). It is formed by sedimentation from the Yangtze River but also faced a decline in the rate of progradation like the CD (Du et al., 2016). The southern part of the island is facing to the Yangtze River Estuary Deep Waterway (YREDW) (Fig. 1a). To maintain the deep waterway, concrete dyke-groyne systems approximately 50 km long were constructed on both sides of the channel between 1998 and 2010 (Fig. 1a). From 2003 to 2017, more than 100 km^2 of intertidal wetlands in the north of the YREDW was enclosed in stages by new dykes (Fig. 1c). From 2006 to 2015, approximately 2.3 million m^3 sediments were dredged from the deep-water channel every year, and cumulatively 130 million m^3 dredged sediments were filled into the nearby enclosed area to create new wetlands (Table 1). The dredged sediments were pumped hydraulically and dumped nearby into the enclosed area by a trailing suction hopper dredger. After that, the newly created wetlands underwent a process of natural succession from water to marsh wetlands. Meanwhile, the natural wetlands and saltmarshes (dominated by *P. australis*, *S. mariqueter* and *S. alterniflora*) were developed outside the south dyke.

The FJ is located on the north shore of Hangzhou Bay (Fig. 1a and e). In the early 1980s, FJ contained large area of coastal wetlands which provided feeding grounds for a variety of waterbirds (Ge et al., 2007). After the reclamations for industry and agriculture use during 1990s, there was no any engineering work conducted for retaining/acquiring sediments or rehabilitation. Therefore, FJ has been chosen to represent unrestored site, contrasted with other sites (CD, EN and HES). As the result of reclamation and the erosive characteristic of the FJ coast, the habitats had been lost almost completely and the waterbird abundance dropped drastically. Therefore, no waterbird survey was carried out at FJ after 2003.

Table 1
Summary comparison of the solutions to wetland rehabilitation among the four study sites.

Sites	Location	Types of sediment retention or acquisition
Chongming Dongtan wetland (CD)	31°25′N-31°38′N 121°50′E–122°05′E	**Type I**: High-tide embankment built to hasten natural aggradation of intertidal wetlands outside the dykes.
Eastern Nanhui wetland (EN)	30°50′N-31°07′N 121°50′E–122°03′E	**Type II**: Groynes and breakwaters built to promote natural sediment retention, accretion and aggradation and habitat seaward progradation.
Hengsha Eastern Shoal wetland (HES)	31°16′N-31°22′N 121°50′E–122°07′E	**Type III**: Dredged sediments from nearby shipping channel used to fill enclosed region to create new habitat and natural sediment accretion and aggradation promoted outside the new dykes.
Fengxian and Jinshan wetland (FJ)	30°73′N-30°84′N 121°37′E–121°62′E	**Type IV** (Contrast): No intervention methods taken after reclamation beginning in the 1980s

2.2. Restorative effort assessment

In this study, the indices of positive change in total wetland area, wetland vegetation area, waterbird abundance and species richness were used to assess the success and relative benefit of restorative efforts. The total wetland area and wetland vegetation area for four sites were extracted from a set of multitemporal LandSat Thematic Mapper (TM) images (30 m × 30 m spatial resolution) from 1984 to 2018 downloaded from Geospatial data cloud (http://www.gscloud.cn/sources/index?pid=1&type=1). According to the time when the remote sensing images were taken and the tide levels of the sites at this moment (according to the tide table), a series of remote sensing images with the producing time at same stages of tidal levels were selected and used for extracting shorelines by identifying grayscale thresholding waterlines (Zhao et al., 2008) and for obtaining the area of coastal wetlands. The panchromatic and multispectral images of LandSat were fused using Pan-sharpening fusion method and initialized using spectral enhancement methods.

The habitat and wetland vegetation classes of seawall, water, tidal flats and *S. alterniflora* were identified and selected as training samples. Additionally, a supervised classification was carried out to get the wetland vegetation area in each of the four sites. As the control project of *S. alterniflora* was conducted only in CD, therefore the aerial extent of both native and invasive species was determined only for the site CD, not for EN, HES and FJ.

Waterbird surveys including species richness and abundance were conducted every month at CD in 2013, 2016 and 2018, at EN in 2012, 2014 and 2018, and at HES in 2003, 2013, 2016 and 2018, which represented the years before, end and after the engineering works (see the Supplementary Information: Figure SI1). Based on the previous survey, one to three fixed transects each with 200 m detection width were set and monitored at CD, EN and HES respectively, which could cover all types of habitats (different dominant saltmarsh communities, mudflat and water area) and support reasonable survey for waterbird numbers and species at different sites (Fig. 1 b-d). All transects were set permanently in each site during the study period, which could be used for comparing the waterbird dynamics before and after engineering works. The species richness of waterbirds at each site was calculated from data across all transects in all months each year. For abundance, the maximum number of waterbirds (highest monthly count) each year at each site were presented. Total annual waterbird counts were also given, as these might include counts of the same birds in multiple months, and they could provide an indication of the habitat usage level by waterbirds across an entire year (see Supplementary Information: Fig. SI1). While the waterbird data for FJ were from a published literature (Ge et al., 2007).

3. Results

3.1. CD: natural aggradation hastened by high-tide embankment

After implementation of the *S. alterniflora* control engineering project, 23.6 km^2 high-tidal wetlands were enclosed by a new dyke in CD (Fig. 2a), and the area of natural wetlands decreased from 101.95 km^2 in 2013 (before the engineering) to 86.8 km^2 in 2016 (the project completion year) (Fig. 2b). Subsequently, the high-tidal embankment hastened the growth of new intertidal wetlands outside the dykes by natural aggradation and promoted shoreline advancing seaward with a progradation rate of 3.83 km^2/yr between 2016 and 2018. The shoreline in the south of CD was less affected by the embankment and retained a relatively stable state (Fig. 2a). In 2018, the total area of wetlands (natural coastal wetland and embanked marsh wetland) was increased to 118.06 km^2 in CD. Due to the continued control on *S. alterniflora* (its area decreased from 17.84 km^2 in 2013 to 0.18 km^2 in 2018), while the native *S. mariqueter* area increased at a rate of 2.47 km^2/yr, i.e. increased from 25.47 km^2 in 2016 to 30.41 km^2 in 2018 (Fig. 3b).

Following the *S. alterniflora* control and native saltmarsh recovery (Fig. 2b), waterbird abundance increased 2.5 times than that in 2013 (maximum monthly count from 9095 in 2013–23072 in 2018, Fig. 2c), while waterbird species richness stabilized at around 76 (Fig. 2c). Total combined annual counts of waterbirds also increased from 43015 in 2013–123841 in 2018 (Fig. SI1).

3.2. EN: sediment retention promoted by building groynes and breakwaters

At the beginning of what became known as the "*ecological silt promotion engineering project*" in EN, the area of coastal wetlands and wetland vegetation increased slowly from 21.68 km^2 to 4.78 km^2 in 2012 (before engineering) to 34.81 km^2 and 6.59 km^2 in 2014, respectively (Fig. 3a and b). Subsequently, with the construction of groynes and breakwater, the sediments were retained and accreted in the silt promotion area, and the shoreline advanced quickly seaward with a progradation rate of 22.5 km^2/yr from 2014 to 2018. Along with the development of intertidal wetlands, saltmarsh vegetation developed naturally to 22.24 km^2 in 2018 (Fig. 3b). Compared with 2012, the engineering works were responsible for the development of 103.12 km^2 of new coastal wetlands (5 times greater), 14.86 km^2 enclosed wetlands and 17.46 km^2 saltmarshes (4 times greater) in 2018 (Fig. 3b).

In 2012 (before engineering), the waterbird count at EN was 5170

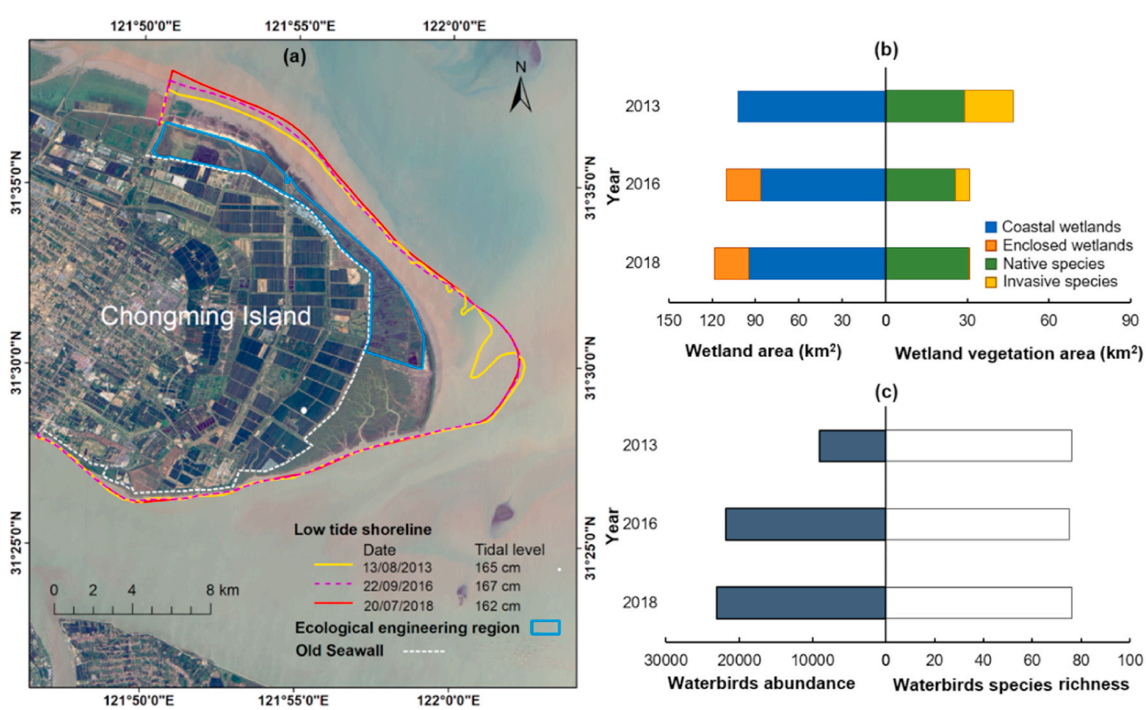

Fig. 2. The dynamics of wetlands, wetland vegetation and waterbirds in CD during the 2013–2018 period. (a) shows the shoreline at low tidal levels, (b) shows the dynamics of wetland area, native species vegetation and invasive species vegetation, (c) shows the dynamics of number and species of waterbirds (Note: the ecological engineering works began in 2013 and were completed in 2016).

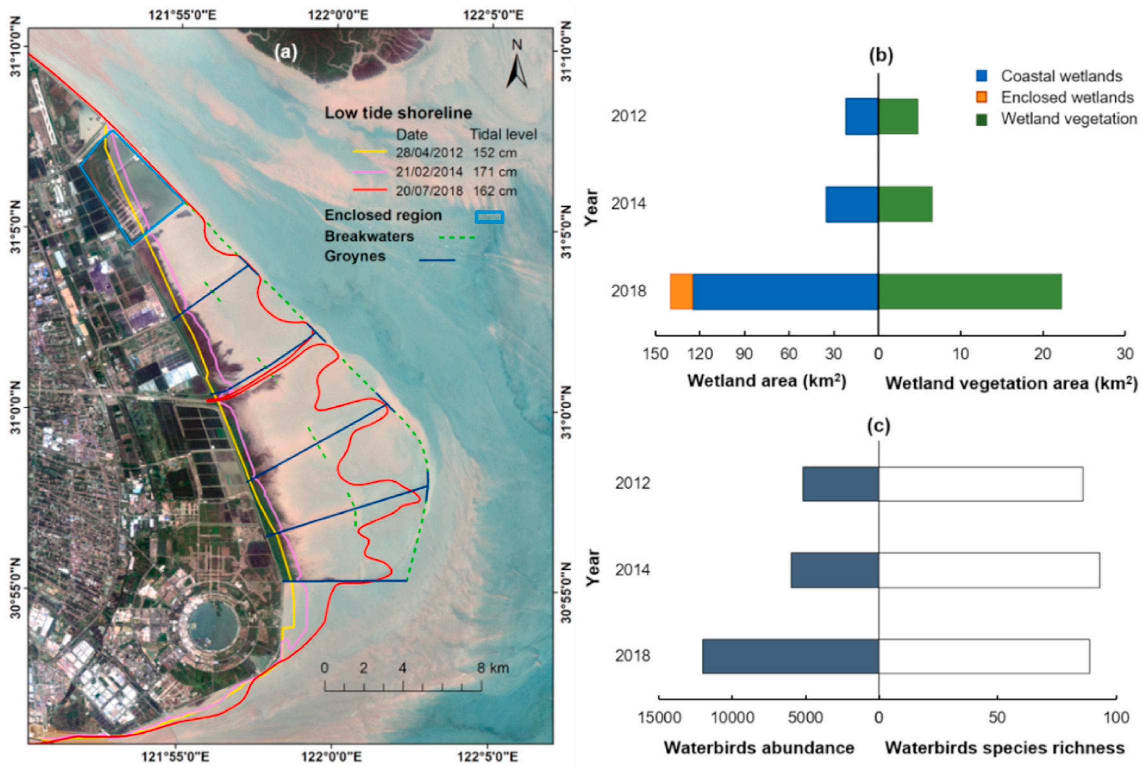

Fig. 3. The dynamics of wetlands, wetland vegetation and waterbirds in EN during the 2012–2018 period. (a) shows the shoreline at low tidal levels, (b) shows the dynamics of wetland area and wetland vegetation area in EN, (c) shows the dynamics of number and species of waterbirds (Note: the *ecological silt promotion engineering project* was conducted from 2013 to 2018).

(Fig. 3c). In 2014, early in the period of the engineering works, a similar monthly maximum count of 5972 waterbirds was made. From 2014 to 2018, concomitant with the greatly increased area of coastal wetlands and saltmarshes after the engineering works, the waterbird abundance reached 2.3 times (12022 in 2018) compared with that in 2012 (Fig. 3c and Fig. SI1). The waterbird species richness stabilized at around 90 before and after the construction of silt promoting engineering (Fig. 3c). Over the same period, total annual waterbird counts increased from 34274 to 64028.

3.3. HES: creation of new wetland habitat by utilization of dredged sediments

In 2003 (before the engineering works), there was only 6.17 km^2 coastal wetlands and 1.88 km^2 wetland vegetation in HES (Fig. 4a and b). After that, 106.11 km^2 intertidal wetlands were enclosed in stages by building a new dyke from 2003 to 2018 and by filling the enclosed area with dredged sediments from the nearby YREDW from 2006 to 2015 (Fig. 4a and b). Depending on the filling of dredged sediments and natural succession, the newly created wetlands were composed of vegetated or bare lands and water surface in 2018. Meanwhile, the embankment also promoted the development of 31.81 km^2 of natural intertidal wetlands outside the new dyke (Fig. 4a and b). By 2018, the wetlands area had increased 21 times and the vegetation wetlands 36 times than those in 2003, respectively (Fig. 4b).

Along with the increase in the total wetland area and the vegetation wetland area, the number of waterbirds increased remarkably, from less than 150 in 2003 to a maximum monthly count of 12388 in 2016 after completion of filling the enclosed area with dredged sediments (Fig. SI1). By 2018, waterbird numbers had increased further to 15042, 121 times higher than that in 2003. The total annual waterbird counts also increased to 91193 or 150 times than that in 2003. The species richness reached a total of 97 species in 2018, which was more than 3 times than that in 2003 (Fig. 4c).

3.4. FJ: No engineering or habitat restoration works in the reclamation coast

Between the years of 1984 and 2003, 22.1 km^2 coastal wetlands had been reclaimed for industry and agriculture use in FJ. After implementation of reclamations during 1990s, there was no engineering work conducted for retaining/acquiring sediments or wetland restoration. As the result of reclamation and the erosive characteristic of the FJ coast (Fig. 5a), the total wetland area dropped from 52.75 km^2 in 1984 to 26.64 km^2 in 2003, and the wetland vegetation area decreased 35.4% to just 5.94 km^2 by 2003 (Fig. 5b). As a result of habitat loss, the bird abundance decreased 50% to only 346 birds in 2003 compared with that in 1984 (Fig. 5b). According to the coastal wetland survey in 2018, the total wetland area and the wetland vegetation area in FJ were 9.23 km^2 and 1.71 km^2 respectively (Fig. 5b). Without any restoration engineering work, the waterbird habitats in FJ kept decreasing after the reclamations in 1990s.

4. Discussion

In many of the world's populous coastal cities and their environs, the development and utilization of coastal zone have been made at the expense of the coastal ecosystem functions and services they provide (Amano et al., 2018). The restoration of coastal wetlands has now become crucial for conservation of the habitats and their biodiversity (Amano et al., 2018). However, with the incontrovertible decrease of sediment inputs from river upstream to the coasts and estuaries and the challenges of sea-level rise induced by climate change, coastal zones are facing erosion risks and big challenges for coastal wetland habitat restoration (Li et al., 2018). Sediment replenishment is critical for restoration and conservation of the estuarine and coastal habitats

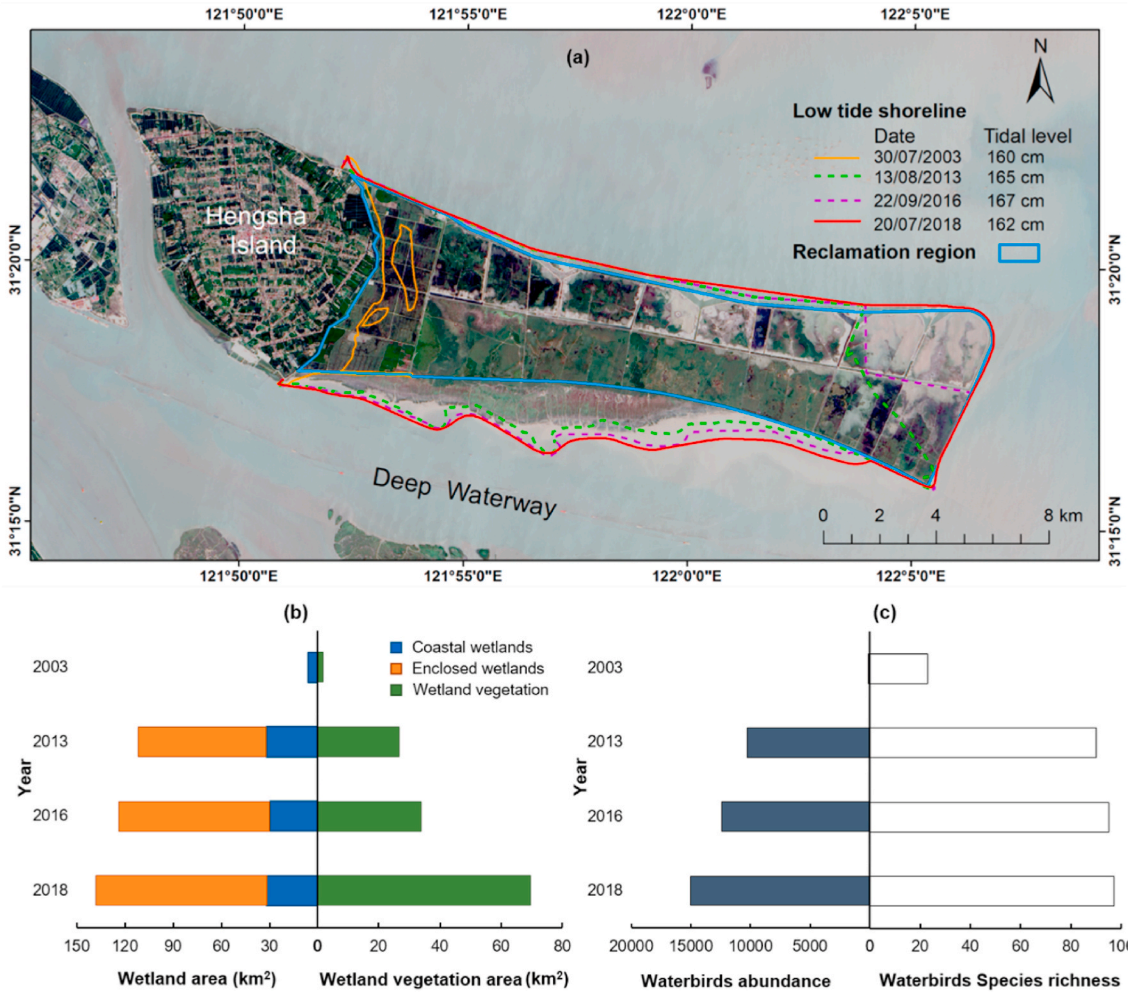

Fig. 4. The dynamics of wetlands, wetland vegetation and waterbirds in HES during the 2003–2018 period. (a) shows the shoreline at low tidal levels, (b) shows the dynamics of wetland area and wetland vegetation area in HES, (c) shows the number and species of waterbirds (Note: the engineering works were undertaken between 2003 and 2017).

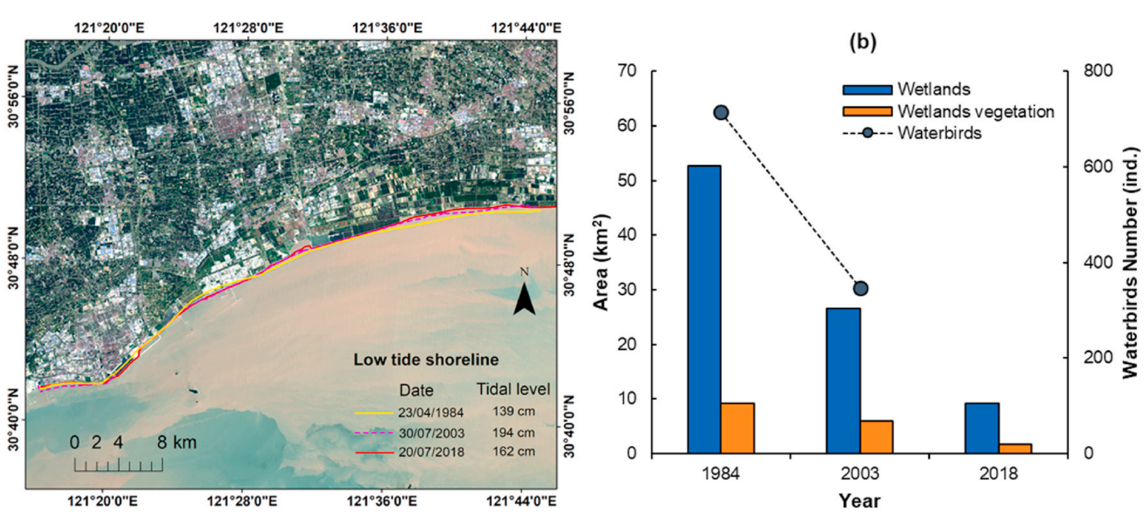

Fig. 5. The dynamics of wetlands area and waterbird abundance in FJ during the 1984–2018 period. (a) shows the shoreline at low tidal levels, (b) shows the dynamics of wetlands and waterbirds in FJ (Note: the reclamations were conducted in 1990s and no waterbird survey was carried out in FJ after 2003).

(Syvitski et al., 2009; Li et al., 2018). The results of this study have demonstrated that both moderate silting structures and ecological utilization of nearby dredged sediments are the available solutions for retaining/acquiring of sediments during coastal habitat restoration in situations of permanently reduced sediment discharge. Well situated, hard silting structures could promote the development of coastal wetlands, saltmarshes and waterbird habitats through effective sediment retention and deposition. Ecological utilization of dredged sediments from nearby has been also demonstrated to provide an economically viable use of dredge spoils to nourish and create new wetland habitats.

In terms of the efficiency of increasing habitat area, the rate of increase in wetland area at CD, EN and HES (with engineering works) were all much higher than that at FJ (with no intervention), which indicated that moderate coastal restoration engineering could be an efficient mechanism for retaining or utilizing sediments in the development of coastal habitats. Construction of coastal hard structures, such as groynes, breakwaters and artificial submerged reefs, are the available solutions to retain and supplement sediments within the silt promotion region and to reduce coastal erosion (Sumer et al., 2001; van Rijn, 2011). The EN case has demonstrated both the feasibility and effectiveness of this solution. Well situated coastal groynes and breakwaters could effectively intercept and retain precious sediments from the estuary to promote natural sedimentation build-up and facilitate a rapid intertidal wetland development. The ecological silt promotion engineering works could result in a creation of more than 100 km^2 of new wetland habitats in EN. Dredged sediments have been widely used as a natural resource for creation or recovery of wetland habitats, marsh recovery and have also led to improvements in fisheries. (Alvarez-Guerra et al., 2008). Many studies have been sought to determine how to beneficially use dredged materials and sustainably manage such operations (Streever, 2000; Yozzo et al., 2004; Alvarez-Guerra et al., 2008). The HES case in this study has demonstrated a feasible solution whereby ecological use of dredge sediments nearby could be beneficially employed to create new wetlands. In addition, the new dykes can promote natural accretion and aggradation outside, and increase the of natural wetland area. The CD case in this study has demonstrated that the construction of dykes in the high tidal zone could promote natural siltation. As in the silted tidal flat system, construction of dykes would normally accelerate the growth of intertidal wetlands outside the dyke (Du et al., 2016).

The efficiency of retaining sediments and accretion in the site CD is lower than that in EN. This could be because of the promoted progradation in CD mainly by natural aggradation which greatly depends on the sediment input from the estuary. While for the site EN, in addition of natural aggradation, the constructed groynes and breakwaters can accelerate sediment accretion and promote wetlands development through weakening hydrodynamics and promoting sedimentation in the engineering area. Although the efficiency of retaining sediments and accretion in CD is not as high as that in EN, the progradation rate outside the new dyke in CD (3.83 km^2/yr) has been higher than the historical progradation rate of 0.3 km^2/yr (Du et al., 2016).

The results of the wetland vegetation survey from this study indicate that the growth rate of wetland vegetation area is HES > EN > CD > FJ. In the site HES, as the suitable conditions (mainly the elevation) have been created in a short period by filling with dredged sediments in the enclosed area, this could allow wetland vegetation to colonize on the new habitats quickly (Yuan et al., 2020). The practice of establishing *S. alterniflora* marshes on dredged spoils along the eastern and Gulf coast of the United States since 1969 has also demonstrated that the saltmarshes could established successfully on the dredged materials. These recreated saltmarshes provide many ecological functions and new habitats for wildlife (Streever, 2000; Yozzo et al., 2004). In the EN case, the saltmarshes have quickly advanced seaward along with progradation of the tidal flats in the engineering area. The newly developed saltmarshes can not only play as the important habitats, but also promote sediment retaining and deposition and act as an important buffer to protect the coastal defenses (Li et al., 2018). In the CD case, the control of the invasive species might reduce the total vegetation area, while the restoration of native saltmarsh *S. mariqueter* has resulted in optimized wetland vegetation structure (Yuan et al., 2020) and provided a range of habitats utilizable by waterbirds. While the FJ case indicates that necessary restoration engineering works should be taken to stop the continuously decreasing trend in the wetland habitats.

Waterbird populations suffer multiple types of threats, while the habitat loss is the most common threat to waterbirds (Wang et al., 2018). In our study, the waterbird abundance and the total annual counts at the three restoration sites CD, EN and HES have increased significantly along with the increases in quality and quantity of the habitat area (Fig. SI1). After the implementation of the engineering works, the waterbird abundance has increased 1.5 times in CD and 1.3 times in EN, while the species richness has kept stable. The HES case demonstrates a successful practice that the newly created wetlands by ecological use of dredge sediments nearby can provide new valuable habitats for waterbirds, where the waterbird abundance has increased 121 times and species richness increased 3 times. However, whether the newly created marshes by the dredged sediments could provide the waterbird habitats as good as the natural marshes still need further investigation (Streever, 2000). In addition, because the sites CD, EN and HES have been recognized as the important stopover sites for migratory birds, the waterbirds abundance shows obviously seasonal differences following migration time. The waterbird numbers would be much higher in southward migration during winter and the lowest in northward migration during summer (see Fig. SI1). The results again indicate that coastal wetland restoration plays a critical role in rehabilitation and waterbird conservation, especially for migratory birds.

Economic and environmental benefits are also the important factors to be considered in the coastal habitat restoration (Li et al., 2018). The sand nourishment has been considered as a costly method in the long-term if adequate volumes of compatible sand are not available at nearby (economic) locations (van Rijn, 2011). In the HES case, the engineering work of creation wetland habitats by utilization of sediment dredged from nearby shows obvious economic and environmental benefits. In the YRD, approximately 2.3 million m^3 of sediments for maintenance of the deep-water channel were dredged and disposed offshore every year before this engineering work, which cost billions of RMB for dredging, transportation and disposal of the dredged sediments (Du et al., 2016). The disposal of dredging sediment offshore not only wastes precious sediment resources, but also potentially influences marine environment and habitats both of dredged channel and offshore spoil area, such as increasing turbidity, reducing oxygen levels, changing water chemistry, releasing toxic substances, and impacting the composition and abundance of benthic community (Szlauer-Łukaszewska and Zawal, 2014; Martínez-Colón et al., 2018; Kanellopoulos et al., 2020). The engineering work of creation wetland habitats by utilizing dredged spoils in the HES case could avoid potential environmental damage caused by disposal of dredged spoils. Moreover, as an active buffer zone in the transport of pollutants between land and sea, the newly developed vegetation on the created wetlands could not only provide new habitats for waterbirds, but also act as an efficient sink for contaminants as the wetland vegetation could act as a trap for many contaminants (Reboreda and Caçador, 2007; Kim et al., 2020). Although this study has not monitored the possible environmental influences of the dredged spoils on the vegetation, waterbird habitats and benthos in the newly created wetlands, the concern on the possible environmental influences of utilizing dredged spoils is worth to be further investigated in the HES case.

The CD case is the most expensive among the three rehabilitation sites, which cost about 1.3 billion RMB for the construction of 24 km^2 ecological engineering project. This includes building 27 km of enclosed dykes and 4 sluices and a series of pumps, *S. alterniflora* control, habitat optimization, and restoration of native saltmarshes etc (Li et al., 2021). In the HES case, the engineering construction of new wetland creation by building up the enclosed area and filling with dredged sediments could be quite expensive. However, considering the reduction in cost for the disposal of dredge spoil offshore and much less management for the newly created wetlands, the efficiency in cost-benefit would be very high. In the EN case, the cost for this restoration engineering is the lowest and the efficiency in cost-benefit would be the highest. The investment for building groynes and breakwaters is not only much less than that of dykes, but also is required only at the initial stage of the engineering work. After the engineering work, wetland habitats could been quickly restored in the silt promotion area by the natural tidal processes and succession without any further expense.

5. Conclusion

The restoration of coastal wetlands and addressing the problem of "sediment scarcity" in coastal deltaic systems have now become crucial for conservation of the habitats and their biodiversity. In this study, three solutions have been adopted for the restoration sites, including the engineering for rehabilitation and promoting sediment deposition in CD, the engineering for promoting sediment deposition and settlement in EN, and the engineering using dredged sediments to create new wetland habitats in HES, in contrasting to the unrestored wetlands in FJ. The comparison and assessment on these restoration engineering works have demonstrated that both moderate silting structures and ecological utilization of dredged sediments from nearby are the available and effective solutions for retaining/acquiring of sediments during coastal habitat restoration in situations of "sediment scarcity". Well situated, hard silting structures could promote the development of coastal wetlands, saltmarshes and waterbird habitats through effective sediment retention and deposition. Ecological utilization of dredged sediments from nearby has been also demonstrated to provide an economically viable use of dredge spoils to create new wetland habitats. While the FJ case has indicated that necessary restoration engineering works should be taken to stop the continuously deteriorating trend in the coastal wetlands. While selecting these solutions to restoration and rehabilitation of coastal wetlands, the objectives of restoration, local hydro-sedimentation dynamics, bio-physical conditions and social-economic effects should be fully considered for a feasible and successful restoration.

Author contributions

The study was conceived and designed by LY, DL, LZ and JK; XY, SB, QM and WW performed the bird field survey; LY, BT and ZZ performed remote sensing data and wetland area change analyses; LY, DL and JK wrote the first draft of the manuscript and coordinated subsequent revisions. All authors have agreed to the final manuscript.

Declaration of competing interest

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

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Appendix A. Supplementary data

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