Mismatch between watershed effects and local efforts causes the failure of China's large-scale coastal restoration

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Abstract

Coastal restoration is considered a key solution to counteract coastal degradation and mitigate climate change. In recent years, China's coastal restoration projects have multiplied, with more than one billion US dollars spent during 2016-2019. Due to the sudden die-off of *Sueda* vegetation in the Liaohe River Estuary, local managers have spent more than 30 million dollars to restore these saltmarshes since 2015. However, these projects either failed or yielded limited results. Combining long-term remote sensing data to ground-based surveys, we found that saltmarshes in the Liaohe River Estuary have experienced two dramatic diebacks separated by a recovery period. Rapid changes in vegetation canopy were primarily driven by freshwater availability, especially river discharge. We suggest that mismatch between degradation drivers at the watershed-scale and restoration efforts at the local-scale hinders the success of coastal restorations: efforts to restore these saltmarsh will always be thwarted unless freshwater availability is replenished. Therefore, we propose a watershed-based solution, incorporating water replenishment from the river basin in estuarine restoration project. Given the declining trends in river freshwater inputs in low-middle latitude coastlines, such wetland deterioration might also occur in many world's estuaries. Lessons from China's large-scale coastal restoration projects can help other regions to avoid similar shortcomings, and improve our restoration practices before too many resources are lost.

Keywords: Coastal restoration; Estuary; Freshwater availability; River discharge; Scale mismatch; Water replenishment

1. Introduction

Coastal wetlands are highly productive and valuable ecosystems, providing food and shelter for endangered species, coastal protection from storms and erosion, carbon sequestration, and cultural services such as tourism and education (Costanza et al., 1997; Barbier et al., 2011). However, coastal wetlands are in rapid decline due to global stressors associated with climate change, such as increasing sea surface levels and number of extreme events, as well as local stressors like land reclamation, pollution and eutrophication (Halpern et al., 2007; Gedan et al., 2009; Suding 2011; Murray et al., 2019). To counteract degradation, enhance biodiversity and mitigate climate change, the creation and restoration of wetlands is rapidly becoming one of the key global solutions for coastal sustainability (Zedler et al., 2000; Temmerman et al., 2013; Bayraktarov et al., 2016; Liu et al., 2016).

There have been surprisingly few studies reporting the failure of large-scale coastal restoration projects and the reasons that led to these failures (Zedler 2007; Bayraktarov et al., 2016). This lack of information could be due to biases in the scientific literature towards publishing successful rather than failed restoration practices. This bias can hinder the incorporation of new scientific results in the decision-making process and in restoration practices.

Active restoration methods for coastal wetlands, such as saltmarshes, mangroves and seagrasses, mainly focus on planting seeds, seedlings or propagules, hydrological restoration or re-connection, and managed retreat (Temmerman et al., 2013; Bayraktarov et al., 2016; Liu et al., 2016). Restoration efforts commonly fail due to inadequate restoration approaches, poor site selection, unforeseen extreme events (e.g., flood damage or severe storms), time, labor and cost constraints (Geist and Galatowitsch, 1999; Bayraktarov et al., 2016). The mismatch between degradation drivers and restoration techniques always leads to restoration failure (Zedler et al., 1996; Bergen et al., 2000; Liu et al., 2020a). Unless such obstacles are overcome, commitment to new restoration projects would likely remain limited. For example, in the Tijuana Estuary salt marsh, USA, the survival of planted *Batis maritima* L., *Jaumea carnosa* Gray and *Salicornia bigelovii* Torrey was less than 10% before addressing a tidal restriction that caused high salinity (Zedler et al., 2003). In contrast, submerged aquatic vegetation communities in the Chesapeake Bay have been successfully restored after reducing the inputs of in situ nutrients, wastewater-treatment effluent nitrogen and total suspended solids, which caused light attenuation (Ruhl and Rybicki 2010). Therefore, identifying the key drivers affecting ecosystem degradation and recovery of coastal wetlands is critical to restoration efforts.

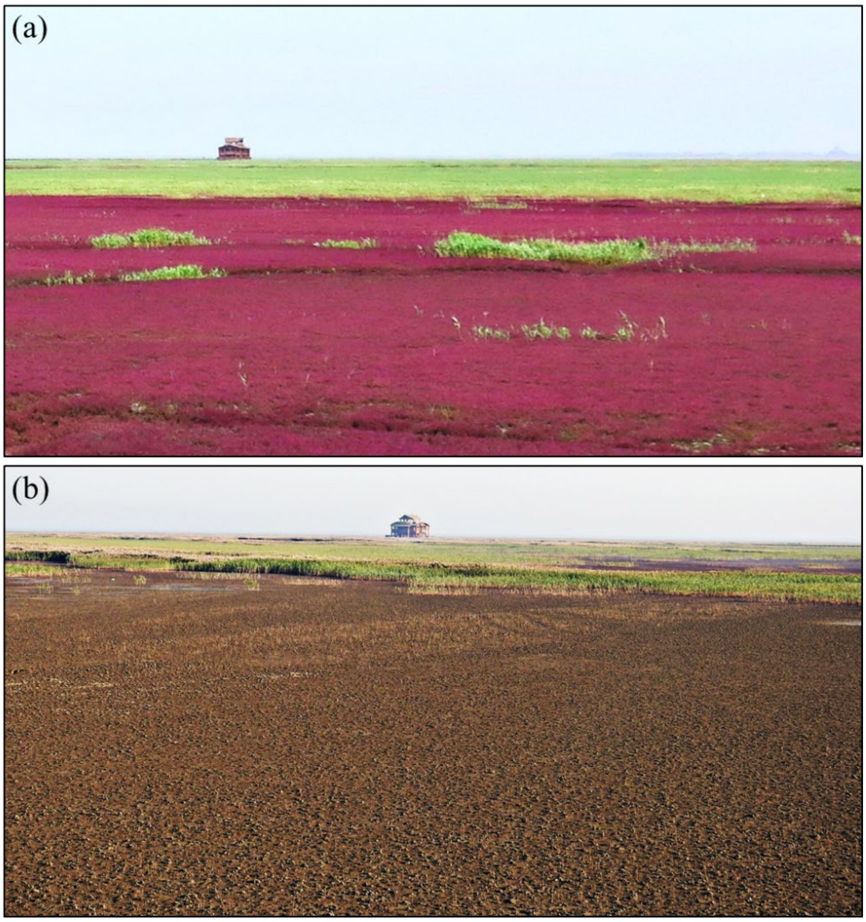

Figure 1. Photographs of the study site. This salt marsh ecosystem experienced an intensive vegetation die-off since fall 2015. The red beach in June 2015 (a) and (b) an expansive bare land in September 2016. The purple and green vegetation covers are *Suaeda salsa* and *Phragmites australis*, respectively. Photographs were taken by Zezheng Liu.

Since the beginning of the 21st century, the number of China's coastal restoration projects increased drammatically (Liu et al., 2016). More than one billion US dollars has been spent to coastal restoration efforts from 2016 to 2019 (Ministry of Natural Resources of the People's

Republic of China 2019). The Liaohe River Estuary, a national nature reserve, experienced a large-scale die-off event starting in the fall of 2015 (Figure 1). The rapid decline in marsh vegetation communities appears to have significantly restricted ecological services, such as carbon storage, provision of tourism resources and support for biodiversity (Chen 2018; Fu et al., 2019). These impacts, and an awareness of the importance of coastal wetlands, motivated the local government to restore marsh vegetation starting in 2015 (Liu et al., 2020a). Local managers have spent more than 30 million dollars to restore saltmarshes, by planting seeds, transplanting and controlling grazers (Department of Forestry of Liaoning Province, 2017). However, the outcomes were always limited or often the restoration failed (Figure 2). Liu et al. 2020a found that planted *S. salsa* died in the high restored marsh at the end of the second growing season (Figure 2c). Figure 2 a and b show that adding seeds of *Suaeda* also failed to produce a vegetation canopy. Although grazers crabs can be effectively controlled with cages, this method has many negative effects on local flora and fauna. This approach not only requires a lot of manpower and financial resources, but also harms plants because of human trampling when setting cages and taking out crabs. In addition, fishes and birds were also trapped in the cages, damaging the local biodiversity (Figure S1). To restore habitat more effectively, managers need to know why marsh vegetation communities have been degraded, and to improve restoration strategies based on scientific understanding of the causes of degradation.

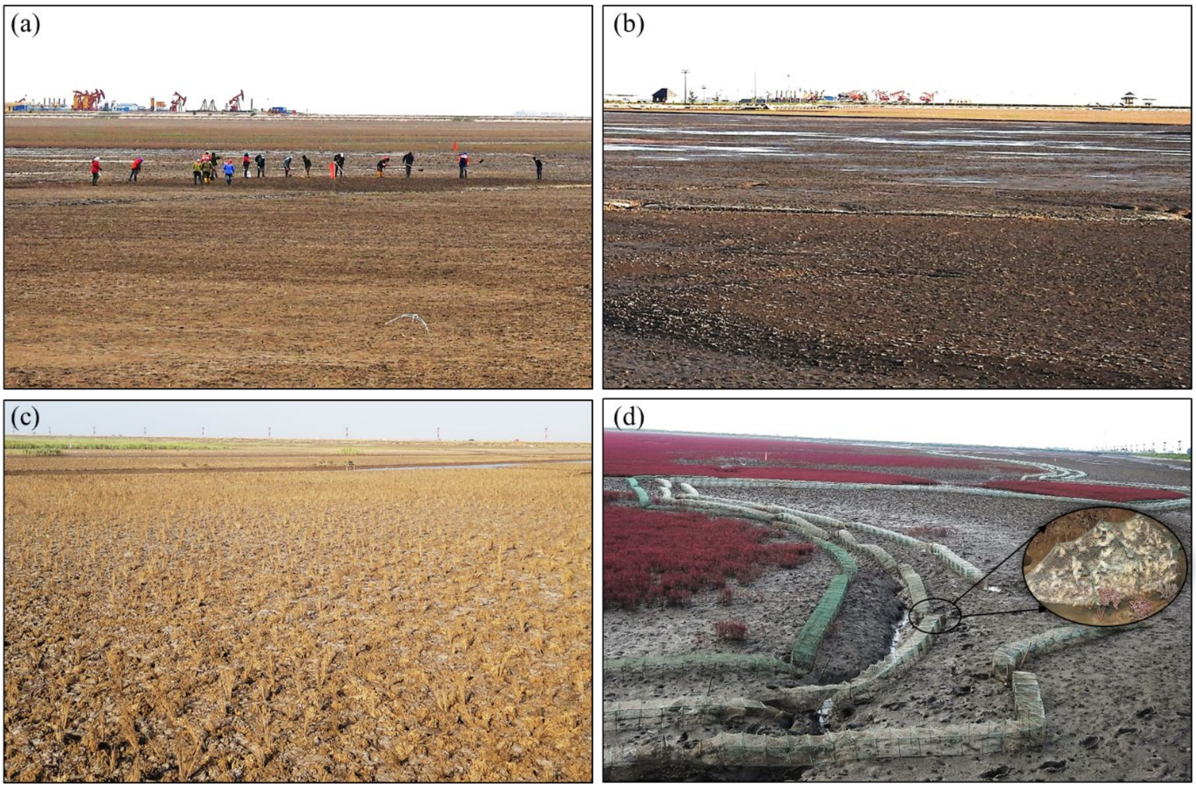

Figure 2. Restoration practices in the Liaohe River estuary. (a) Planting seeds of *Suaeda salsa* and (b) related outcomes. (c) Transplanting of *Suaeda salsa*. (d) Controlling herbivore crabs with cages. Photographs were taken by Zezheng Liu.

In this study, we used long-term remote sensing and ground-measured data to (1) evaluate the dynamics of the vegetation communities in the Liaohe River Estuary from 1994 to 2019, and (2) identify the key drivers affecting the sudden, catastrophic ecosystem die-offs and recovery of vegetation communities. Based on the results, we propose an alternative restoration and management solution. Knowledge of the restoration outcomes in the Liaohe River Estuary can help other regions avoid similar problems, and improve the restoration and protection of coastal ecosystems before too many of them are lost.

2. Study site and methods

2.1 Site description

The Liaohe River Estuary (LRE) is located in the Liaodong Bay, a part of the Bohai Sea, northeastern China. LRE was designated a Ramsar Site of International Importance that provides suitable habitat for resident birds and an excellent stopover point for migratory birds on the East Asian-Australasian Flyway (Figure 3b, https://rsis.ramsar.org/ris/1441). More than 70% Saunders's Gull *Larus saundersi* in the world have been recorded here. This gull is in the IUCN Red List of vulnerable species (Zhou et al., 2017). This area has a typical temperate semi-humid monsoon climate, with annual precipitation of 623 mm and evaporation of 1669 mm, and a mean annual temperature of 8.4°C (Liu et al., 2018). The study area experiences an irregular and semidiurnal tide, with an average tidal range of 2.7 m, and tidal currents mainly directed from northeast to southwest (Zhu et al., 2010).

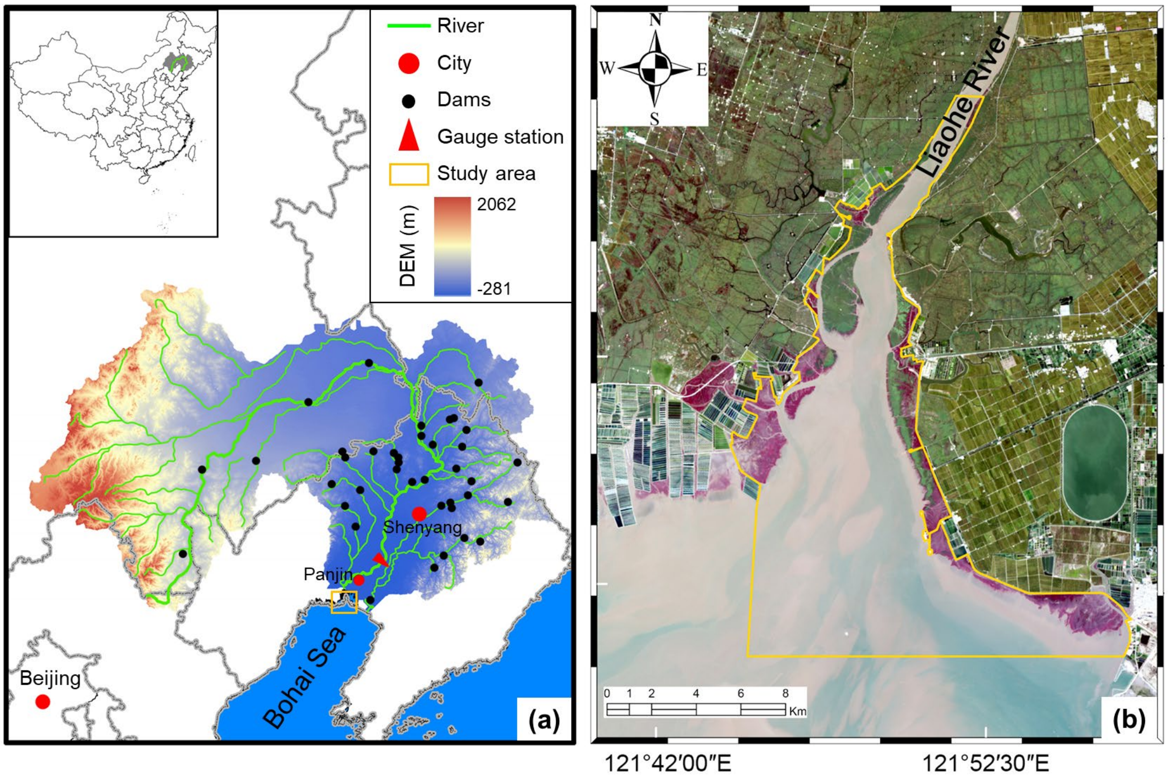

Figure 3. Location of the Liaohe basin in China (a) and study area in the Liaohe Estuary with a true color remote sensing image on September 15, 2014 (b). Locations of major dams are extracted from Lehner et al., (2011).

The Liaohe River Basin is one of the seven major basins in China. It is located in the lower part of northeastern China, flowing through four provinces (Inner Mongolia Autonomous Region, Hebei Province, Jilin Province, and most parts of the Liaoning Province) before it finally discharges into the Bohai Sea (Figure 3a). The terrain of the Liaohe River Basin is relatively flat, with a length of 1390 km and a drainage area of 228,960 km^2. The river supplies water to about 34 million people (Table S1, MWR 2019). The mean annual runoff per capita is 371 m^3/person, which is the second lowest among the seven major river basin of China (Table S1, MWR 2019). The streamflow regime has been dramatically altered by the construction of numerous upstream dams and diversions of water for agricultural and municipal uses (Figure 3 a; Ren et al., 2002). The middle and lower reaches of the Liaohe River Basin form the vast Liaohe River Plain. Most areas of the plain belong to a temperate semi-humid and semi-arid climate .The plain is also an important industrial, agricultural and commercial center in northeastern China, with more than one million acres of irrigated paddy fields (Ma et al., 2016, Xu et al., 2019). More than 1100 reservoirs ha ve been built in the Liaohe River Basin with a total storage capacity of 44.9 km^3, and the total number of dams and water gates in this basin are 689 and 2071, respectively (Dai et al., 2009; MWR, 2019).

The two dominant plant species in the study area are virtually monospecific stands of the annual forb *Suaeda salsa* (L.) Pall. with purple leaf color, and the perennial grass *Phragmites australis* with leaf color of green (Figure 1a). *Suaeda salsa* is known to be more tolerant to salinity and water stresses than *Phragmites australis* (He et al., 2012). However, *S. salsa* is susceptible to grazing by the herbivorous crab *Helice tientsinensis* Rathbun, which is abundant in this salt marsh (Liu et al., 2020a). *Suaeda salsa* marshes (named Red Beach) not only provide an important habitat for invertebrates and birds (e.g. the gull *Larus saundersi*), but are also important for tourism and recreation, offer job opportunities, and drive economic development (Bardzinska-Bonenberg and Liu, 2019; Song et al., 2020). The total value of the Red Beach for recreation services is more than one hundred million dollars per year (Song et al., 2020). Unfortunately, *Sueada salsa* vegetation in the Liaohe Estuary dramatically declined since Fall 2015, transforming the famous red beach to a bare flat (Figure 1; Liu et al., 2020b).

2.2 Measurement of marsh vegetation dynamics

We use remote-sensing (Landsat) imagery to map marsh vegetation variability in the LRE from 1994 to 2019. The boundary of our study area was determined by a series of Google Earth aerial photographs and Landsat remote sensing images, and the study area was almost free of new land reclamation project from the 1990s to 2019 (Figure 3b). To improve the recognition accuracy and minimize the effects of seasonal changes, all selected Landsat images were taken at low tide on clear, cloud-free days during August-October (Table S2). ENVI 5.3 Imagine software was used to process all standard corrections (e.g. radiation correction, atmospheric correction and geometric correction). A classification tree method was developed to classify *S. salsa*, *Phragmites australis* and un-vegetated areas by ENVI 5.3 Imagine software (Figure S2; Jia et al., 2015; Qiao et al., 2018). To improve the classification accuracy, we corrected some misclassifications based on visual interpretation of true color remote sensing images, because these three types of land cover are easy to be distinguished (Liu et al., 2020b). To better determine the growth condition of *Phragmites australis*, we calculated the average Normalized Difference Vegetation Index (NDVI) at random points, which were generated using the random point tool in ArcMap 10.2 with the minimum allowed distance of 100 m inside a 100 m buffer of *Phragmites australis* zone.

2.3 Data sources of key environmental variables

Annual and monthly water and sediment load data at Liujianfang gauging station in the Liaohe river basin were collected from the Bulletin of China River Sediment (http://www.mwr.gov.cn/sj/#tjgb) and hydrological data in the Liaohe river basin from hydrological reports. Liujianfang gauging station is the nearest gauge station to the sea along the Liaohe River, and can reflect the total discharge of water and sediment to the estuary (Figure 3a; Zhao et al., 2017). Annual and monthly average air temperature and precipitation in Panjin City, where the Liaohe River discharges into Bohai Sea (Figure 3a), were collected from the Liaoning Statistical Yearbook (http://www.ln.stats.gov.cn/).

The water quality in the Liaohe river at Xing'an station in Panjin City was collected from the Data Center of the Ministry of Ecology and Environment of the People's Republic of China from 2004 to 2018 (https://datacenter.mee.gov.cn/websjzx/queryIndex.vm). The indicators of water quality used here are dissolved oxygen (DO), potassium permanganate index (COD_{Mn}), ammonia nitrogen (NH_3-N). Concentrations of major pollutants (petroleum and heavy metals) in the Liaohe river were collected from Bulletin of China's marine ecological environment status (http://www.chmem.cn/gjgb/index.htm). Before 2004, petroleum flux into the sea from the Liaohe river was retrieved from Wang et al., (2011). Additionally, the annual average sea surface salinity in the Liaohe estuary was collected from Bulletin of China's marine ecological environment status. The monitoring of water quality strictly follows the national Environmental Quality Standards for Surface Water (GB3838-2002).

2.4 Data analysis

Environmental factors have high legacy effects on plant growth. For example, previous-year precipitation affects current-year net primary production (Sala et al., 2012; Reichmann et al., 2013; Monger et al., 2015), environmental variables in our study were smoothed by a 3-year moving average to represent the current-year situation. For example, a 3-year moving average of annual precipitation in 2018 in the regression analysis is represented by the average annual precipitation of 2016, 2017 and 2018. In the same way, a 2-year moving average was calculated. To reduce the impact of marsh accretion and erosion on the regression analysis, we only analyzed the relationships between environmental variables and vegetation growth during 1994-2001 and 2010-2019, because the increase in area of *Suaeda salsa* during 2001-2010 is mainly due to the formation of a river mouth bar rather than marsh recovery (Figure 5b). The relationships between area of *Suaeda salsa,* NDVI of *Phragmites australis* and key environmental factors were analyzed with a linear regression model using the SPSS 22.0 statistical software package (IBM, Armonk, New York, USA). The relationships between annual runoff and precipitation and mean sea surface salinity were also analyzed based on the same linear regression model. The level of statistical significance was set at $p < 0.05$.

3. Results and discussion

3.1 Spatial and temporal dynamics of marsh vegetation community

Using satellite-based data, our analysis indicates that marsh vegetation community in the LRE have undergone a dramatic change from 1994 to 2019 (Figure 4 a). We divided the dynamics of marsh vegetation community over 26 years into four distinct periods (Figure 4 a). Period I (1994-2001): The area of *Suaeda* and *Phragmites* was relatively stable during 1994-

1999. Dramatically, 72% of the 1999 *Suaeda* population and 69% of *Phragmites* population were rapidly lost from 1999 to 2001 (Figure 4 a; Figure 5 a). Period II (2001-2010): Although the total area of *Suaeda* has increased fivefold (Figure 4 a), the new encroached area was mostly located in a newly formed river mouth bar rather than in the 1999-2001 degraded area (Figure 5 b). In fact, the 1999 *Suaeda* area was still shrinking, and *Phragmites* population almost disappeared in this period. Period III (2010-2014): *Suaeda* and *Phragmites* populations have increased considerably (by 72% and 2100%, respectively), not only on the river mouth bar, but also in the previously degraded areas (Figure 5 c). In the area near the river mouth, more than 700 ha of *Suaeda* population was converted into *Phragmites* population (Figure 5 c). Period IV (2014-2019): 77% of the *Suaeda* population was suddenly lost in this period. Although the area of *Phragmites* population did not show any apparent change, its Normalized Difference Vegetation Index (NDVI), representing vegetation biomass, has undergone a precipitous decline within the past five years (Figure 4 a).

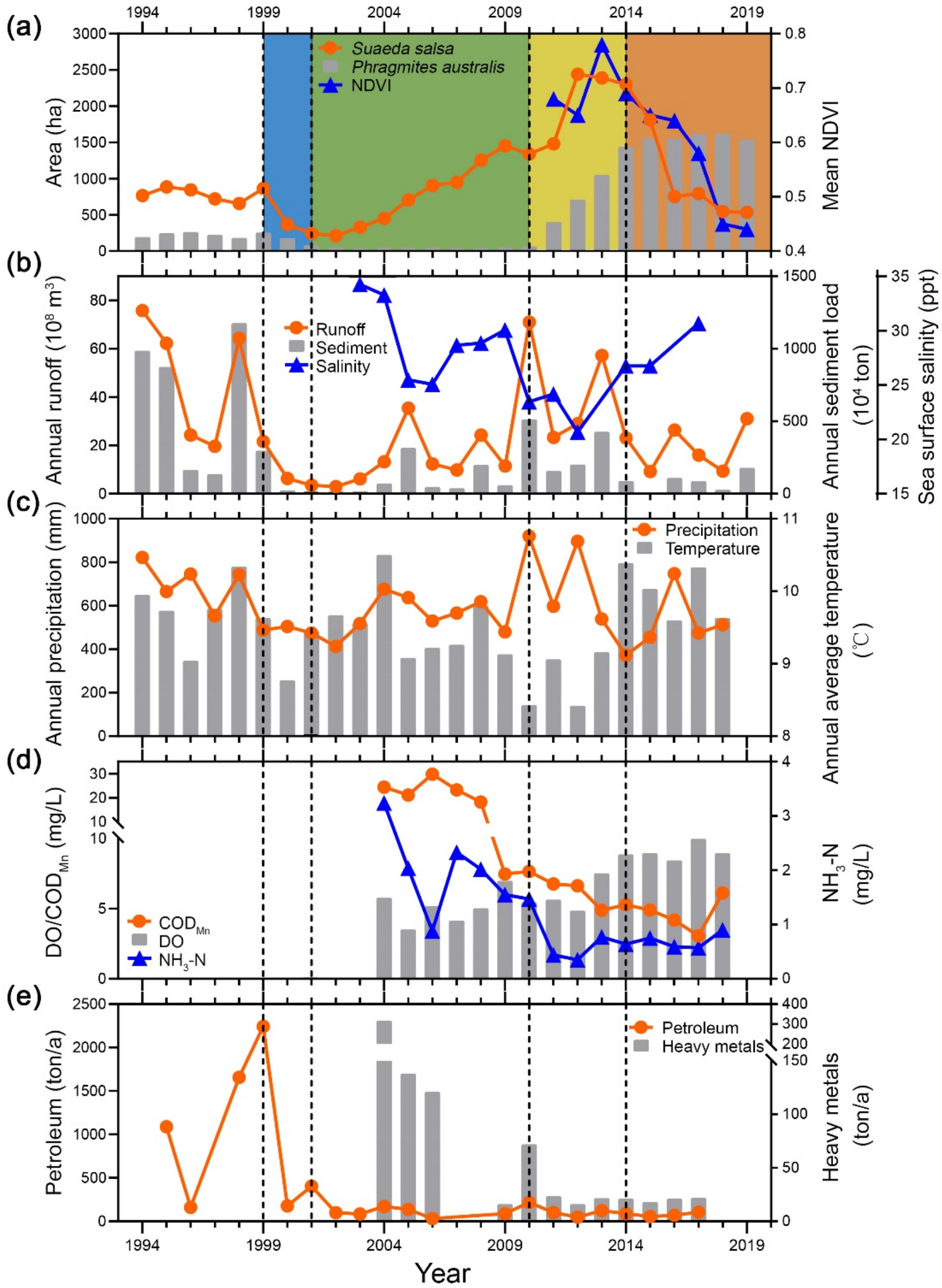

Figure 4. Time series showing variations in vegetation cover and key environmental variables.

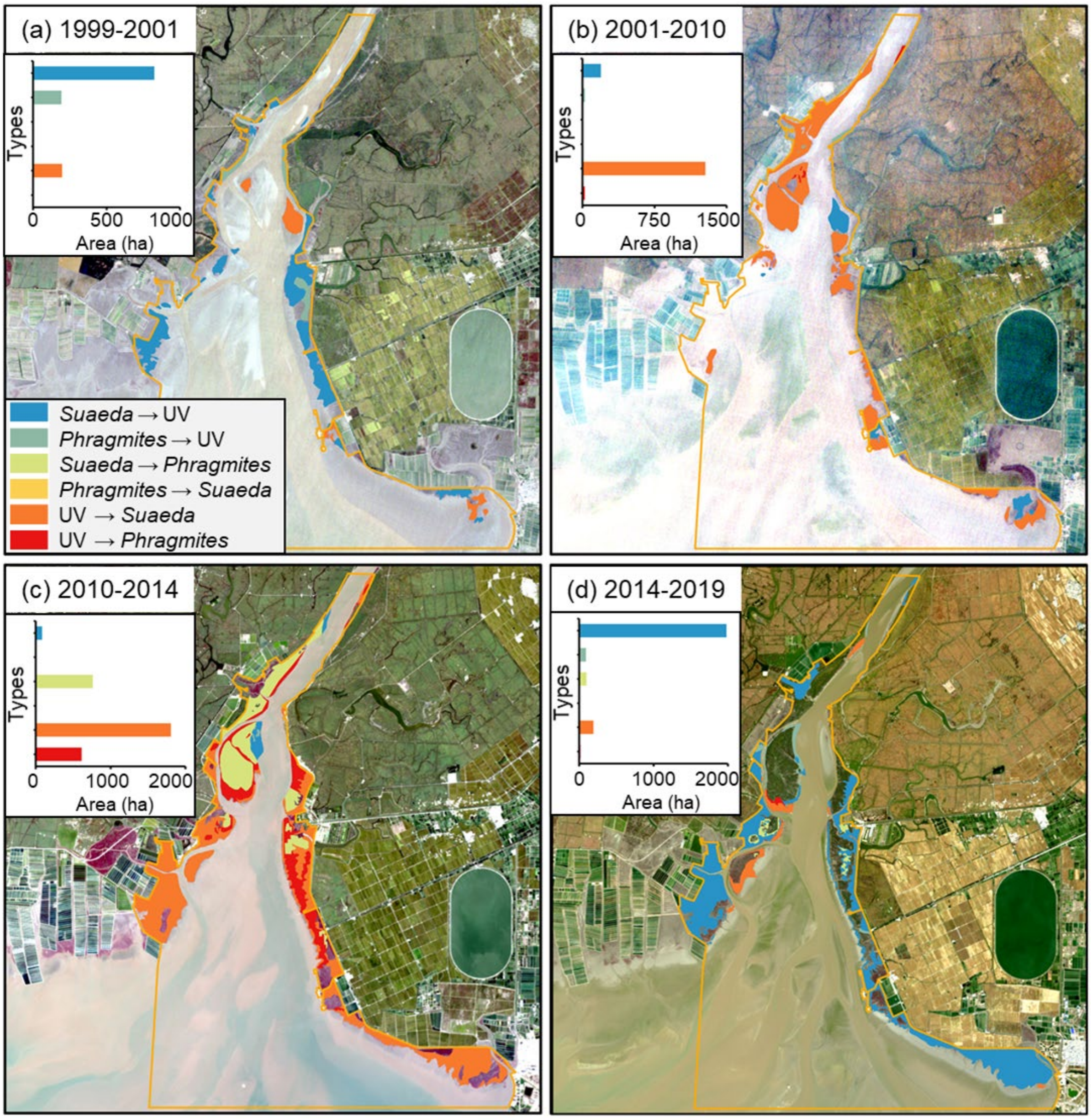

Figure 5. Conversion of marsh vegetation in (a) Period I (1994-2001), (b) Period II (2001-2010), (c) Period III (2010-2014) and (d) Period IV (2014-2019). UV means un-vegetation area, such as bare land, tidal flat and water.

3.2 key drivers of marsh vegetation change

Our analysis indicates that the dynamics of marsh vegetation community in the LRE were largely attributed to river runoff, precipitation and sea surface salinity, but not to water quality, nutrient and pollutants inputs (Figure 6; Figure S5). 2-year moving windows also were examined and produced results similar to those from the analysis of 3-year moving windows (Figure S4 and S5 b). Increases in river runoff and precipitation as well as a decrease in sea surface salinity were significantly correlated to increases in total area of *Suaeda* population during 1994-2001 and 2010-2019 (Figure 6 a-c). Similarly, increases in river runoff as well as a decrease in sea surface salinity were significantly correlated to increases in NDVI of *Phragmites* population in period IV (2014-2019) (Figure 6 d, f). In 2010, for example, when river runoff and precipitation were relatively high, with relatively low sea surface salinity, areas of *Suaeda* and *Phragmites* populations began to increase. Conversely, in 2014, when runoff was lower and sea surface salinity higher, the population of *Suaeda* and *Phragmites* began to rapidly degrade. Similarly, during the period of *Suaeda* degradation in 1999-2001, river runoff and precipitation were at a low level (Figure 4, b and c). Furthermore, correlations between precipitation and NDVI of *Phragmites* population as well as sea surface salinity were sensible in sign, but none were significant (R^2 = 0.085, P = 0.484; R^2 = 0.246, P = 0.085) (Figure 6 e and h). However, increased river runoff was significantly correlated to decreased sea surface salinity (R^2 = 0.326, P = 0.042) (Figure 6 g).

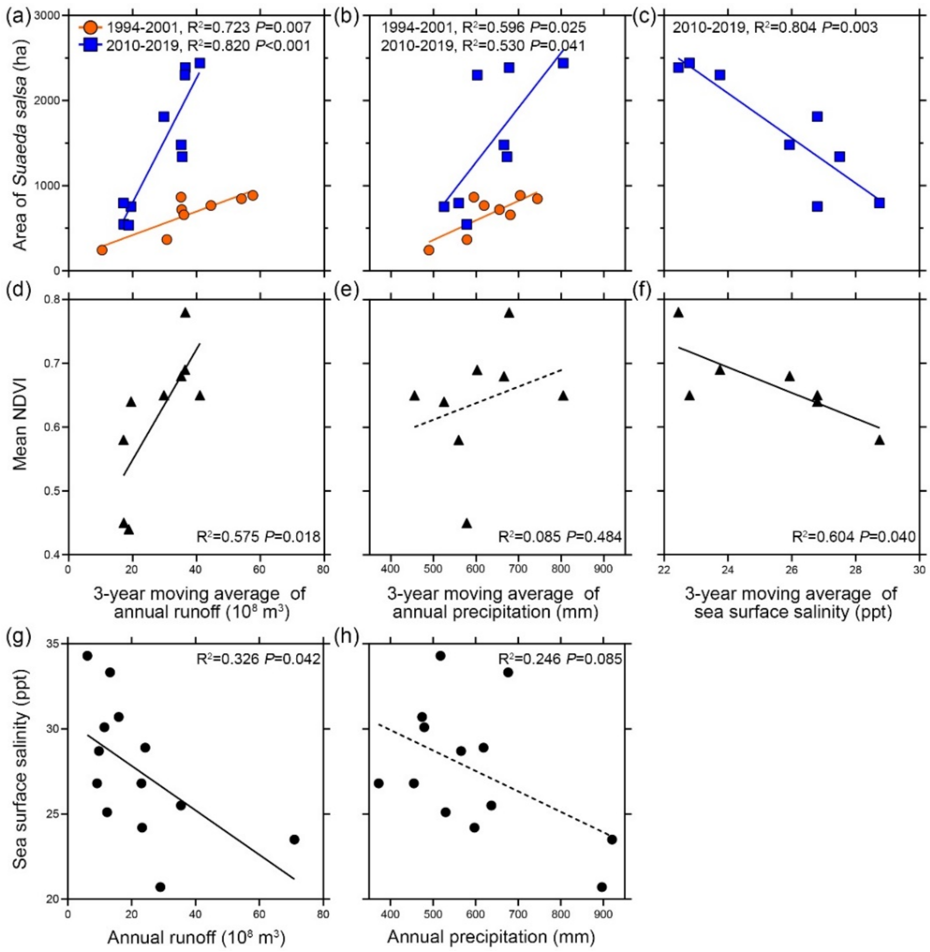

Figure 6. Relationships between vegetation growth and key environmental factors (a-f).

Relationships between freshwater availability and sea surface salinity (g and h). Solid lines denote significant correlations ($P \leq 0.05$), whereas dotted lines denote non-significant correlations.

Freshwater availability controls the dynamics of marsh vegetation community through abiotic environmental stresses and consumer control (He et al., 2017; Liu et al., 2020a). Increased river freshwater inflow can reduce sea surface salinity in the estuary (Figure 4 g; Lin et al., 2001). Therefore, *Suaeda* population was replaced by brackish water *Phragmites* population during 2010-2014, and the impact of river inflow tapered away with distance from the mouth (Figure 5 c). In contrast, reduced freshwater inflow can increase salinity in the water column, resulting in the degradation of marsh vegetation community (Liu et al., 2020a). Consistent with earlier studies (Silliman et al., 2005), a reduction in freshwater also triggers the outbreak of plant-eating grazers *Helice tientsinensis* Rathbun in the temperate coastal saltmarshes of China, which can eliminate drought-stressed *Suaeda salsa* (He et al., 2017). Our previous study in LRE also found that abundance and herbivory strength of *Helice tientsinensis* is higher in the high marsh with low flooding frequency than in the low marsh with high flooding frequency (Liu et al., 2020a).

It is difficult to distinguish the contributions of river discharge and precipitation to the dynamics of marsh vegetation community. Since the correlation between river runoff and growth of marsh vegetation is much stronger than precipitation (compare Figure 4 a and 4b, 4d and 4e), the dynamics of marsh vegetation community could be mainly attributed to river runoff. Furthermore, sea surface salinity has a significant relationship with annual river runoff, but not with annual precipitation (Figure 6 g and h), which is consistent with earlier study suggesting that the increase in sea surface salinity of the Bohai Sea is principally caused by a decrease in total river discharge into the Bohai Sea, not by precipitation (Lin et al., 2001). In addition, from 1985 to 2018, there is a significant tendency towards a decrease in the series of annual river runoff events observed ($R^2 = 0.181$, $P = 0.012$), however, no significant trend was identified in the series of annual precipitation ($R^2 = 0.084$, $P = 0.097$) (Figure S 6). The similar trends were observed in the upstream of the Liaohe River Catchment from 1955 to 1998 (Ren et al., 2002). From the perspective of ecosystem management, precipitation cannot be controlled, however, freshwater runoff can be replenish by river regulation (Cui et al., 2009). In the context of coastal wetlands protection and restoration, therefore, the role of river runoff in the dynamics of marsh vegetation community should deserve managers' attention.

Nutrient and pollutants inputs may also influence the dynamics of marsh vegetation community in estuary ecosystems. Ruhl and Rybicki (2010) suggested that long-term reductions in anthropogenic nutrients enhanced the abundance and diversity of submerged aquatic vegetation community in the Potomac River Estuary, a major tributary of the Chesapeake Bay. Deegan et al., (2012) found that nutrient enrichment drives the loss of salt marsh in Plum Island Estuary, Massachusetts, USA. Huang et al., (2012) also demonstrated that the concentrations of heavy metals Fe and Cu in the surface water is negatively correlated with the biomass and density of wetland vegetation in the Pearl River Estuary of China. Our previous study in the LRE, however, indicated low levels of heavy metal pollution in surface sediments, and there was no significant difference in heavy metal concentration between bare land and *Suaeda* population in 2016 (Liu et al., 2018). Moreover, the export fluxes of COD_{Mn}, NH_3-N, petroleum and heavy metals in the Liaohe River decreased in the study period due to the implementation of water pollution control (Figure 4 d and e; Sun et al., 2015). Pollutant fluxes did not show any apparent change during the massive marsh degradation in 2014 (Figure 4 d and e). This suggests that water quality, nutrient and pollutants inputs may not be a significant factor in marsh dieback in this study area.

Accelerated sea level rise and erosion by storm surges are increasingly regarded as one of the greatest threats to coastal wetlands (Gedan et al., 2009; Fagherazzi et al., 2020). However, in the LRE the average rates of vertical accretion/elevation change in the *Phragmites* and *Suaeda* marshes are 17.5/9.05 mm yr^{-1} and 14.2/6.05 mm yr^{-1}, respectively. These marshes thus keep pace or gain elevation with respect to local rate of relative sea level rise (2.4–5.5 mm yr^{-1}) (Wang et al., 2016). The frequency and intensity of storm surges in the north Bohai Sea, a shallow semi-enclosed sea with an average depth of only 18 m, were small and did not show any apparent increase during the massive marsh degradation in 2014 (Feng et al., 2018). Consistent with earlier studies (Liu et al., 2020b), *Suaeda* marsh begins to degrade in areas away from the marsh seaward edge and the river inlet (Figure S 3). We therefore believe that sea level rise and storm surges have not been a major driver in the marsh dynamics in the LRE.

In addition, *Spartina* spp. invasion was also seen as a contributing factor in the reduction of coastal wetlands surface in China (Sun et al., 2015), however, *Spartina* species were not found in this study area.

4. Toward watershed-based coastal restoration

Coastal estuarine ecosystems are located at the ecotone between a river and the sea environment and are subject to powerful continental-oceanic interactions (Elliott and McLusky, 2002; McLusky and Elliott, 2004). Estuarine ecosystems are often influenced by material inputs from river basins, such as freshwater, sediments, nutrients and pollutants. For example, the seawater is measurably diluted with freshwater flowing into the estuary from rivers, and the pattern of dilution of seawater varies with seasons and locations, largely depending on river discharge (Lin et al., 2001; Gibson et al., 2002). The stability and productivity of the estuaries on the Mexico's Pacific coast were demonstrably affected by a sediment load reduction caused by hydroelectric dams in the river upstream (Ezcurra et al., 2019). The diversity and abundance of native species of submerged aquatic vegetation in the Potomac River Estuary increased due to anthropogenic nutrient inputs reduction (Ruhl and Rybicki 2010). Despite growing recognition of watershed-scale effects, few estuarine restorations focus on this mismatch between ecological drivers at the watershed scale and local restoration efforts at the estuary scale (Sayles and Baggio, 2017). Neglecting the role of material inputs from the river basin can undermine estuarine restoration projects (Liu et al., 2020a). Therefore, estuary restoration must be coordinated among multiple government and non-government agencies operating from across scales and in different locations throughout a watershed (Baird 2005; Crowder et al., 2006; Cumming et al., 2006; Sayles et al., 2018).

From a global perspective, freshwater transport from rivers to the coastal ocean experienced significant changes in recent decades. The cumulative discharge from many mid-latitude rivers decreased by 60%, due to climate change, damming, irrigation and inter-basin water transfers (Milliman et al., 2008; Li et al., 2020). Estuarine ecosystems in mid-latitude regions have also experienced extensive loss, degradation and fragmentation (Gedan et al., 2009; Murray et al., 2019). Great efforts to restore coastal ecosystems have been conducted in several estuaries around the world, such as in the Hunter estuary in Australia (Howe et al., 2009), Scheldt

Estuary in Belgium, the Huelva estuary in Spain (Castillo and Figueroa, 2009), the Tijuana (Zedler and West, 2008; Zedler, 2010) Nisqually (Ellings et al., 2016) and Hudson estuaries (Montaltoet al., 2006) in USA, the Yangtze (Chen et al., 2017), Beilunhe (Ning et al., 2014), Jiulongjiang (Chen and Ye, 2011) and Liaohe Estuaries (Liu et al., 2020a) in China. However, the role of river freshwater inputs was always overlooked in restoration projects.

Therefore, watershed-based restoration and protection of coastal estuaries, incorporating river basin regulation in estuarine restoration, should be taken into consideration. To the best of our knowledge, however, this solution does not get the widespread attention of managers and is implemented only in few estuaries, such as the water-sediment regulation scheme in the Yellow River Estuary (Cui et al., 2009; Gao et al., 2019) and artificial channel diversions in the Mississippi River Delta (Day et al., 2007; Paola et al., 2011). Vegetation communities quickly re-established after freshwater replenishment in the Yellow River Estuary, providing favorable habitat conditions for invertebrates and waterfowl (Cui et al., 2009). Following the historic 2011 Mississippi River flood, the Wax Lake Delta gained land area and vegetation communities changed from fully submerged to emergent herbs (Carle et al., 2015). Watershed-based solutions may be a more cost-effective estuarine restoration technique than local expensive restoration projects involving transplanting, planting seeds, or the construction of artificial habitats.

Based on the above discussion, we recommend that estuarine restoration projects should be planned at the river basin scale. Managers should try to increase river baseflow to the wetlands of the LRE, as well as to other estuaries worldwide facing similar problems. Water replenishment to the Liaohe River Estuary is feasible: for example, more than 100 million m^3 freshwater in the Chaihe and Qinghe Reservoirs were added to the Panjin and Yingkou branches to irrigate paddies during the drought period of July 2020 (Liaoning daily, 2020). To counteract the ecological degradation of the Xianghai wetland – a national natural reserve in the Liaohe-Songhuajiang River basins - the Songliao Water Resources Commission diverted 80 million m^3 of freshwater (Ministry of Water Resources of the People's Republic of China 2020). The volume of replenished freshwater should meet the sustainable environmental and ecological requirements in the LRE, which was estimated to be between 1 to 1.5 billion m^3 per year (Zhang et al., 2010; Zhang et al., 2017). This volume should be adjusted accounting for the annual precipitation. Time of water replenishment should be in the early stage of the growing season of plants (May- August) (Figure 7), for example, water is replenished in the Yellow River estuary in June (Cui et al., 2009). Therefore, water replenishment must be coordinated among multiple government and non-government programs to balance reservoir functioning and water allocation for multiple targets (Mao et al., 2020). In addition, current velocity in the river should be controlled within a certain threshold in the periods of replenishment, because water-imposed drag forces influence the establishment of seedling and may lead to marsh erosion (Marani et al., 2011; Cao et al., 2020). Sediment loads due to replenished water also should be taken into consideration, which plays an important role in estuarine geomorphologic evolution and vegetation community change (Carle et al., 2015; Gao et al., 2019). Admittedly, the watershed-based coastal restoration strategy discussed above is subject to numerous simplifications, but it lays a foundation for future practical implementations. Further studies are needed in this area.

5. Conclusions

Here, we present the results from a 26-years investigation (1994-2019) of marsh vegetation community dynamics and their relationships to key environmental factors in the Liaohe River Estuary of China. We found that vegetation community has experienced two dramatic diebacks (72% and 67% decrease in vegetation area) and one recovery period (82% increase in vegetation area). Rapid changes in saltmarsh extension were primarily driven by freshwater availability, especially river discharge, not by water quality, nutrient and pollutants inputs, sea level rise, and biological disturbances. Our results suggest that a reduction or increase in river freshwater inputs do result in a deteriorating or improving habitat, affecting marsh extension and vegetation biomass. Without increasing river runoff, efforts to restore the marsh by adding propagules and controlling grazers produced limited results or even failed. Given the notably declining trends in freshwater fluxes from rivers to the ocean in low and middle latitude regions, such changes have probably also occurred in many other estuaries around the world. Therefore, we call for managers to pay attention to this scale mismatch, and restoration and management of estuarine wetlands in the future should be planned at a watershed scale rather than at a local scale.

CRediT authorship contribution statement

Competing interests

The authors declare no competing interests.

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Supplementary material

Table S1. Runoff of the Seven Major Rivers in China. Mean annual runoff refers to the average annual value of the past 50 years. Data is from MWR 2019.

River	Length (km)	Drainage Area (km^2)	Mean Annual Runoff ($10^8 m^3$)	Population (10^8)	Annual Runoff per Capita (m^3/person)
Yangtze River	6300	1808500	9280	3.79	2449
Yellow River	5464	752443	628	0.92	683
Songhua River	2308	557180	733	0.51	1437
Liaohe River	1390	228960	126	0.34	371
Pearl River	2320	453690	3360	0.82	1098
Haihe River	1090	263631	288	1.10	262
Huaihe River	1000	269283	611	1.42	430

Table S2. Information on the remote sensing images used in this study.

Satellite	Sensor	Data	Satellite	Sensor	Data
Landsat-5	TM	1994-09-08	Landsat-5	TM	2007-09-28
Landsat-5	TM	1995-09-27	Landsat-5	TM	2008-09-30
Landsat-5	TM	1996-10-15	Landsat-5	TM	2009-10-03
Landsat-5	TM	1997-10-18	Landsat-7	ETM+	2010-09-28
Landsat-5	TM	1998-10-05	Landsat-5	TM	2011-09-23
Landsat-5	TM	1999-10-08	Landsat-7	ETM+	2012-09-17
Landsat-7	ETM+	2000-09-16	Landsat-7	ETM+	2013-08-19
Landsat-5	TM	2001-09-27	Landsat-8	OLI	2014-09-15
Landsat-7	ETM+	2002-09-22	Landsat-8	OLI	2015-09-02
Landsat-5	TM	2003-10-19	Landsat-8	OLI	2016-09-20
Landsat-5	TM	2004-08-18	Landsat-8	OLI	2017-09-23
Landsat-5	TM	2005-09-22	Landsat-8	OLI	2018-09-10
Landsat-5	TM	2006-10-11	Landsat-8	OLI	2019-08-28

Figure S1. Fish (a-c) and birds (d) were trapped in cages.

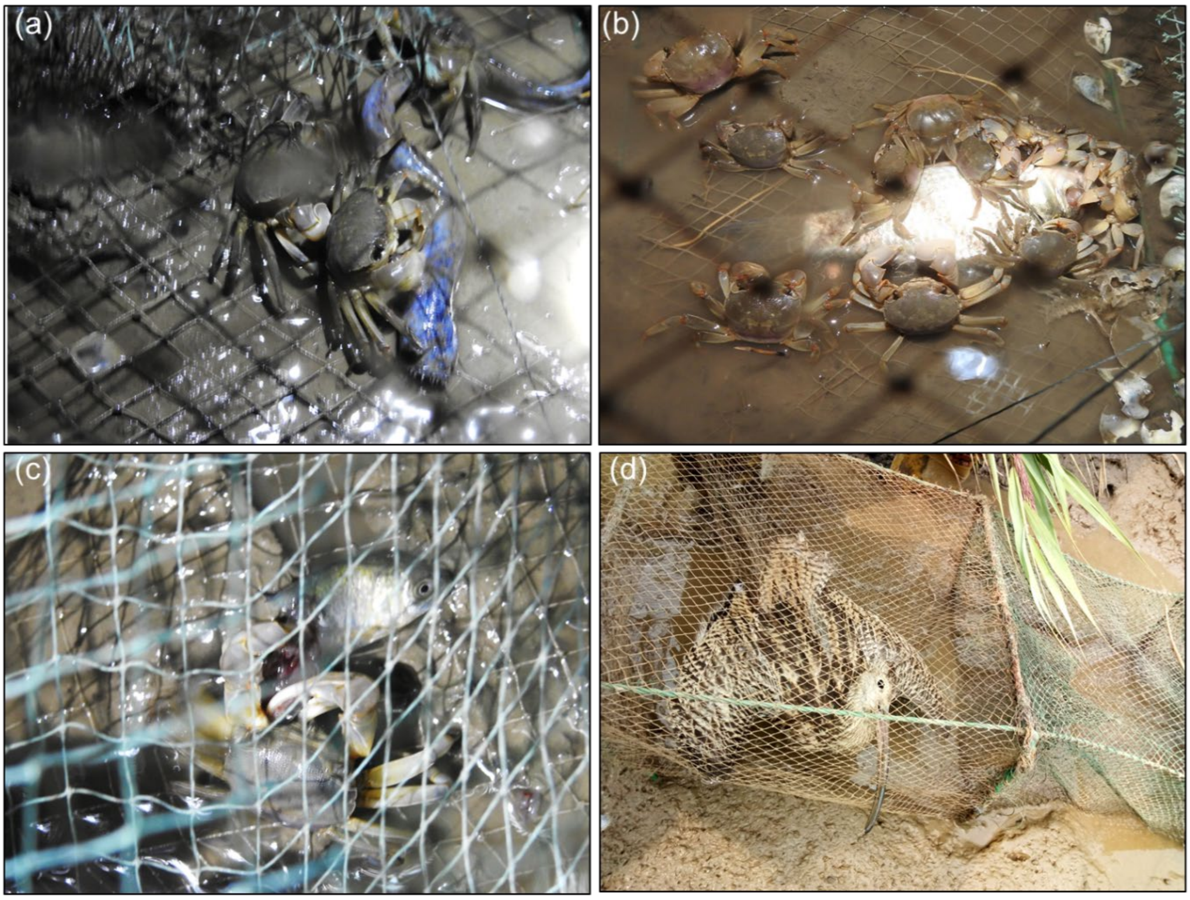
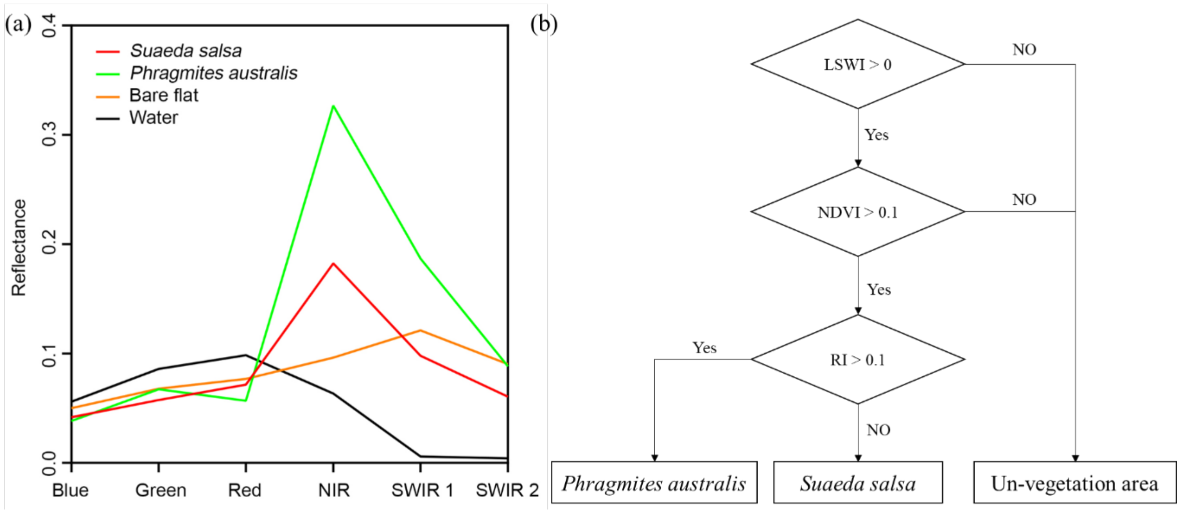
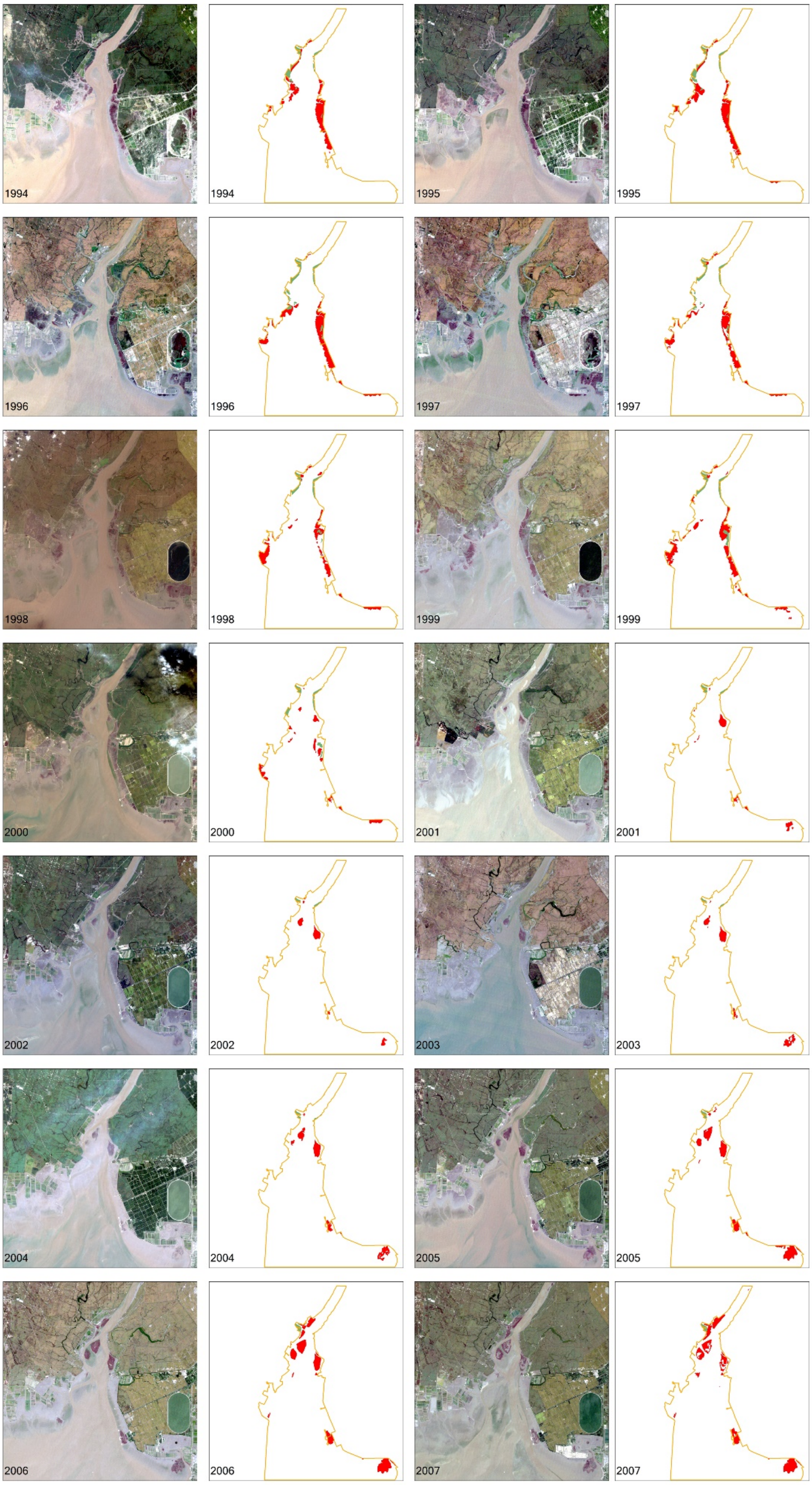


Figure S2. Reflectance of different land cover types (a) and classification tree used in this study. LSWI = (NIR – SWIR 1) / (NIR + SWIR 1), NDVI = (NIR - Red) / (NIR + Red), RI = (Red – Green) / (Red + Green). Green, Red, NIR and SWIR 1 are the reflectance values of green, red, near-infrared and short-wave infrared band, respectively.



656 Figure S3. distribution of marsh vegetation from 1994 to 2019.



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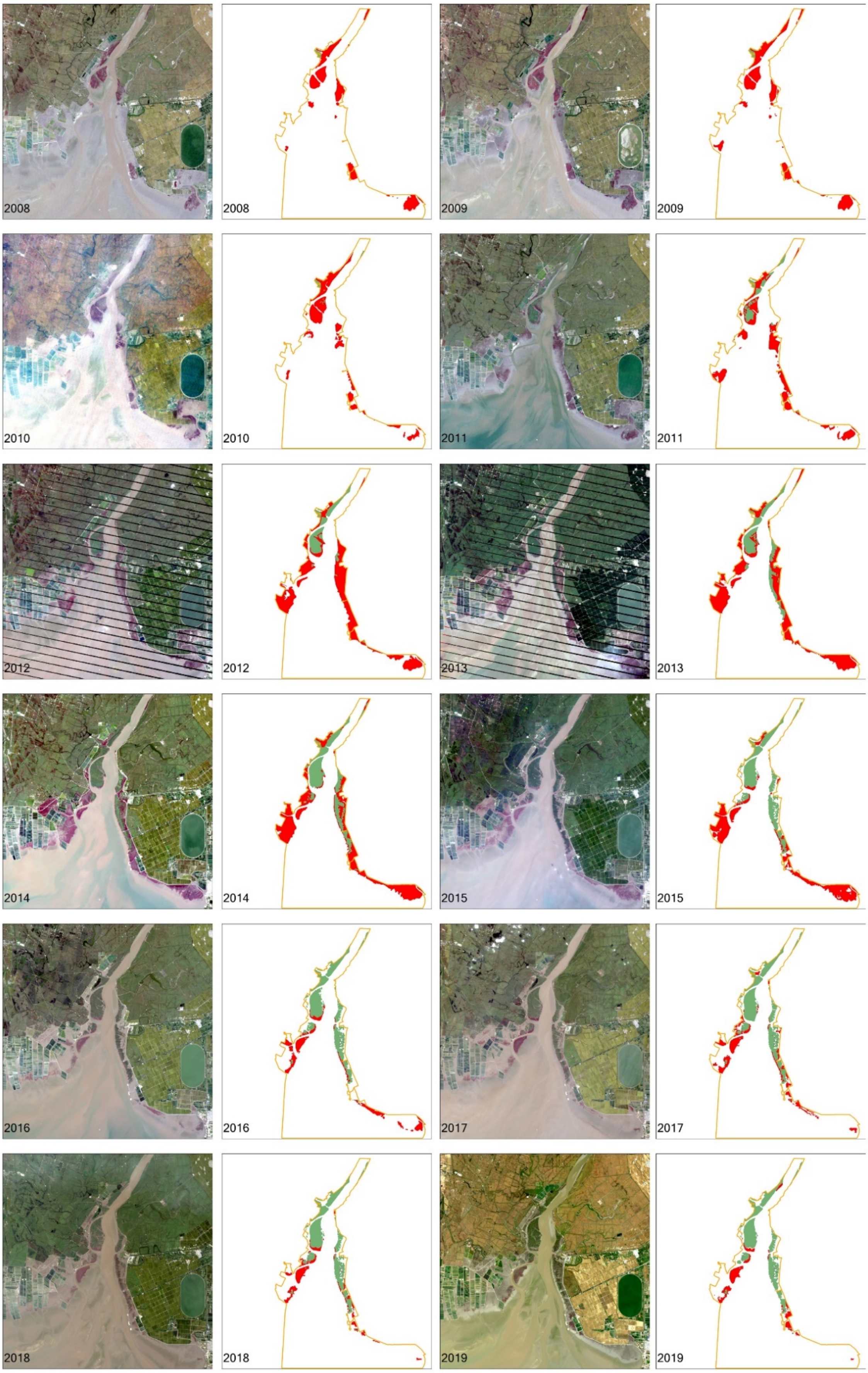
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Figure S4. Relationships between vegetation growth and river runoff (a and d), precipitation (b and e), sea surface salinity (c and f). Solid lines denote significant correlations ($P \leq 0.05$), whereas dotted lines denote non-significant correlations.

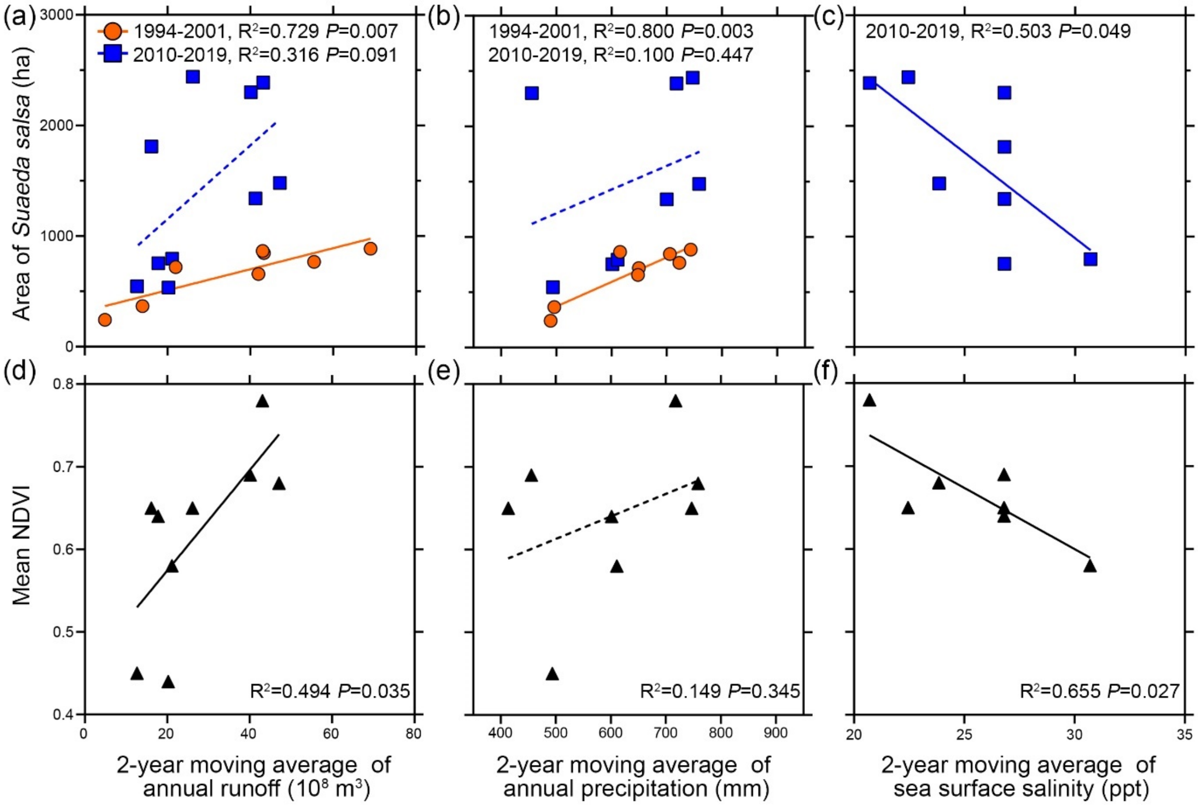

Figure S5. Relationship between vegetation growth and water quality, nutrient and pollutants inputs in 1-year moving windows (a), 2-year moving window (b) and 3-year moving window (c). Solid lines denote significant correlations ($P \leq 0.05$), whereas dotted lines denote non-significant correlations.

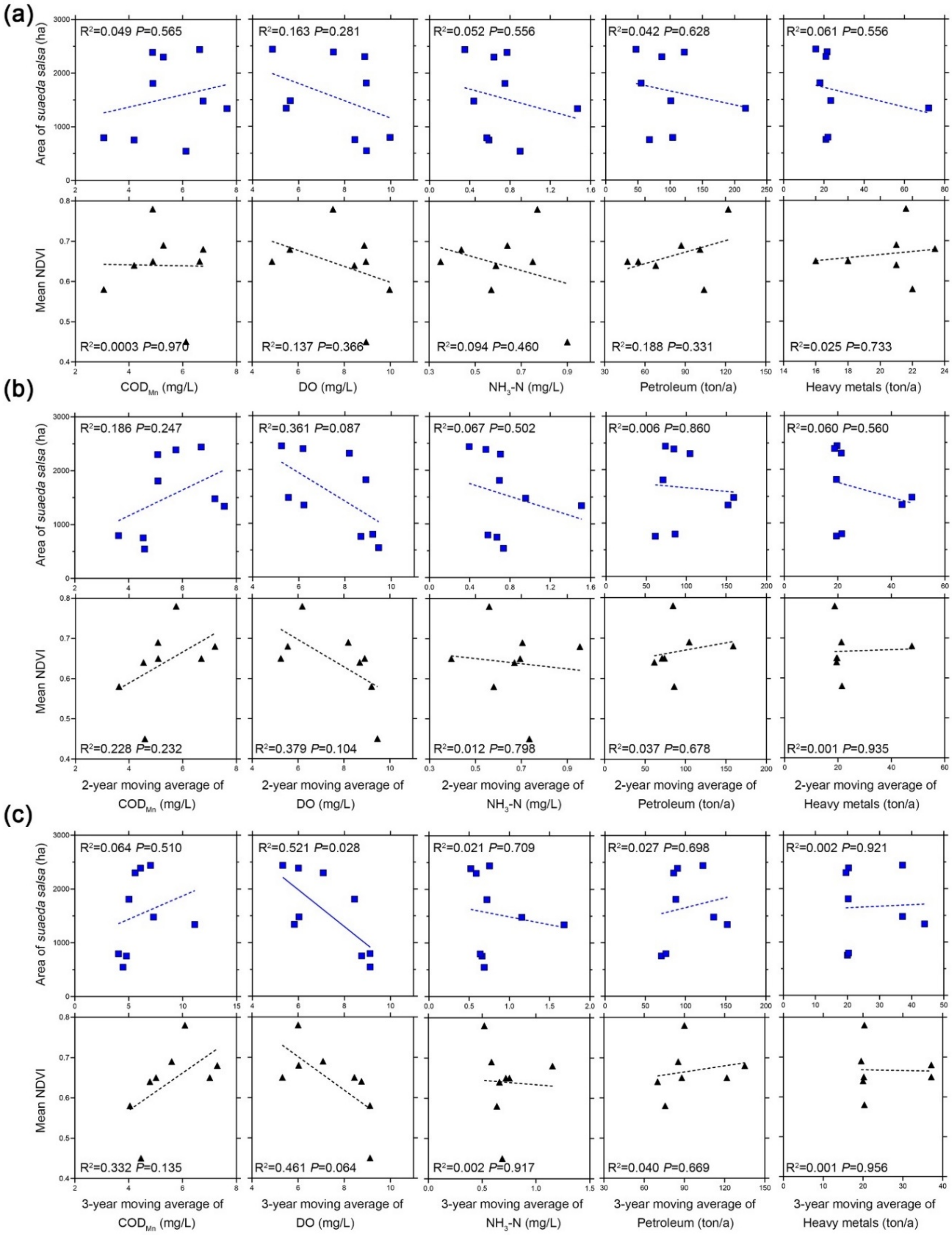

Figure S6. Annual precipitation (in Panjin), and observed river runoff (in Liujianfang Station) from 1985 to 2018. The solid line denote significant correlations ($P \leq 0.05$), whereas the dotted line denotes a non-significant correlation.

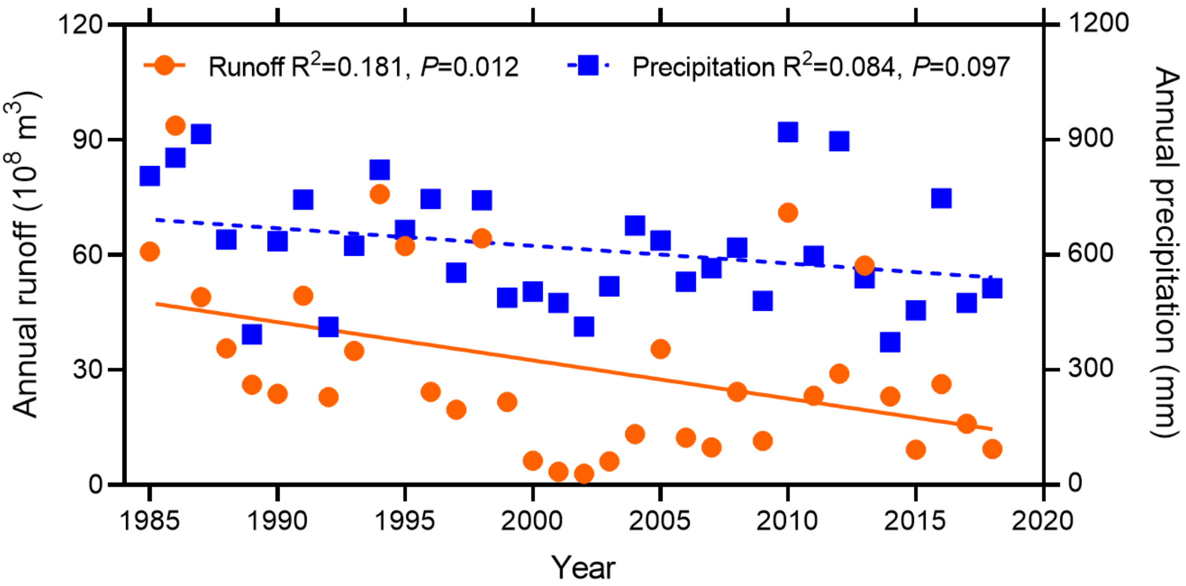

Figure S7. Monthly river runoff (a) and precipitation (b) in the Liaohe Estuary.