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Trade-offs among ecosystem services in coastal wetlands under the effects of reclamation activities

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ABSTRACT

In recent years, unsustainable reclamation activities have damaged coastal ecosystems and degraded ecosystem services, as a result, management of coastal wetland ecosystem services has received increasing attention. Because different ecosystem services respond differently to management, there are inevitable trade-offs among some of these services, and understanding these trade-offs has gradually become a research hotspot. As the research on trade-offs has deepened, it has become possible to quantify them. Based on the interpretation of remote sensing and socioeconomic data for the Yellow River Delta's coastal wetlands since 1989, the InVEST model was used to evaluate the spatial patterns and dynamic evolution of coastal wetland ecosystem services and the trade-offs among them. Production possibility frontier curves were then selected to identify optimal combinations of ecosystem service values; a trade-off intensity index was constructed to quantify the magnitude of the trade-offs, and the effects of reclamation activities on the trade-offs and synergies among ecosystem services were analyzed. A trade-off existed between material production and habitat quality from 1989 to 2015, and its intensity increased yearly. In addition, the relationship between carbon storage and material production transformed from a synergy into a trade-off in 2008, although the strength of the trade-off has been increasing since then. The results of our study will provide a scientific basis for improving wetland management and ecological restoration by alerting managers to the need to mitigate the tradeoffs.

1. Introduction

Wetlands are one of the world's most threatened ecosystems (MEA, 2005) for complex reasons, such as invasion by exotic species, water pollution, and human activities. Many of their problems result from their proximity to large human populations, which increasingly exploit the services provided by these rich ecosystems. In recent years, with accelerating economic development and the expansion of urbanization in coastal zones, reclamation of wetlands has become a significant problem (Murray et al., 2014). Large-scale reclamation activities provide large socioeconomic benefits, but also jeopardize the coastal wetland ecosystems and the services they provide through changes such as habitat fragmentation and habitat loss (Bulleri and Chapman, 2010). To maximize the benefits from wetland use while maintaining the ecosystem's health, wetland managers must balance the competing needs of socioeconomic development and conservation. Because this balance inevitably involves trade-offs, trade-off analyses have become increasingly necessary. Quantifying these trade-offs (and the synergies that sometimes emerge) can improve the management and protection of coastal wetlands by revealing the optimal allocation of wetland services.

Trade-offs among multiple ecosystem services occur when an improvement in one ecosystem service is achieved at the expense of a decrease in another; conversely, when an improvement in one ecosystem service leads to an increase in another, the relationship is a synergy. Trade-offs generally exist between provisioning services, between provisioning services and regulating services, and between supporting services (Chisholm, 2010; Martín-López et al., 2012). Synergies exist between all four categories of ecosystem services, and are relatively common in regulating, supporting, and cultural services (Lamsal et al., 2015). Trade-offs or synergies are not static or homogeneous, as they exhibit temporal dynamics and spatial heterogeneity (Rodríguez et al., 2005). In addition, the reversibility of trade-offs varies: some ecosystem services could recover to their original state if the factor that disturbed the service is mitigated or eliminated (Rodríguez et al., 2005), whereas other services exhibit threshold behavior, and cannot recover without strong human assistance (Bai et al., 2010).

More and more scholars have begun to study the trade-offs and synergies between ecosystem services based on continuous improve-

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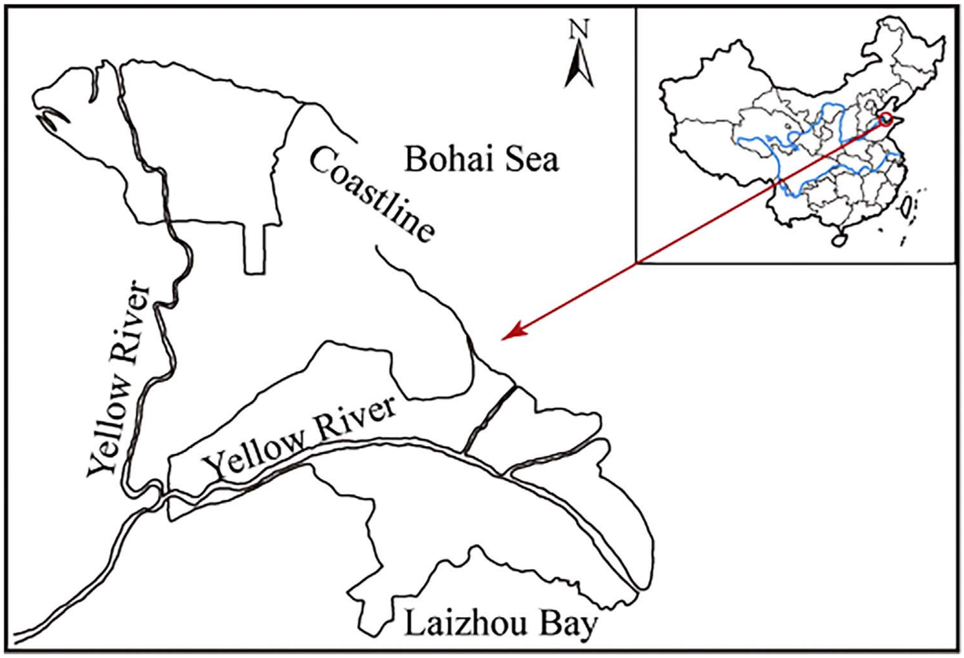

Fig. 1. Map of the coastal wetlands in the Yellow River Delta.

ment of valuation methods for these services (Schröter et al., 2014; Cavender-Bares et al., 2015a,b). Some trade-offs between ecosystem services are direct, inter-related, and self-generated (Nappi et al., 2010). In contrast, others result from indirect interactions caused by the actions of a co-driver (Jansson et al., 2015). Direct trade-offs are often revealed by the presence of statistically significant correlations (Bennett et al., 2009). There are two main methods used to reveal indirect trade-offs: service cluster analysis based on statistical clustering theory (Raudsepp-Hearne and Mooney, 2010) and impact analysis of iterations based on relational matrices (Altman et al., 2011). In addition, process-based ecosystem service models can be used to analyze trade-offs among services. These models can integrate changes in ecosystem processes with an assessment of ecosystem services and ecosystem management (Min et al., 2012). Because of excellent ability to evaluate process-based ecosystem services combined with service cluster analysis and impact analysis, the InVEST model was used in the present study. Studies of landscape-scale co-drivers of ecological processes (such as climate change and urbanization) have focused on the impact of land use or cover changes on the trade-offs or synergies between ecosystem services (Carreño et al., 2011).

Current methods for analyzing trade-offs and synergies include threshold analysis (Viglizzo and Frank, 2006), extreme-value analysis (Lv and Cheng, 2007), multi-objective analysis (Yan and Li, 2013), and model analysis (Haines-Young et al., 2012). Lautenbach et al. (2010) noted that the commonly used methods for examining trade-offs and synergies were based on graphical comparisons, whereas Butler et al. (2012) described scenario analysis and model simulation. There have also been studies that revealed trade-offs by assessing the economic and environmental benefits at a landscape scale in different years (Carreño et al., 2011). However, these studies only identified trade-offs between ecosystem services by comparing the magnitude of the shift in ecosystem services, and did not quantify the intensity of the individual trade-offs. Production possibility frontier curves (PPFs) offer a way to provide this quantification. This approach is based on an economic concept in which analysis can reveal the maximum outputs of combinations of two resources that simultaneously use the same set of inputs, thereby quantitatively describing the magnitude of the trade-off between the two outputs (King et al., 2015). Ager et al. (2016) used spatial optimization to analyze alternative restoration scenarios and examined the trade-offs based on PPFs for the relationships between selected restoration objectives.

However, studies of trade-off intensity have received little attention. Bradford and D'Amato (2012) used the root-mean-square error (RMSE) to quantify the trade-offs among multiple benefits. RMSE quantifies the average difference between each individual benefit and the mean benefit, and thus describes the magnitude of their difference from the mean, which in the Bradford and D'Amato study represented the distance from the "1:1 line" that represented equal benefits. Although RMSE is a simple but effective metric to quantify the trade-offs, it requires large amounts of measured value data, it is unsuitable for analyzing the data produced by evaluation models, and the "1:1 line" that represents equal benefits may not represent the optimal trade-off. Xue (2013) defined a trade-off strength coefficient (*TI*) that equals the ratio of the rate of change of a service to the rate of change of a target service. This approach can be used to evaluate the sensitivity of the change in a service value with respect to changes in the target service. Unfortunately, the same *TI* value may correspond to different combinations of ecosystem services, which may make it an ineffective indicator of trade-offs when the goal is to find the optimal combination.

Although early studies indicated that changes in ecosystem services could be attributed to the effects of land reclamation (Shi et al., 2013; Wang et al., 2014), there is still a need to quantify the relationship between reclamation and its effects on ecosystem services. In this paper, the InVEST model was used with remote sensing and socioeconomic data for the Yellow River Delta's coastal wetlands to evaluate the spatial patterns and dynamic evolution of coastal wetland ecosystem services since 1989 and to identify trade-offs and synergies among the services. By combining this analysis with PPF curves, a trade-off strength index was constructed and used to analyze the effects of reclamation activities on the trade-offs or synergies between ecosystem services. The results of our study will provide scientific support for the planning and implementation of reclamation activities and will contribute to the protection and management of wetland ecosystems in coastal areas.

2. Methods and models

2.1. Study area

The Yellow River Delta (37°35′N to 38°12′N, 118°33′E to 119°20′E) is located on the western coast of the Bohai Sea, in the northern part of

China's Shandong Province (Fig. 1). It has been formed by the sediments carried by the Yellow River, which changed its flow path in 1985, and the delta now covers an area of approximately 5500 km^2. It is China's largest and youngest estuary wetland, and is among the most active regions of land–ocean interaction in the world and among the largest river deltas. However, dam construction in the upper reaches of the Yellow River has greatly reduced inflows into the estuary, leading to increasing degradation of large areas of the delta's tidal wetlands as a result of long-term low inputs of freshwater and sediments (Cui et al., 2009; Sun et al., 2010). The Yellow River Delta Nature Reserve was established in 1992, and has played an important role in protecting newborn wetland ecosystems and rare or endangered birds, and it forms the northernmost border of the overwintering territory of *Grus japonensis* in China; it is also an important breeding place for *Chroicocephalus saundersi*.

With accelerating economic development and expansion of urbanization in the delta, the natural environment has become more fragile and threatened, particularly in the coastal wetlands. The ecosystems have been considerably changed by reclamation activities such as port construction, construction of tidal barriers, aquaculture, oil field development, and road construction. The delta's habitat has therefore suffered from severe disturbance, which has damaged biodiversity, and ecosystem structures and functions are facing serious threats (Ottinger et al., 2013). Reclamation may have permanently changed the natural properties of the coastal wetland ecosystems in the Yellow River Delta, and these changes inevitably affect the provision of ecosystem services by these wetlands.

According to the Millennium Ecosystem Assessment (MEA, 2005), ecosystem services can be divided into four broad categories: supply, regulation, support, and culture. We referred to the literature on ecosystem services (Arcidiacono et al., 2015; Harmáčková and Vačkář, 2015; Li et al., 2016) to select the major ecosystem services for Yellow River Delta Wetlands. The results from two Chinese studies (Han and Zhang, 2009; Xu et al., 2013) suggested, based on both economic valuation and emergy evaluation, that material production, carbon storage, and habitat quality were the dominant ecosystem services in the Yellow River Delta Wetlands.

2.2. InVEST model

In this study, version 3.1.1 of the InVEST model (http://www.naturalcapitalproject.org/invest/) was used to map and value two of the three major ecosystem services: habitat quality and carbon storage. (The third major service—material production—was analyzed using a market-value method, which is described in Section 2.3.) InVEST is a toolkit of 16 small models that are suited to terrestrial, freshwater, and marine ecosystems, and that simulate factors such as carbon storage, habitat quality, fisheries, coastal blue carbon (i.e., carbon captured by the ocean), coastal vulnerability, and marine water quality.

2.2.1. Carbon storage model

InVEST was used to estimate the amount of carbon stored over time in the delta's landscape. The model can also output value maps based on the market and social values of carbon sequestration using current carbon trading prices (USD/t) and the inflation rate, but the valuation tools were not used in the present study because our focus was on the relative size of the three ecosystem services and the impact of reclamation activities on the relationships among the ecosystem services.

In this model, the total carbon stock for a given parcel of land is divided into four basic carbon pools: aboveground biomass (C_{above}), belowground biomass (C_{below}), soil (C_{soil}), and dead organic matter (C_{dead}). Aboveground biomass comprises all living plant material above the soil (e.g., bark, trunks, branches, leaves). Belowground biomass encompasses the living root systems of aboveground biomass. Soil organic matter is the organic component of the soil. Dead organic matter includes litter as well as fallen and standing dead wood. So the model's calculation of total carbon (C_{total}) can be expressed as follows:

$$C_{\text{total}} = C_{\text{above}} + C_{\text{below}} + C_{\text{soil}} + C_{\text{dead}} \tag{1}$$

The model runs on a gridded map of cells (raster format), which were obtained from a shapefile of maps using the feature to raster tool in version 10.0 of the ArcGIS geographical information system (GIS) software (www.esri.com). The default grid size of 270 m × 270 m was used for the feature to raster tool. Each cell is assigned a land use and land cover (LULC) type, such as aquaculture area, cropland, or tidal flat. In addition, it was necessary to obtain carbon density (Mg ha^{-1}) tables for the four carbon pools in Eq. (1) for the diverse LULC types in the study area. The data sources for the shapefiles and carbon density tables are presented in Section 2.5.

2.2.2. Habitat quality model

We developed a bird species database for the study area and selected nationally protected bird species from the database. The species whose habitat was too specialized (such as *Ichthyaetus relictus*), species that were too common (such as *Cygnus cygnus*), and species that could not be studied due to limited data were excluded from the list. *Grus* spp. were selected as the indicator species for wetland waterfowl to evaluate the effects of human disturbance on habitat for coastal wetland birds. This genus was an appropriate choice due to its combination of rarity and high sensitivity to environmental changes (Song et al., 2009). Although *Grus* spp. cannot represent habitat suitability for all waterbirds, they are representative of the waterbirds that have been prioritized for management in the study area, and their main habitat (*Phragmites australis* marshes) represents the majority of the waterbird habitat that has been created in the nature reserve. At the same time, *Grus* spp. have a relatively high requirement for environmental quality, and are therefore more sensitive to environmental changes than many other waterbirds; they therefore represent a sensitive indicator of the quality of the dominant habitat in the reserve.

The habitat quality model assumes that habitat quality is affected by four factors: the relative impact of each threat created by different reclamation activities, the distance between the habitat and a threat source (i.e., the spatial impact of the threat), the level of protection (legal, institutional, social, and physical) from disturbance in each cell of the grid, and the relative sensitivity of each habitat type to each threat. This model also runs using the gridded LULC map, and requires a table of threat data, a table of habitat types and their sensitivity to each threat, the sources of the threats, accessibility to sources of degradation (a GIS polygon shapefile that contains data on the relative protection that legal, institutional, social, and physical barriers provide against threats), and a half-saturation constant k whose value is set by the user. The accessibility information was not available for the study area, and is not required by the model, so it was not included in the present analysis. Each threat source needs to be mapped within the grid, and the threat layers were extracted from the LULC layers using the reclassify tool in version 10.0 of ArcGIS.

In our analysis, the default values suggested in the software's user manual were used when no data were specifically available for China. However, Chinese values for many parameters were obtained from the research literature (see Section 2.5). The habitat value results are expressed as values from 0 to 1 in the model. The higher the value, the higher the habitat quality. The habitat quality in land parcel x that has LULC class j is given by Q_{xj}:

$$Q_{xj} = H_j\left(1 - \left(\frac{D_{xj}^z}{D_{xj}^z + k^z}\right)\right) \tag{2}$$

where H_j represents the habitat score assigned to LULC class j, whose value ranges from 0 to 1; D_{xj} is the total threat level in grid cell x with LULC or habitat type j;z is a scaling factor that is set to 2.5 by default in the model, and k is the half-saturation constant and is set to half of the size of landscape grid resolution (Liu, 2014). Eq. (2) was obtained from the InVEST user manual.

The sources of the data used in this analysis are presented in Section 2.5.

2.3. Market-value method

Material production services were assessed using the market-value method, as follows:

$$V = \sum Y_i \cdot A_i \cdot P_i \tag{3}$$

where V is the total market value of material production services in the study area, Y_i is the mean value of the i-th material's yield per unit area, A_i is the production area of the i-th material, and P_i is the mean value of the market price of the i-th material in a specified year. In the present study, the total values of the material production services provided in Dongying City, where our study area located, were divided into the grain and cotton value, the *Phragmites australis* value, the aquaculture product value, the salt-production value, and the fruit production value.

Yield increases due to technological innovations, changes in consumer preferences, changes in market prices caused by changes in supply and demand, and changes in inflation-induced values between the years were not accounted for. Excluding these factors prevented the calculation process from becoming too complicated and too sensitive to factors that are difficult to accurately account for. However, such factors could be accounted for in future research.

2.4. Quantitative indices

2.4.1. Reclamation intensity indices

The major reclamation types in the Yellow River Delta include aquaculture ponds, salt-production fields, agriculture, port construction, tidal barrier construction, and residential land use. A new reclamation intensity index was introduced to quantify the impact of reclamation activities on the coastal wetland ecosystem services, and six specific versions of this index were constructed that correspond to these six kinds of reclamation activities. Our index was developed based on the index proposed by Fu et al. (2010), whose reclamation intensity index (R_{Fu}) is expressed as the reclamation area (ha) per unit of shoreline length (km). The calculation formula is $R_{Fu} = S/L$, where S is the total area (ha) of a certain reclamation activity at the start of a given study period, and L is the total length (km) of the coastline in the study area. However, Fu et al. did not account for the effect of the distance from a threat source in calculating the reclamation intensity. As a result, we modified their index to account for that effect.

Considering that reclamation activities in our study area have changed ecosystem services mainly through changing the LULC types in the coastal zone, the original area-based calculation formula is reasonable. However, the threats to habitat quality created by different reclamation activities are different and their corresponding threatened areas are also different. Therefore, a threat factor (μ) needs to be considered when constructing a suitable index. The areas (πD^2) threatened by different reclamation activities (except for tidal barriers, which will be described separately) can be defined by the maximum threat distance (D, which represents the radius of a circle centered on the threat). Therefore, our revised version of reclamation intensity index for the i-th reclamation activity (R_i) is calculated as follows:

$$R_i = \mu_i \cdot \frac{D_i^2}{D_0^2} \cdot \frac{S_i}{L} \tag{4}$$

where μ_i is the threat factor score for the i-th reclamation activity; D_i is the maximum threat distance for the i-th reclamation activity; D_0 equals the threat distance for cropland (which is the minimum threat distance); S_i is the total area of the i-th reclamation activity within the study area (ha); and L is the total length (km) of the shoreline in the study area.

The values of the threat factor and maximum threat distance for each reclamation activity were calculated from the questionnaire results in Supplemental Appendix S1: for aquaculture ponds, $\mu_a = 0.44$ and $D_a = 1.77$ km; for cropland, $\mu_c = 0.47$ and $D_c = 1.12\ \text{km} = D_0$; for port construction, $\mu_p = 0.92$ and $D_p = 5.52$ km; for residential areas, $\mu_r = 0.82$ and $D_r = 3.22$ km; and for the salt-production fields, $\mu_s = 0.49$ and $D_s = 1.32$ km. These data were obtained from questionnaire results. See Section 2.5 for details.

The tidal barriers have distinctive characteristics that should be noted, because they require a different approach from that in Eq. (4). The barriers block the normal exchanges of materials and energy between beaches and the ocean, which causes shrinkage of beaches and affects normal plant succession processes. Therefore, it is necessary to consider both changes in the area occupied by the barriers and in the threat factor, which depends on the barrier's design criteria (i.e., the height of the dike, the strength of the building material, the thickness of the parapet wall; Jin et al., 2016). To facilitate the calculation, a comprehensive storm surge strength index (σ_{ss}) was used to represent the contribution of this strength to the reclamation intensity R_t. The values of σ_{ss} from 1989 to 2015 were calculated based on data from a previous study (Jin et al., 2016); these values were 1.00 in 1989, 1.40 in 1998, 1.56 in 2008, and 1.77 in 2015. Therefore, the formula for the reclamation intensity of tidal barriers is:

$$R_t = \sigma_{ss} \cdot \mu_t \cdot \frac{2L_t \cdot D_t + \pi \cdot D_t^2}{\pi \cdot D_0^2} \cdot \frac{S_t}{L} \tag{5}$$

where the threat factor is $\mu_t = 0.89$, L_t is the length of the tidal barrier, the maximum threat distance is $D_t = 9.70$ km, and L is the total length (km) of the shoreline in the study area. In this analysis, μ_t and D_t were obtained from the questionnaire.

2.4.2. Intensity indices for trade-offs

PPF curves are an economic concept that refers to the possible boundary formed by combinations of the maximum production of two services under a given combination of resource conditions and technical constraints, and can be used for a variety of production portfolio choices. PPFs have also been called Pareto frontiers or efficiency frontiers. Some studies have utilized PPFs to examine the trade-offs between combinations of objectives or ecosystems (Schröter et al., 2014; Cavender-Bares et al., 2015a,b; King et al., 2015), but the intensity of the trade-offs must still be explored.

In this study, PPFs were used to determine the trade-offs between the material production and habitat quality services and between the material production and carbon storage services. Taking the trade-offs between the material production and habitat quality services as an example, the raster layers in the GIS were first transformed into point layers, and the point layers for the two services being compared were overlaid in the GIS software to obtain a single composite layer, in which each point or geographic location had attributes that corresponded to the values of the two services. Values of each service were normalized by dividing the value of the service in a given year by the maximum value of that service during the study period, which therefore received a normalized value of 1. The attribute table for that point was then extracted, the normalized values of the services were summed, and the total values were then arranged in ascending order. Finally, the results were used to identify all combinations of ecosystem services that provided the maximum total value, and that data was used to create a PPF curve that represented the optimal trade-off for any combination of service values.

Assuming that the direction of the trade-off is chosen to optimize the combination of the two services, the optimal state can be regarded as the equilibrium state, and only one optimal state can be achieved under a given set of conditions; that is, only one PPF curve can be achieved. Based on this assumption, the shortest distance between the

PPF curve and a point corresponding to the simultaneous mean values for the two services represents the value of the trade-off intensity index. If P represents the point corresponding to the simultaneous mean values for the two services, its location is (x_0, y_0), and the PPF curve is expressed as $y = f(x)$, then the location of any point Q on the curve is $(x, f(x))$. The distance between P and Q is calculated as follows:

$$D_{PQ} = |P(x_0, y_0) - Q(x, f(x))| = \sqrt{(x_0 - x)^2 + (y_0 - y)^2} \tag{6}$$

Therefore, the formula for the trade-off intensity index is:

$$D_{PQmin} = \min\{|P(x_0, y_0) - Q(x, f(x))|\} = \min\{\sqrt{(x_0 - x)^2 + (y_0 - y)^2}\} \tag{7}$$

The greater the value of D_{PQmin}, the stronger the trade-off that results from the balance; the smaller the D_{PQmin}, the weaker the trade-off. Similarly, for the three ecosystem services considered simultaneously, the trade-off intensity index can be calculated as the shortest distance between the mean values of the three services and the surface in a three-dimensional graph of the optimal state; because this curve exists in three dimensions, it is referred to as a PPF surface.

One limitation of the present approach is that it allows one or more service values to decrease to 0 in the optimal result. In future research, it will be necessary to modify the method to constrain service values by defining a minimum value for services that are considered to be essential for an area (e.g., provision of wildlife habitat) and a maximum value for services that would produce unsustainable levels of damage if they covered too much of the study area (e.g., reclamation for agriculture).

2.5. Data processing and analysis

To assess these ecosystem services using the direct market-value method and the InVEST model, it was necessary to obtain LULC maps, threat factor distribution layers, model input table data (threat data, sensitivity data), and relevant socioeconomic data.

2.5.1. LULC maps

LULC maps from 1989, 1998, 2008, and 2015 were selected as our base map data, and this dataset was obtained from interpretation of SPOT5 satellite images from August 1989, April 1998, May 2008, and May 2015, and the data were interpreted using a combination of object-oriented methods and visual interpretation. The image resolution was 20 m. The final LULC maps were validated using field survey data and data from Google Earth, and the overall accuracy was 90%.

The landscapes in the LULC maps were divided into 17 categories with reference to the Current Land Use Classification of China (GB/T21010-2007; Ministry of Land and Resources, 2007) and the needs of this study. Because the InVEST model needs to run on gridded maps in GIS raster format, the LULC vector map was processed using the feature to raster conversion tools provided by ArcGIS 10.0. As noted earlier, the grid cell size was 270 m × 270 m.

2.5.2. Threat factor distribution layers

The threat factor distribution layers were extracted from the LULC raster maps using the reclassify tool provided by ArcGIS10.0

2.5.3. Model input table data

The input table data for the habitat quality module (the threat factor scores and sensitivity analysis scores) were derived from a questionnaire (see Supplemental Appendix S1). Of the nine *Grus* species recorded in China, seven have been recorded in the Yellow River Delta: *Grus canadensis*, *Grus virgo*, *Grus japonensis*, *Grus grus*, *Grus leucogeranus*, *Grus vipio*, and *Grus monacha*. In the study area, waterbird surveys usually focus on the period from November to March, which is the main migration and overwintering period for these birds. China Coastal Waterbird Census Report (September 2005 to December 2007) (2009) shows that, in the Yellow River Delta, the total number of *Grus* spp. individuals observed in 2007 was 500, which is the smallest number recorded during the census period since the census began in 1998. Liu (2015) reports that by 2014, the total population had increased greatly, to 5442 individuals.

The favorite types of LULC for these birds are *Phragmites australis* marshes and open water (Song et al., 2009). Based on the LULC classification, reservoir ponds, rivers or canals, other bodies of shallow water, tidal flats, and areas of *Phragmites australis*, *Suaeda salsa*, *Spartina alterniflora*, and *Tamarix chinensis* represented potential habitat, and other vegetation or LULC types were unsuitable as habitat.

Threat factors in the study area resulted from both human disturbances and natural disasters. Natural disasters include floods, droughts, and storm surges, which have happened only rarely in recent years, data on their distribution was not available. Therefore, only human-induced threat factors were considered. Human disturbances included reclamation activities and other human activities. The typical reclamation activities included construction of tidal barriers, port construction, aquaculture, and salt-production fields. Other typical human activities were mainly farming, oil extraction, construction of residential areas, and ecological restoration projects. During the winter, birds are less affected by human disturbances, and sometimes feed in cropland.

The input table data for the carbon module of InVEST were obtained by means of a literature review to determine the aboveground biomass and belowground biomass for the study area as well as the associated carbon density values (Dong et al., 2010; Zhang et al., 2012; Xu et al., 2014; Song, 2015), the soil carbon density data (Cao et al., 2013; Yu et al., 2013; Song, 2015), and the dead organic matter carbon density data (Song, 2015). The results were then summarized to provide a final wetland carbon density table for each LULC type in the study area. It was assumed that the carbon density of a given type of LULC did not change during the study period.

2.5.4. Socioeconomic data

After processing the socioeconomic data (see Supplemental Appendix S2), the yield per unit area, production area, and mean market price for each product were obtained. Where regional data were unavailable, they were replaced with the national average or with averages for other regions that had comparable conditions. To facilitate the calculation and spatial mapping, the grain and cotton planting area were assumed to account for half of the cropland area. The aquaculture products represented the average of the main products of freshwater aquaculture and marine aquaculture (such as *Penaeus chinensis*); products in reservoirs or ponds were based on the values for *Eriocheir sinensis*. For fruit tree orchards, the yield and its value were set to the average for all kinds of fruit orchards in the study area (e.g., jujube, mulberry, other fruits). Note that because the orchards are small and scattered in the study area, the maps did not show the distribution of all fruit tree orchards. In addition, all forest LULC types were assumed to be fruit tree orchards because little or no afforestation or wood harvesting was recorded in government statistics for the study area.

All the parameter names used in the analysis are defined in Appendix S3 in the Supplemental material.

3. Results

3.1. Spatial distributions and temporal changes of ecosystem services

3.1.1. Material production

The distributions of the material production values from 1989 to 2015 were mapped (Fig. 2). For cells in which there was no supply of material production services in the LULC types, the value of the cell was set to 0. The maximum values increased gradually throughout the study period, with values increasing from 0.40×10^4 Yuan/ha in 1989–2.13×10^4 Yuan/ha in 1998, 2.70×10^4 Yuan/ha in 2008, and 6.64×10^4 Yuan/ha in 2015.

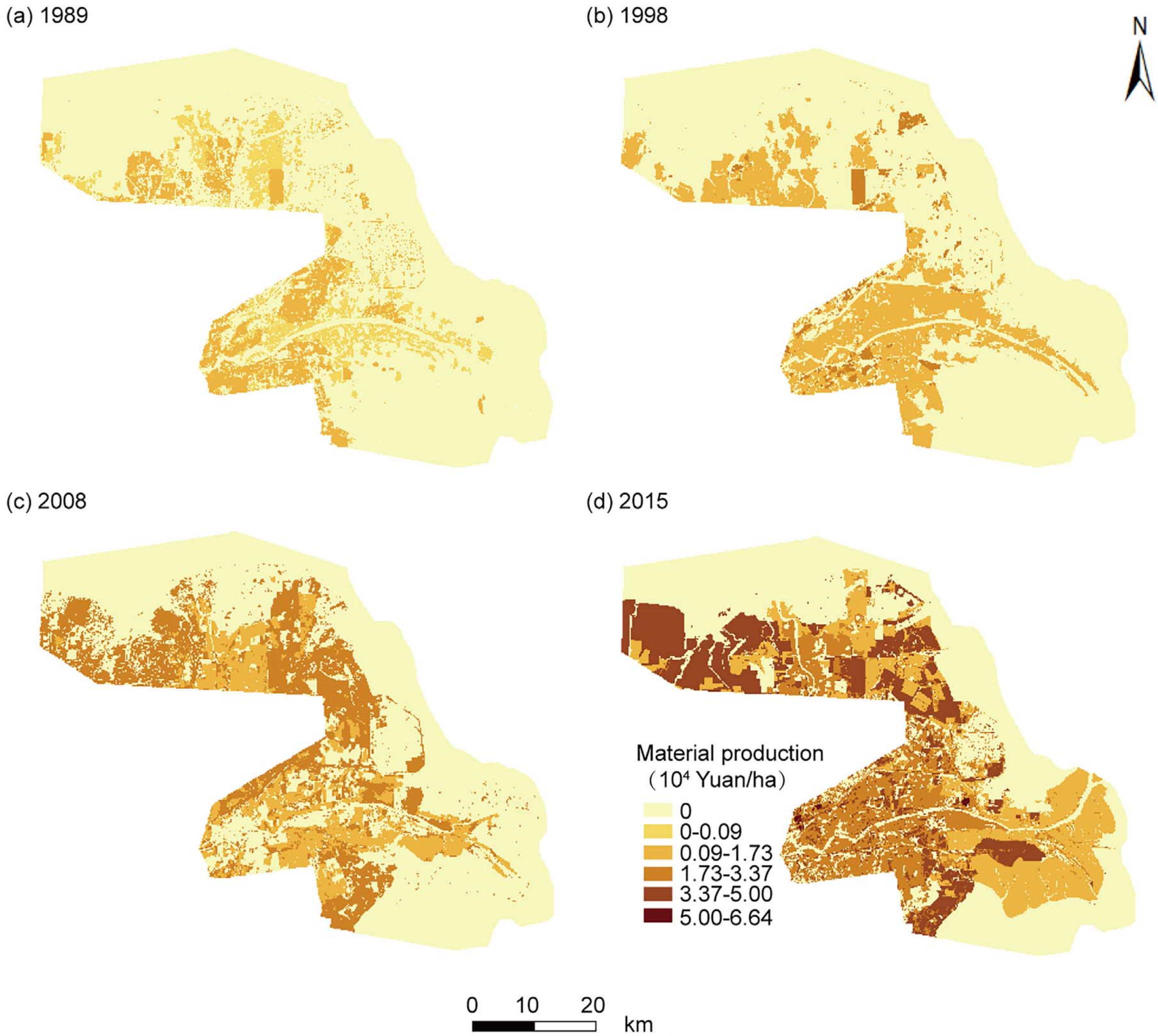

Fig. 2. Distribution of the material production ecosystem service values from 1989 to 2015.

Most of the cells with higher values of material production were located far from the shoreline of the Bohai Sea and along the main stream of the Yellow River, which were the target areas for the regional freshwater restoration engineering project (Cui et al., 2009), but were also found in the central and northern parts of the Yellow River Delta, which represents the main location of fruit tree orchards. The areas nearest to the coastline had low material production values throughout the study period.

3.1.2. Carbon storage

The changes in the distribution of the carbon density values from 1989 to 2015 were also mapped (Fig. 3). From the results of the model evaluation, the maximum carbon density in each grid cell was 769.02 Mg ha^{-1} in the fruit tree orchards, and the minimum value was 59.72 Mg ha^{-1} in the port LULC type.

Table 1 shows that the average carbon density remained relatively stable in the study area, but that the total carbon storage has increased greatly since 2008 (by more than 1.03×10^5 Mg).

3.1.3. Habitat quality

The prepared data were used as inputs for the InVEST model to obtain the habitat quality distribution for *Grus* spp. in the four years of the study (Fig. 4). The habitat quality evaluation showed that the minimum habitat quality score in the four years was 0, which appeared in the non-habitat LULC types, and the maximum score was 1, which appeared primarily in the in areas of vegetation and bodies of shallow water. Except for the non-habitat LULC types, scores mostly ranged between 0.91 and 1.00, which suggests little difference in habitat quality among the suitable habitats. The average habitat quality scores in the study area, including all non-habitat cells in the grid, were 0.77 in 1989, 0.79 in 1998, 0.64 in 2008, and 0.55 in 2015. Thus, the average value of the habitat quality scores initially remained stable, and then decreased.

In future research, this simplistic analysis could be improved by evaluating habitat quality more precisely (e.g., distinguishing more clearly between the values of different habitats) and by accounting for differences in the habitat quality among waterbird species once data on these differences becomes available.

3.2. Effects of reclamation activities on multiple ecosystem services

3.2.1. Patterns of ecosystem service bundles in the landscape

We first extracted the values of the carbon storage, habitat quality, and material ecosystem services for the 17 LULC types, then calculated the mean values of each ecosystem service in each LULC type (Fig. 5). The mean habitat quality scores in each LULC type generally had a value of 0.99–1.00, except for non-habitat cells of the grid, which had a value of 0. The mean values of each ecosystem service were arranged in ascending order, then five grades were classified averagely.

In Fig. 5, the landscapes were divided into two groups within the graphs: types above and including the port LULC, and types below the port LULC. The former landscapes are human-dominated landscapes,

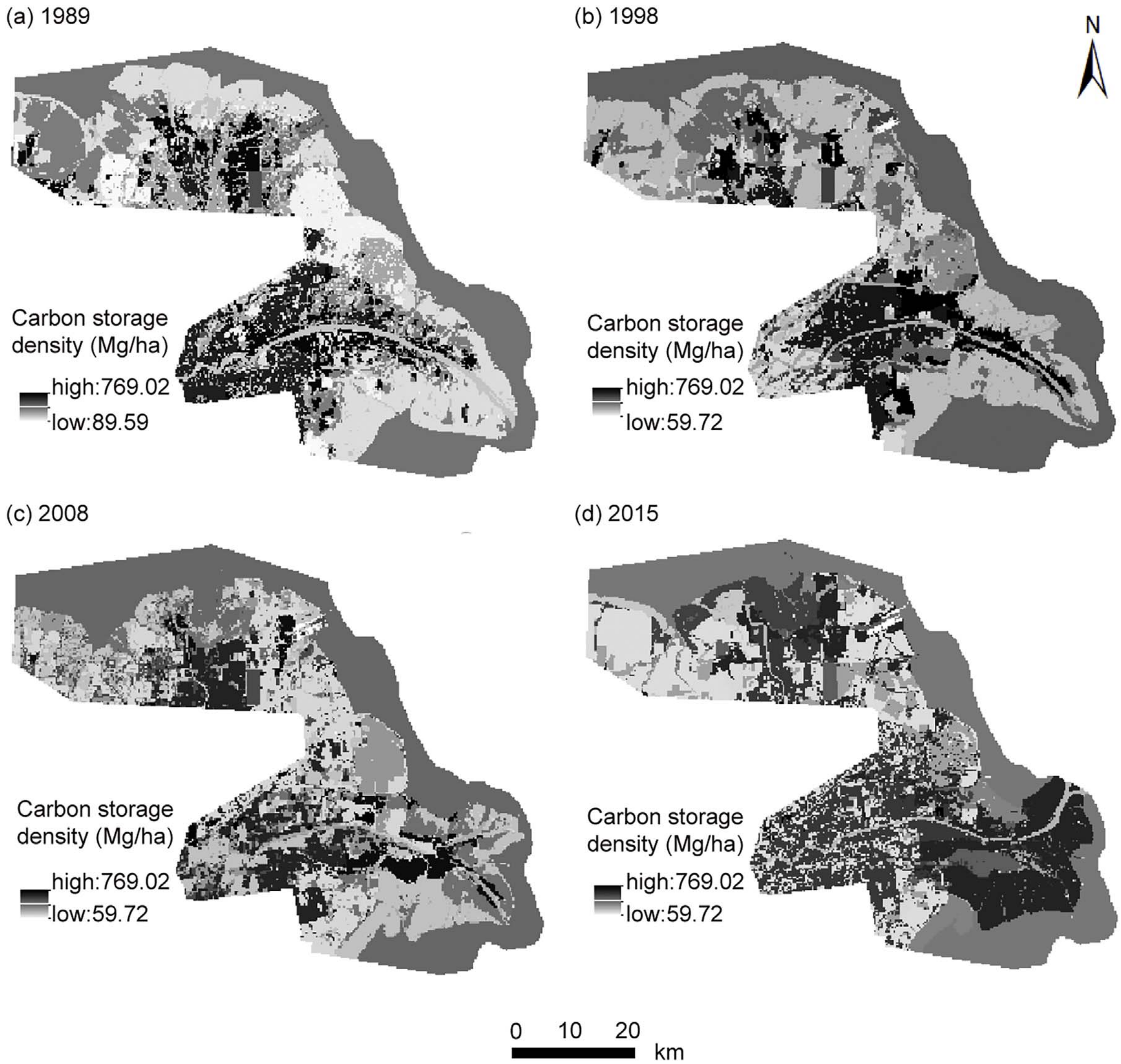

Fig. 3. Changes in the carbon density distribution from 1989 to 2015.

Table 1
Average density and total storage of carbon (C) in the study area.

	1989	1998	2008	2015
Average C density (Mg ha^{-1})	181.94	180.36	181.85	209.63
Total C storage ($\times 10^6$ Mg)	6.800	6.752	6.807	7.839

whereas the latter are natural or semi-natural landscapes. In some landscapes, all three ecosystem services had scores of 0 because those landscapes didn't exist in that year. The results show obvious differences in the distribution of the three ecosystem services. During different years, natural landscapes all had a habitat quality grade of 5, whereas the human-dominated landscapes generally had habitat quality scores of 0, except for ponds. For the oil field, aquaculture pond, salt-production field, bare land, tidal barrier, port, and residential area landscapes, all three services commonly had grades of 0.

3.2.2. Intensities of various reclamation activities

The reclamation intensities of the different reclamation activities from 1989 to 2015 were calculated using Eqs. (4) and (5), and we calculated the total intensity (R_{total}) as the sum of the reclamation intensities of all other reclamation activities (Table 2).

In 1989, the intensities for the port areas, residential areas, and salt-production fields were all 0, so the corresponding reclamation intensity was also 0. However, all three intensities increased rapidly throughout the study period. The intensities of aquaculture, port construction, and tidal barrier construction increased throughout the study period, whereas the intensities of cropland decreased from 1989 to 2008 and then increased greatly, and the intensities for residential areas increased initially, remained stable until 2008, and then increased greatly in 2015. The intensities for the salt-production fields remained small throughout the study period, but showed a large overall increase in 2015. The total reclamation intensity increased steadily, but with a particularly rapid increase (more than doubling) from 2008 to 2015, which can be attributed to the rapid increase of urbanization and tidal barrier construction between 2008 and 2015.

3.2.3. Relationships between reclamation intensity and ecosystem services

To identify the contribution of different reclamation activities to changes in the three ecosystem services, multiple-regression analyses were performed for the relationships between the reclamation intensity indices (the R values in Table 2) and the total values of each ecosystem service (carbon storage, material production, and habitat quality) during the 4 years in the study area. Taking R_a, R_c, R_p, R_r, R_s, and R_t as independent variables, and the three ecosystem services as dependent variables, we developed the following regression equations:

Fig. 4. Changes in the distribution of habitat quality for *Grus* spp. during the study period.

$$M = -14985.409 + 218.591\ R_a + 202.092\ R_s + 4.022\ R_t \quad (8)$$

$$H = 33117.312–51.814\ R_a + 66.827\ R_s - 1.700\ R_t \quad (9)$$

$$C = 5881032.233–1332.127\ R_a - 8602.430\ R_s + 2001.325\ R_t \quad (10)$$

where M represents material production, H represents habitat quality, and C represents carbon storage; R_a represents aquaculture ponds; R_s represents salt-production fields; and R_t represents tidal barriers. Note that R_r (residential areas), R_c (cropland), and R_p (ports) were not included as statistically significant independent variables in this analysis. For each regression, the goodness of fit was excellent ($R^2 \geq 0.9$, $p < 0.01$). Thus, aquaculture generally increased material production, but decreased habitat quality and carbon storage. In contrast, salt-production increased material production and habitat quality, but decreased carbon storage. Finally, tidal barrier construction decreased habitat quality, but increased material production and carbon storage.

We also calculated the correlation (Spearman's r, because the values were not normally distributed) between R_{total} (the sum of the values for all activities combined) and each ecosystem service, and found a strong and significant positive correlation between R_{total} and material production ($r = 0.9$, $p < 0.01$; two-tailed test). In contrast, R_{total} was negatively correlated with habitat quality ($r = -0.800$), and positively correlated with carbon storage ($r = 0.800$), but the correlations were not significant ($p > 0.1$).

3.3. Trade-offs among ecosystem services

3.3.1. Correlations among multiple ecosystem services

The Kolmogorov-Smirnov test showed that the distributions of the three ecosystem services were not normal, so Spearman's correlation was used to detect significant relationships among them (Fig. 6). The results showed significant correlations ($p < 0.01$; two-tailed) among the ecosystem services during the four years of the study period: $r > 0.36$ for material production versus carbon storage in 1989 and 1998, $|r| > 0.48$ (a negative correlation) for material production versus habitat from 1989 to 2015, and $r > 0.1$ for habitat versus carbon storage.

Fig. 6 shows significant negative correlations between the material production services and the habitat quality service in all years, which means that there was a trade-off between these services. The magnitude of the correlation coefficient increased from 1989 to 2008, indicating an increasingly negative correlation, then decreased from 2008 to 2015, indicating a less negative correlation. In contrast, the correlation between carbon storage and habitat quality remained significant and positive throughout the study, suggesting that there was a synergistic relationship between them. The strength of this correlation (thus, of the synergy) decreased from 1989 to 1998, and then increased from 1998 to 2015. The carbon storage and material production services were significantly positively correlated in 1989 and 1998, and then became significantly negatively correlated in 2008 and 2015; that is, the relationship between the two services transformed from a synergy into a trade-off relationship, although the strength of the correlation

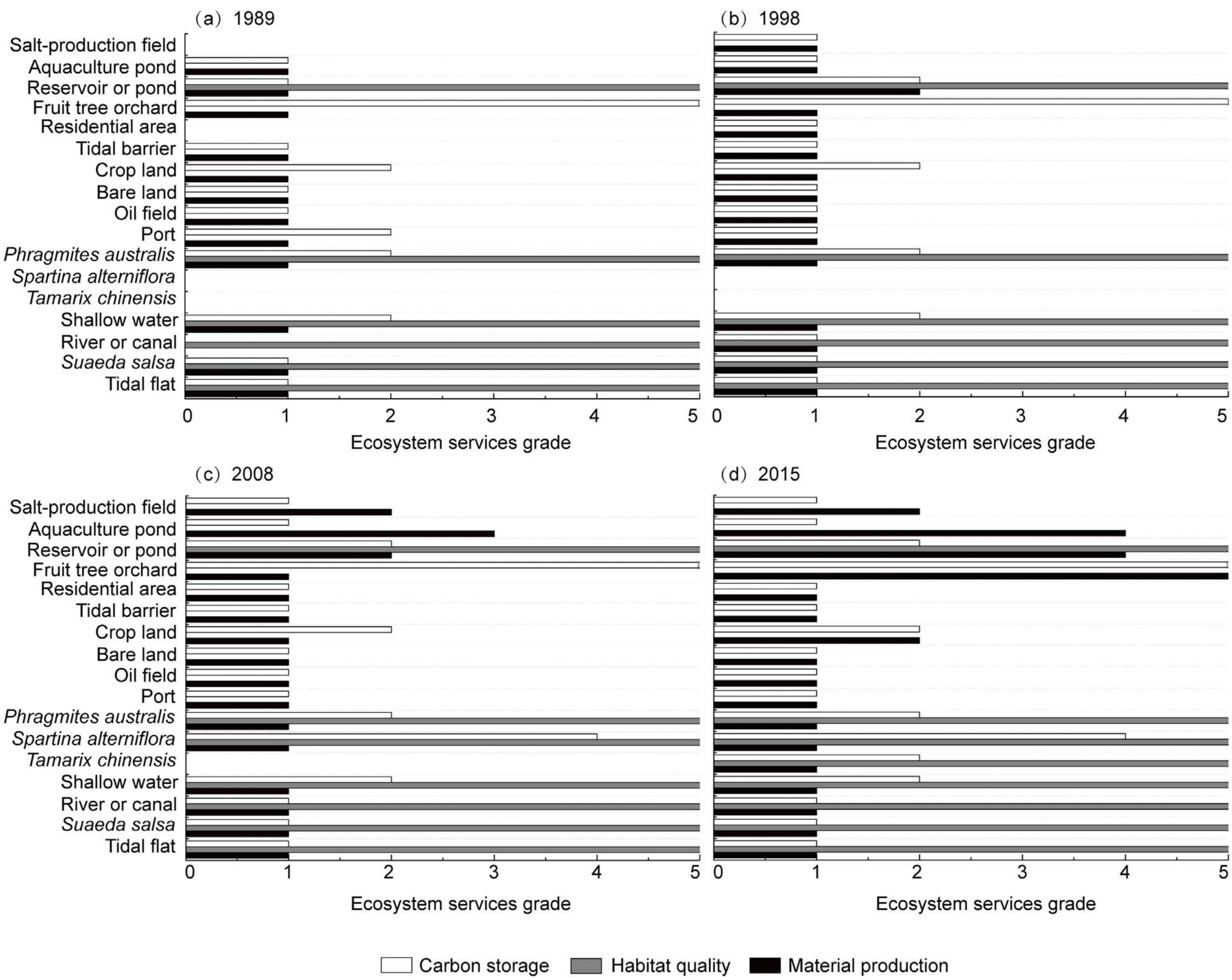

Fig. 5. The changes in ecosystem service values for each of the three services (material production, habitat, and carbon storage) throughout the study period. Ecosystem service grades ranged from 0 (a land use completely unsuitable for providing the service or a land use that did not exist in that year) to 5 (the most suitable land use for providing the service).

Table 2
Changes in the intensity of each reclamation activity (R_a, aquaculture ponds; R_c, cropland; R_p, ports; R_r, residential areas; R_s, salt-production fields; R_t, tidal barriers) and in the total intensity of these reclamation activities (R_{total}) throughout the study period.

Year	Reclamation activity intensity						
	R_a	R_c	R_p	R_r	R_s	R_t	R_{total}
1989	66.78	64.05	0	0	0	503.54	634.37
1998	71.40	46.84	29.50	12.56	15.90	551.13	727.34
2008	170.79	44.80	44.67	12.10	11.03	623.79	907.18
2015	240.69	121.89	74.96	121.49	28.96	1263.2	1851.20

decreased. The driving factors that caused these changes in the relationships among multiple ecosystem services will be explored in our future research.

Fig. 7 illustrates the three-dimensional PPF surface formed by the values of the three ecosystem services in 2015. At the optimal combination point, the carbon storage, material production, and habitat quality values were 769.0 Mg ha^{-1}, 6.64×10^4 Yuan/ha, and 0, respectively. However, because a habitat value of 0 is unlikely to be acceptable to managers, it is necessary to set a minimum habitat value and look for a new optimum that provides this value. For example, for a minimum acceptable habitat value of 0.99, the combined value would then become the total economic value, with values of 317.68 Mg ha^{-1} for carbon storage and 0.64×10^4 Yuan/ha for material production.

3.3.2. Changes in the intensity of trade-offs among multiple ecosystem services

The previous results demonstrated that tradeoffs existed between the material production and habitat quality services from 1989 to 2015, and between the material production and carbon storage services from 2008 to 2015. To search for the optimal combinations, all the combinations of the pairs of ecosystem services were compared and PPF curves (tradeoff efficiency curves) were then fitted to this dataset (Fig. 8).

The normalized ecosystem service values formed convex-downward potential trade-off curves. Fig. 8a shows that the PPF curves between material production and habitat quality could be divided into three parts. The upper and lower parts formed a straight line, because the initial normalized maximum value of any service is 1 and changes slowly when the tradeoffs are relatively minor; however, the middle part of each curve is convex-downward, indicating the existence of trade-offs between the services. If the habitat quality service is regarded as an opportunity cost for material production, and decreases with increasing material production, then there were huge differences among the opportunity costs as the material production service increased during the different stages of the curve. Initially, the opportunity cost was small, and it increased gradually, but at some point, habitat quality began to decline rapidly; as a result, management strategies must be implemented to minimize the risk to habitat quality, such as planting perennial cash crops that can also serve as wildlife habitats. From 1989–2015, the total ecosystem services continued to

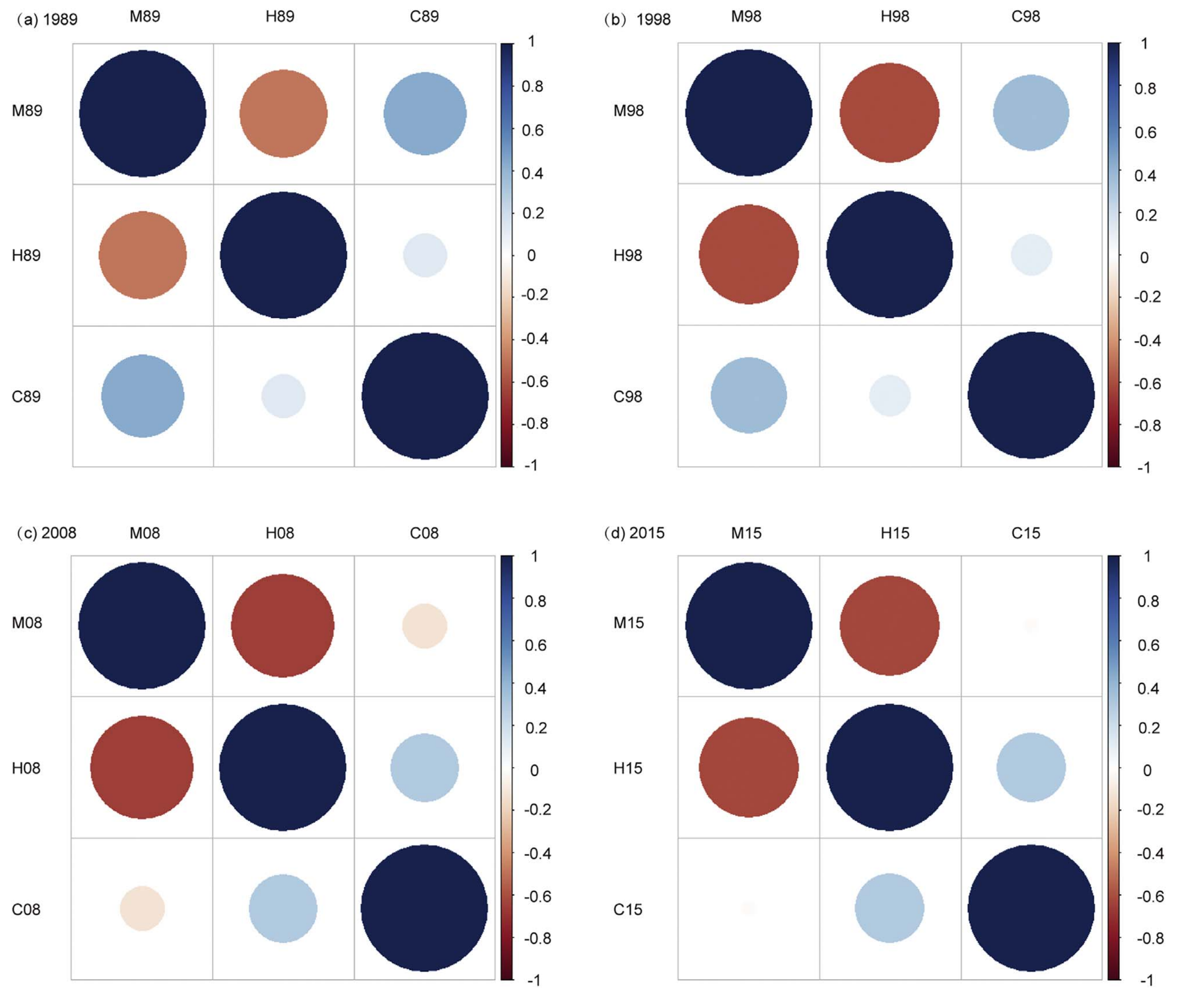

Fig. 6. The correlations (Spearman's *r*) among the three types of ecosystem services (M, material production; H, habitat quality; C, carbon storage) from 1989 (e.g., M89) to 2015 (e.g., C15).

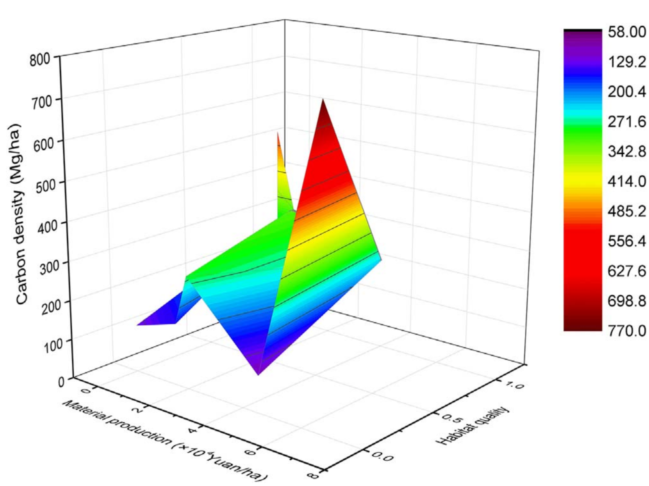

Fig. 7. The three-dimensional PPF surface for the three ecosystem services in 2015.

increase (from P1 to P4), so the PPF curve expanded outward. Fig. 8b shows a similar pattern, but with different details, for the PPF curves between material production and carbon storage services; the curves were convex downward in 2008 and 2015, and as the total ecosystem services increased, the PPF curve also expanded outward.

The PPF curve for 1989 shows that the point nearest to P1 is on the upper part of the convex curve, whereas in the PPF curves for 1998, 2008, and 2015, the points nearest to P2, P3, and P4 (respectively) are all on the upper part of the straight lines. Based on this data, the trade-off intensity indices and their changes during the study period were calculated (Table 3). As noted in the Methods section, this index is based on the coordinate system defined by the normalized values of the services, so its value is relative. Table 3 shows that the trade-off intensities increased rapidly for both trade-offs.

4. Discussion

To the best of our knowledge, the present study represents the first use of PPFs to quantify the trade-offs among ecosystem services. Using PPF curves, it's easy to visualize the nature of the trade-offs and identify their intensity at various points. In previous studies (King et al., 2015; Ager et al., 2016), PPF curves were used to weigh the trade-offs or synergies between ecosystem services, but the comparison was only qualitative; in essence, the analysis only compared the benefits under different scenarios or objectives to weigh the pros and cons of various combinations of services, seeking the optimal solution. The trade-off intensity was not determined.

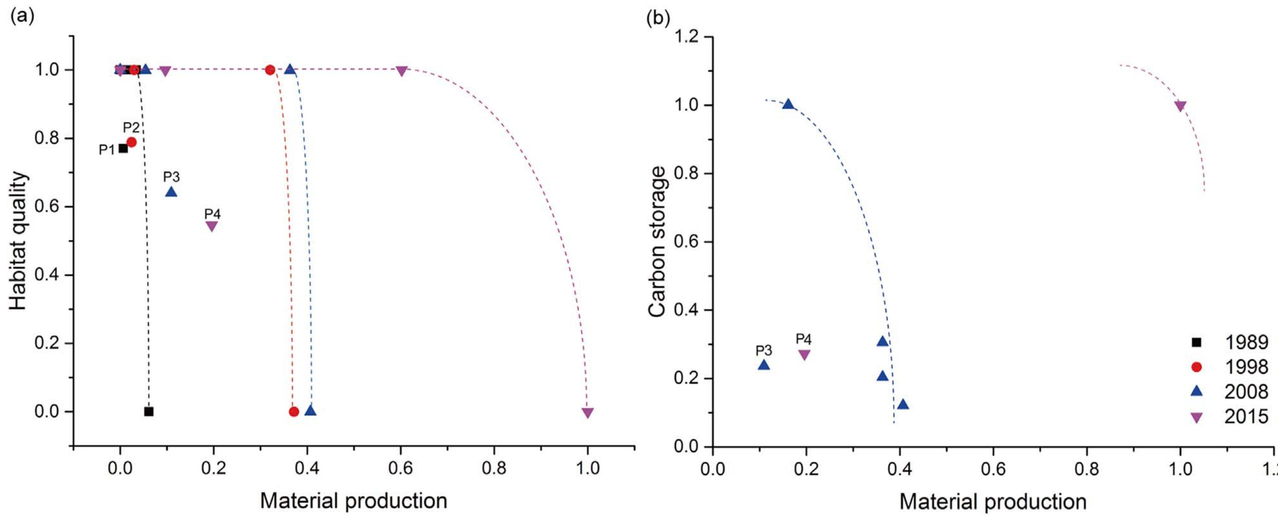

Fig. 8. PPF curves for the relationships between (a) material production and habitat quality services, and (b) carbon storage and material production services from 1989 to 2015. P1 to P4 represent the mean values of the two ecosystem services in 1989, 1998, 2008, and 2015, respectively.

Table 3
Changes in the trade-off intensity indices among the three ecosystem services during the study period.

Trade-off	Trade-off intensity index			
	1989	1998	2008	2015
Material production versus habitat quality	7.1	25.7	43.3	54.5
Material production versus carbon storage	–	–	44.5	150.2

In economics, points inside the PPF curve (i.e., between the curve and the origin of the graph) represent economically inefficient situations, such as incomplete use of a resource; in contrast, points outside the curve represent states that cannot be achieved under current resource or technical constraints. In our study, the mean value points were all inside the PPF curves, suggesting that the current ecosystem service allocation is inefficient and could be improved. There are many roads to achieve the optimal combination, but only one will have the lowest opportunity cost, which corresponds to the shortest distance from a point to the PPF curve. In economics, expansion of the resource supply, improvement of product quality, and technological progress in a dynamic economy can move the PPF curves outwards, giving the economy access to a greater number of both products. Similarly, for ecosystem services, natural or human-dominated activities such as agriculture, forestry, and aquaculture can expand their values and provide more of these services, although this may often exacerbate the conflicts among them, leading to trade-offs rather than synergies. It is also possible for the PPF curves to move outwards when synergies exist that increase the total value of the services compared to their total value in the absence of synergies. Thus, rather than being content with finding only a single optimal combination of services, managers should look for alternatives that create synergies and therefore increase the overall value (i.e., move the curve outwards). As for a 3D PPF surface such as the one in Fig. 7, it may also reveal the potential for such synergies, but the analyses may be complicated, and the problem of this complexity must be solved in future research.

The driving factors behind LULC changes in the Yellow River Delta are both natural and socioeconomic. In this study, only the socioeconomic factors were considered. As an economy develops and becomes more urban, socioeconomic factors such as land reclamation activities and other human activities to support a growing population play an increasingly significant role in LULC changes. Of the total reclamation intensity from 1989 to 2015 in our study, the construction of tidal barriers contributed the most to the intensity, owing to its notable role in creating a barrier between the terrestrial and marine ecosystems. The rapid development of large-scale aquaculture in coastal areas led to this activity having the second-highest intensity, but also led to the degradation of ecosystem services.

When assessing the market value of ecosystem services, ecosystems are regarded as the supplier of these services, and the services change in response to policies and development plans that change the balance and intensity of reclamation activities. The relationships among ecosystem services also differ among the services. The managers responsible for managing these relationships must seek acceptable trade-offs between the supply of ecosystem services and the trade-off intensity. On the other hand, when humans enjoy the benefits brought by ecosystem services, they appear as consumers of these services, and there is a trade-off between the consumption of different services that requires them to weigh the strength of the trade-off. Thus, the market value of ecosystem services is determined by the balance between supply and demand.

There are still some unanswered questions in this study. First, although we have analyzed the relationship between reclamation activities and changes in the trade-off intensity, it is difficult to identify the specific drivers of these changes, such as which specific reclamation activities contributed most to movement of the trade-off curves. Because the relationships between the intensities of different reclamation activities and the trade-off intensity differ among the combinations of activities in complicated ways, and because the present analysis was limited by the use of data from only 4 years, it was hard to precisely simulate changes in the relationships between selected reclamation activities and the trade-off intensity. Moreover, reclamation activities led to changes in ecosystem services, which ultimately led to changes in the trade-off intensities between ecosystem services. Complex functional mechanisms also affected the relationships between activities, and simple correlation analysis of the evaluation results cannot adequately describe their relationship.

A second problem is the need to facilitate decision-making related to wetland management. To support this goal, it will be necessary to develop ecological management policies based on an evaluation of ecosystem services scenarios, and then, based on those scenarios, future ecosystem services trade-offs and trade-off intensities could be quantitatively analyzed. Such scenarios could define minimum and maximum values for a given service, thereby eliminating solutions that produce unacceptably low levels of a service such as habitat or unsustainably high levels of a service such as agricultural production.

Finally, the present study only focused on the trade-offs or synergies between the dominant ecosystem services. This represents an incomplete accounting because it omitted many potentially important services, thereby making the analysis less accurate than it could eventually become, and preventing us from finding solutions in which trade-offs between some pairs of services could be mitigated by synergies between other pairs.

5. Conclusions

Our study showed how PPF curves and surfaces provide an effective framework for analyzing the trade-offs among ecosystem services. This paper visualized the spatial patterns of three key ecosystem services (habitat quality, carbon storage, and material production) and the dynamic evolution of their values since 1989 in the coastal wetlands of the Yellow River Delta. Based on the PPF curves, we quantified the intensities of the trade-offs between pairs of ecosystem services and the relationships between the intensity of the reclamation activities and the resulting changes in the ecosystem services. Our analysis revealed different responses of the ecosystem services to different LULC changes, but overall, the reclamation activity intensity increased greatly, leading to increasing trade-offs among services. The approach developed in the present study shows promise for helping managers to find the optimal allocation of ecosystem services, and will therefore provide a scientific basis for improving wetland management and ecological restoration. It therefore has theoretical and practical significance for promoting sustainable development of coastal areas.

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Appendix A. Supplementary data

Supplementary data associated with this article can be found, in the online version, at http://dx.doi.org/10.1016/j.ecolind.2017.05.005.

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