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A method for identifying suitable biodiversity offset sites and its application to reclamation of coastal wetlands in China

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ABSTRACT

We explored the potential for biodiversity offsetting to be applied in regions with considerable development pressure. We developed a method to identify suitable locations for restoration offsets and applied this to coastal reclamation in the Yellow River Delta region of China, an internationally important area for migratory birds, but in which 44% of wetlands have been reclaimed. We evaluated the suitability of sites for offsetting based on their ecological similarity to development sites, the potential of biota to migrate between sites and socio-economic criteria. We predicted that 60–100% of all reclamation in the Yellow River Delta between 1980 and 2015 could be theoretically offset provided no constraints were placed on where offsetting occurs within the region. However, where potential offset sites were constrained to areas with high suitability only 8–15% of historic coastal reclamation could be offset. Spatial options for offsetting also declined where time lags before restoration were longer. Our results indicated that strict in-kind biodiversity offsetting becomes increasingly challenging in highly modified landscapes because of a lack of spatial options for offsets and a tendency for potential offset sites to be dissimilar to the habitat that originally occurred on developed sites in these landscapes. Policies that seek to enable development within highly modified landscapes by providing flexibility for offsetting in space and time risk providing offsets that are ecologically dissimilar from development sites and have limited capacity for biota to migrate to or from them. Our methodology can be used as a planning tool to indicate the level of development within a landscape or region beyond which no net loss is unlikely to be feasible.

1. Introduction

Coastal wetlands have been reclaimed for other land uses across the world (MA (Millennium Ecosystem Assessment), 2005) and reclamation is likely to intensify because of the increasing scarcity of suitable land in coastal areas and continuing demand to develop these areas (MacKinnon et al., 2012). For example, between 1980 and 2015, 472 km^2 of coastal wetlands have been reclaimed in the Yellow River Delta region of China (Yu et al., 2017). While coastal land reclamation provides a lever for economic development (Murray et al., 2014), it also leads to the loss and degradation of habitat for biodiversity (Bulleri and Chapman, 2010; Chapman and Blockley, 2009), such as the macrozoobenthos (Dugan et al., 2008; Yan et al., 2015) which, in turn, are important for other biota such as migratory birds (Arocena, 2007; Evans et al., 1999; Goss-Custard et al., 2006).

Biodiversity offsetting is emerging as a key policy employed to manage the impacts of development on natural habitats, having been adopted in approximately 40 countries (Maron, 2015; Gibbons et al., 2018). Biodiversity offsetting is a policy instrument in which impacts on biodiversity from development are compensated with equivalent gains elsewhere (Moilanen et al., 2009a; ten Kate et al., 2004). The popularity of this policy instrument lies in its potential to simultaneously meet the objectives of biodiversity conservation and economic development (Bull et al., 2013). The main offset methods for coastal wetlands in China are the release of larvae and juveniles into coastal wetlands to increase their natural supply; the construction of artificial reefs to enhance habitats (Yu et al., 2017); establishing small nature reserves (State Forestry Administration, 2017; and financial compensation for restoring degraded ecosystems (Ali et al., 2018). However, no net loss using biodiversity offsetting can only be achieved where there is a sufficient area of suitable, or potentially suitable, habitat available for the biota affected by development.

The same principles used to identify protected areas can be used to identify suitable ‘offset receiving areas’ (Moilanen et al., 2009b). For example, Kujala et al. (2015) and Habib et al. (2013) used the software Zonation and Marxan respectively to identify potential offset areas.

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China mainly used the Analytic Hierarchy Process to identify suitable habitats which is critical to wetland restoration and biodiversity conservation (Dong et al., 2013). Although these methods have been useful illustrating where offsets should occur, one limitation of these methods is that they don't explicitly address some of the additional considerations that are important in the context of biodiversity offsetting (Bull et al., 2013) such as the potential quantum of biodiversity gain associated with protecting or restoring a site (i.e., additionality), the degree of similarity between development and offset sites (i.e., equivalence), proximity to affected areas, the cost of restoration and connectivity to other natural populations (BBOP, 2009).

In this study we sought to retrospectively offset land reclamation in the Yellow River Delta, China between 1980 and 2015 to explore trade-offs that must be considered when striving for no net loss of biodiversity in a region with considerable development pressure with delays before restoration is likely to occur. To do this we developed a method to identify potential offset locations taking into account key requirements for offsetting: the availability of land for offsetting, the equivalence of habitat in offset sites to the habitat affected by reclamation, ecological connectivity, the quantum of gain in biodiversity from the offset, the proximity of potential offset sites to impact sites, and land prices.

2. Materials and methods

2.1. Study sites

The study area (37°35′N to 38°12′N, 118°33′E to 119°20′E) comprises the following sites: (a) the entire coastline of the Yellow River Delta, located at the mouth of the Yellow River in Dongying City of Shandong Province, China; and (b) the south coast of Bohai Bay and the west coast of Laizhou Bay, also in Shandong Province (Fig. 1). Coastal wetlands have been reclaimed in the study area between 1980 and 2015 at an average rate of 13.5 km^2 per year (Yu et al., 2017). There is an urgent need to explore measures that mitigate the impacts of land reclamation on coastal habitat biodiversity. An effective strategy is likely to encompass biodiversity offset which is often included in a mitigation hierarchy as a last step after avoidance, reduction and restoration measures. The main approach in this region is the restoration of degraded habitat, the protection of areas where there is an imminent or projected loss of biodiversity, or financial compensation for restoring degraded ecosystems. However, there are few high-quality wetlands available for protection/averted loss in the selected study area; financial compensation is not underpinned by the principles usually associated with biodiversity offsetting, that outcomes should be equivalent or like-for-like and the overall objective is no net loss or net gain in biodiversity (Ali et al., 2018). Therefore, it is important to identify suitable restoration offset sites.

In the present study, we examined 12 combinations of habitat type and coastal land reclamation: three habitat types (tidal marsh, salt marsh and freshwater marsh) reclaimed for one of four alternative land uses (mariculture, oil field, salt pans or industrial constructions) (Appendix B). We identified potential offset sites as degraded areas generated by climatic droughts, decreased estuarine freshwater input, over-grazing (which cannot restored naturally) (Fan et al., 2012; Guan et al., 2001; He et al., 2017), and abandoned agricultural lands. A survey by Yu et al. (2017) indicated these sites contained few macrobenthos. We considered areas with relatively intact habitat provided little potential as offset sites because of existing protections to most of these sites. All landcover types were identified using data from the Landsat Multi-spectral Scanner and Thematic Mapper with a spatial resolution of tens of meters. These data have been widely used in coastal research (Tian et al., 2016).

Planning units could be defined with a grid of squares, lattice of hexagons, natural ecological divisions, or political/governmental divisions. We used grids of squares as planning units. Selecting an appropriate grid size represents a trade-off between precision and computational time. Using a planning unit of 900 m × 900 m we correctly mapped > 89% of different land use types.

Each grid was coded according to dominant land use as follows: (1) tidal march, (2) salt marsh, (3) freshwater marsh, (4) coastal land reclamation, (5) degraded areas, and (6) other land-use types. Degraded areas (5) were considered potential offset locations. Relatively mature forests and other valuable habitats (6) were not considered as potential offset sites due to existing protection, in which case any gains from using them as offset sites would not be additional.

2.1.1. Habitat indicators

We focused on seven habitat attributes that are indicative of habitat quality for macrobenthos (soil moisture, bulk density, salinity, pH, soil organic matter, soil total carbon and total nitrogren) (Li et al., 2016). Macrobenthos are an important component of biodiversity in coastal wetlands: they have been used as surrogates for wetland function (Balcombe et al., 2005), they represent a critical food resource for other species such as birds (Kristensen et al., 2014) and they play an important role in improving and preserving water quality through mineralization and recycling of organic matter (Everaert et al., 2013). We collected seven habitat attributes at 505 sites across the study area (Fig. 1). Three replicates of soil between 0–5 cm depth were collected at each site. Soil moisture content and bulk density were analyzed by weighing wet soil cores and re-weighing them after drying for 48 h at 60 °C (He et al., 2012). Pore-water salinity and pH were determined by measuring the resulting supernatant of a dry soil with deionized water (1:5 w/v) (Cui et al., 2011; Pennings et al., 2003). Soil organic matter was determined by the Walkley and Black methods (Bai et al., 2012; Thorne et al., 2014). Soil total carbon was measured with a total organic carbon analyzer (TOC-V, Japan) and total nitrogen with a continuous-flow analysis instrument (AA3, Europe). All field work was undertaken in 2016. We spatially interpolated each of the seven habitat attributes across the study area at a 900 m × 900 m grid cell resolution using Ordinary Kriging in ArcGIS 10.0 (see S1, Appendix A). The mean habitat traits of each patch were calculated with Zonal Statistics in ArcGIS.

2.2. Identify the suitability of offset locations

The process to identify potential offset locations is presented in a broadly chronological order, and some steps may be in parallel (Fig. 2). We identified the suitability of potential offset sites as the product of the similarity of the site to development sites, the probability of successful migration/dispersal of macrobenthos to the site and the availability of the site for offsetting. The closer the product was to 1, the higher the suitability of the offset site was. The suitability of a patch (which we define as a 900 m × 900 m planning unit) k as an offset location (P_k) was calculated as

$$P_k = S_k I_k E_k \tag{1}$$

where S_k is the similarity of habitat at the target patch to the habitat affected by the development, I_k is the probability of successful migration from a source patch to the target patch k, and E_k is the availability of patch k, which takes proximity to the impact site and land prices into consideration.

The product of S_k, I_k and E_k was divided into three discrete groups using the Jenks Natural Breaks classification in ArcGIS (Reyers et al., 2009; O'Farrell et al., 2010), which was low, medium and high suitability respectively. We used minimum offset ratios reported in Yu et al. (2017) for each of the 12 combinations of habitat type and development type that occurred across the study area from 1980 to 2015 (see Appendix B), and calculated the percentages of each combination of habitat and development type that could be theoretically offset when the time to restoration was 2, 5 or 20 years, which represents different estimates for macrobenthic communities (Yu et al., 2017; Warren et al., 2002; Moseman et al., 2004). Given we undertook this study in 2017,