Marine Pollution Bulletin 110 (2016) 250–260

Contents lists available at ScienceDirect

Marine Pollution Bulletin

journal homepage: www.elsevier.com/locate/marpolbul

Facilitating political decisions using species distribution models to assess restoration measures in heavily modified estuaries

Maike Heuner [a,c,*], Arnd Weber [a], Uwe Schröder [b], Birgit Kleinschmit [c], Boris Schröder [d,e]

[a] Department Ecological Interactions, Federal Institute of Hydrology, Am Mainzer Tor 1, 56068 Koblenz, Germany
[b] Department Vegetation Studies & Landscape Management, Federal Institute of Hydrology, Am Mainzer Tor 1, 56068 Koblenz, Germany
[c] Geoinformation in Environmental Planning Lab, Technische Universität Berlin, Straße des 17. Juni 145, 10623 Berlin, Germany
[d] Institute of Geoecology, Environmental Systems Analysis, Technische Universität Braunschweig, Langer Kamp 19c, 38106 Braunschweig, Germany
[e] Berlin-Brandenburg Institute of Advanced Biodiversity Research (BBIB), Altensteinstraße 6, 14195 Berlin, Germany

ARTICLE INFO

Article history:
Received 29 October 2015
Received in revised form 25 May 2016
Accepted 13 June 2016
Available online 20 June 2016

Keywords:
European Water Framework Directive
Northern Sea estuaries
Phragmites australis
Scirpus maritimus
Scirpus tabernaemontani
Restoration management

ABSTRACT

The European Water Framework Directive requires a good ecological potential for heavily modified water bodies. This standard has not been reached for most large estuaries by 2015. Management plans for estuaries fall short in linking implementations between restoration measures and underlying spatial analyses. The distribution of emergent macrophytes – as an indicator of habitat quality – is here used to assess the ecological potential. Emergent macrophytes are capable of settling on gentle tidal flats where hydrodynamic stress is comparatively low. Analyzing their habitats based on spatial data, we set up species distribution models with 'elevation relative to mean high water', 'mean bank slope', and 'length of bottom friction' from shallow water up to the vegetation belt as key predictors representing hydrodynamic stress. Effects of restoration scenarios on habitats were assessed applying these models. Our findings endorse species distribution models as crucial spatial planning tools for implementing restoration measures in modified estuaries.

1. Introduction

Estuaries are characterized by their nutritious marshes. Therefore, they have historically been in high demand for human settlement, resulting in an ongoing struggle to protect land for agricultural use against heavy storm surges and river flooding. The succession of land reclamation by organized diking and drainage districts clearly increased with technical improvements in the seventeenth century in Europe (Keddy, 2010). The remaining wetlands – without the embanked areas – were no longer able to dissipate the energy and water volume of storm surges and storm waves. Thus, the function of ecosystem-based shoreline protection became impaired. Embankment and diking disrupted the natural adaptive capacity of shorelines to keep up with sea level rise by sediment accretion (Temmerman et al., 2013). Subsequent deepening and widening of the river channels for economic navigation purposes exacerbates the loss of this natural adaptive capacity. As a consequence, shores in estuaries have been protected against erosion by ship-induced waves with stone linings (Coops and Geilen, 1996).

The intensive use of estuarine functions by, for example, agriculture, human settlement, and transport in the last centuries has caused a mismatch of ecosystem functions. For instance, the larger the extent of drained and embanked land is in order to provide edible plants and animals by agriculture, the smaller is the area available for water and sediment retention. Therefore, the win-win-effects for society and nature through regulating ecosystem services provided by ecosystem functions (Atkins et al., 2011; de Groot et al., 2010; MEA, 2005) are disturbed. Sustainable adaptation to sea level rise including an increasing resilience is a pivotal regulating ecosystem service and an increasingly important topic in flood protection (EC, 2015; Temmerman et al., 2013). Improving these services enhances other services from which the human population can benefit such as recreation or water quality regulation (cf. Atkins et al. (2011); Elliott et al. (2007); Needles et al. (2015)). Additionally, not all the land reclaimed in the past is needed to feed the human population nowadays and could be returned to a more natural state to improve the coastal protection. Therefore, we have an obligation to recreate characteristic estuarine habitats, reducing these past losses.

Notwithstanding the UNESCO Man and the Biosphere programme launched in 1970 (Dorst, 1971), the demand for maintenance, sustainable use, and recreation of estuarine habitats and other wetlands was adopted in the first international convention on wetlands of international importance (Ramsar convention) in 1971 with the number of participants steadily increasing (168 contracting parties by 2014) (Mauerhofer et al., 2015; Shine and de Klemm, 1999). Legal instruments at international, national and local levels (multilevel governance) are used to implement these multilateral environmental agreements (Shine and de Klemm, 1999). Multilevel wetland management is similarly structured in both

* Corresponding author at: Department Ecological Interactions, Federal Institute of Hydrology, Am Mainzer Tor 1, 56068 Koblenz, Germany.
E-mail address: heuner@bafg.de (M. Heuner).

http://dx.doi.org/10.1016/j.marpolbul.2016.06.056
0025-326X/

the USA and the European Union (Peterson et al., 2008; Pinto, 2015), both following the principle of "no more deterioration": The Clean Water Act is an instrument at federal level implemented by the US Environmental Agency (Kelly, 2001; Ravit and Weis, 2014), while the European directives (e.g., Habitats Directive 92/43/EEC, Water Framework Directive WFD 2000/60/EC) are instruments at the European level governed by the European Commission.

The Habitat Directive obliges EU member states to maintain or restore natural habitats and wild species to ensure their sustainable survival in Europe. This includes the unique structural and functional biodiversity of estuaries (Meire et al., 2005). Similarly, the WFD calls for protection and improvement inter alia of the ecological quality and function of estuarine waters (Borja, 2005). The goal of the WFD is to achieve a good chemical, ecological, and hydromorphological status or potential for all surface waters and groundwater by 2015. Most European estuaries are categorized as heavily modified water bodies (HMWB) (CIS, 2003) due to their particular uses such as navigation or land drainage. They can only reach a good or better potential through considerable hydromorphological changes (CIS, 2003, 2006) which requires innumerable habitat restorations. <30% of transitional water bodies including estuaries have a good ecological and hydromorphological status/ potential, and the majority has therefore failed to reach the goal by the end of 2015 (EEA, 2012a, b). Hence, the WFD objectives have to be achieved with more efforts in the 2nd and 3rd planning cycles with target dates 2021 and 2027 (EC, 2012).

In order to realize the objectives and implementing measures such as habitat restoration, lowering of river banks or removal of bank enforcement in detailed, spatially explicit analyses are needed in order to assess the effects of these measures on shoreline habitats on a local scale. However, the link between the implementation of restoration measures and spatial analyses needs to be strengthened in many European countries (EEA, 2012c). In the past, spatial analysis was usually restricted by administrative boundaries, whereas management plans refer to topographic/geographic boundaries (EEA, 2012c). The use of spatial analyses enhances the understanding of cause-effect relations as well as of the measures' effectiveness (Haasnoot and Wolfshaar, 2009). Comparing scenarios allows for evaluating their time and work load and the estimation of direct economic costs. Providing transparent descriptions of the measures as well as the basis for their site-specific assessments, spatial analyses also serve to promote the economic, social and territorial cohesion in policy (EEA, 2012c).

Species distribution models are a useful spatial analysis method and directly support management plans for species habitat recreation and mapping of suitable sites for ecological restoration (Alonso Ponce et al., 2010; Ferrier et al., 2002; Guisan and Thuiller, 2005; Pearce and Lindenmayer, 1998). However, we only found a few studies using species distribution models as a tool for the implementation of habitat restoration as required by the WFD and the habitat directive: i.e. models predicting in particular fish and amphibians in a Danube floodplain (Funk et al., 2013), models calculating the habitat suitability of willows along the Middle Elbe (Mosner et al., 2011), and models predicting the potential habitat for seagrass for the Baltic coast (Schubert et al., 2015) as well as for estuaries in Northern Spain (Valle et al., 2011). All these models were developed to identify suitable habitats for waterbodies with natural conditions.

In heavily modified estuaries, suitable sites are unlikely to be successfully identified using these models because of the large amount of engineered shorelines protecting reclaimed land. The responsible authorities propose restoration sites on the basis of diverse political considerations and their environmental suitability is often less important. Nevertheless, species distribution models can help to identify the necessary modifications in proposed restoration sites in order to reach appropriate environmental conditions for natural estuarine habitats according to the European directives.

Common plant species on estuarine shorelines are emergent macrophytes such as reed and sedges (Clevering and van Gulik, 1997; Coops and Geilen, 1996; Lillebo et al., 2003). These are able to settle on tidal flats, where they are capable of resisting hydrodynamic forces in the form of stress such as current velocities and wave heights (Silinski et al., 2015). Bottom friction on broad tidal flats (cf. Le Hir et al. (2000)) and low dissipative slopes (Bertness, 2006) obviously attenuate these stress drivers on the bare tidal flats located in front of the marsh edge. The elevation relative to mean high water (MHW) reflects the species responses to hydrodynamic forces, especially in low marshes. Acting as ecological engineers by, for example, filtering water (e.g., Clevering and van Gulik (1997); Smith et al. (2009); Zhao et al. (2004)), dissipating wave energy (e.g., Leonard and Reed (2002); Möller et al. (2011); Ysebaert et al. (2011)), trapping sediment (e.g., Rooth et al. (2003); Temmerman et al. (2004); Yang (1998)), and aerating the anoxic sediment (e.g., Brix et al. (1996); Jespersen et al. (1998); Nivala et al. (2013)), emergent macrophytes exert a clear positive effect on restoration areas (Elliott et al., 2007). Moreover, they are among angiosperms, which are classified as a biological quality element used to assess the ecological potential and thereby play an important role in satisfying present legal requirements of the WFD.

Successful restoration of estuarine vegetation needs suitable elevations (Temmerman et al., 2013). To our knowledge, species distribution models for estuarine shorelines with emergent macrophytes are not available. These models depict the abiotic conditions under which these species occur. This knowledge is crucial for the evaluation of different restoration scenarios in order to choose the scenario with the greatest habitat gain by the promoted species. Therefore we investigated the following questions:

1. Which are the key environmental predictors determining the distribution of emergent macrophytes on estuarine shorelines?
2. How should estuarine-engineered banks be restored to enable the development of naturally vegetated shorelines?

2. Materials and methods

2.1. Species and study sites

The ubiquitous species *Scirpus tabernaemontani* (C. C. Gmel.) Palla, *Scirpus maritimus* (L.) Palla, and *Phragmites australis* (Cav.) Trin. ex Steud. are emergent, clonal, fast-growing macrophytes with monotypic stands. They commonly form a distinct plant zonation from low to high elevations and constitute pristine vegetation in lower marshes of the freshwater and brackish part of the Elbe and Weser estuaries. (Focke, 1915; Kötter, 1961). Both rivers are located in North-western Germany entering the North Sea and represent characteristic environmental conditions for European estuaries.

Focusing on our study sites (Fig. 1A), the mean tidal range (2001 − 2010) for the Weser estuary was between 3.8 m (gauge Bremerhaven) and 3.9 m (gauge Vegesack) and for the Elbe estuary between 2.9 m (gauge Otterndorf) and 3.6 m (gauge St. Pauli). Based on averaged parameters at the study sites, the Weser estuary is smaller than the Elbe estuary in depth (13.7 m vs. 18.1 m), discharge (325 m^3/s vs. 707 m^3/s), width (965 m vs. 2033 m), and stream velocity (0.1–0.6 m/s vs. 0.2–0.9 m/s, Vandenbruwaene et al. (2013)). The Elbe main channel exhibits larger transport frequencies with 620 mean passages per week in contrast to 200 passages on the Weser estuary (Peters et al., 2013). In the port of Hamburg (Elbe River) vessels sizes with 14,041 mean gross registered tonnage (Federal Waterways and Shipping Directorate North, 2011) exceed the 9760 mean gross registered tonnage (Senator for Economics and Ports of the Free Hanseatic City of Bremen, 2013) in the port of Bremen (Weser estuary). The vessels on the Elbe have maximum drafts of 15.4 m (Peters et al., 2013) compared to vessels on the Weser with 10.7 m (BAW, 2006). However, the maximum drawdown (Elbe: 0.5 m - 1.2 m, Weser: 0.6 m - 1.3 m) and maximum ship-induced wave heights (primary waves: Elbe: 0.5 m - 1.5 m, Weser: 0.6 m - 1.5 m; secondary waves: Elbe: 1.0 m - 1.4 m, Weser: 0.9 m - 1.2 m) in both