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Salt-marsh restoration: evaluating the success of de-embankments in north-west Europe

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Abstract

De-embankment of historically reclaimed salt marshes has become a widespread option for re-creating salt marshes, but to date little information exists on the success of de-embankments. One reason is the absence of pre-defined targets, impeding the measurement of success. In this review, success has been measured as a saturation index, where the presence of target plant species in a restoration site is expressed as a percentage of a regional target species pool. This review is intended to evaluate and compare success of many different sites on an idealistic concept where all regional target species have the potential to establish in a site, but may not actually do so because the site is unsuitable or inaccessible. Factors affecting suitability and accessibility and management options to increase regional species diversity are discussed. The results show that many sites contain less than 50% of the regional target species, especially when sites are smaller than 30 ha. Higher species diversity is observed for sites exceeding 100 ha and for sites with the largest elevational range within mean high water neap to mean high water spring tide. Most sites younger than 20 years contain more target species than older sites. For future de-embankments it is recommended that clear targets are set from the start. This brings along the need for monitoring. Only 37 out of 70 sites with de-embankment were monitored for plant species assemblages. Setting targets will also allow adaptive management of the site. Management options that are likely to result in higher species diversity are the construction and maintenance of drainage structures and the implementation of a grazing or mowing regime.

Keywords: Managed realignment; Regional species pool; Restoration success; Saturation index; Summerpolder; Target species

1. Introduction

Over the past centuries, large areas of salt marsh have been reclaimed from the sea by the construction of embankments (Dijkema, 1987; Pethick, 2002). These embankments would either function as the main sea defence, protecting the hinterland from all tidal flooding, or as low summerdikes in front of an existing seawall, protecting the reclaimed land (polder) from normal tidal inundations, but not during winter storms (Bakker et al., 2002). The polder was usually used for intensive agricultural exploitation, which often involved the construction and/or maintenance of drainage structures and the application of fertiliser. As a result, the characteristic halophytic communities have largely disappeared. Continuing sediment accumulation in front of the embankments resulted in the development of new salt marshes and, after sufficient vertical accretion, these could in turn be reclaimed.

This process of successive reclamations has now become less acceptable for various reasons. First, the need for extension of agricultural areas has diminished

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(Bakker et al., 1997). At the same time, it has been realised that salt-marsh communities are important habitats that need to be preserved. Apart from their nature conservation interest, salt marshes are important for coastal defence, as they act as a natural buffer for dissipating wave energy (Möller et al., 1996). Moreover, regular tidal inundation on salt marshes ensures the input of fresh sediment, the rate of which may be high enough to compensate for current and future sea level rise. In contrast, most embanked polders are characterised by a sediment deficit and may subside to below mean sea level. The presence of a salt marsh in front of a seawall will thus improve safety of the hinterland and reduce the cost involved in seawall maintenance (King and Lester, 1995). Especially in the United Kingdom, where a combination of the sinking of the land and rising sea level has caused extensive erosion of salt marshes (Cooper et al., 2001), coastal defence is the main incentive for de-embankment (Pethick, 2002).

The idea for using de-embankment to re-create salt marshes is developed from evidence of spontaneous salt-marsh formation after accidental breaching of seawalls due to storm tides (French, 1999). However, not all natural breaches have resulted in successful salt-marsh development, hence it will be important to evaluate the success of different sites in order to provide insight into possible outcomes for deliberate de-embankments. In this paper the results of several natural and deliberate de-embankment cases will be discussed. Under deliberate de-embankment we include complete or partial removal of a seawall or summerdike as well as regulated tidal exchange where sluices or one-way valves have been inserted in the embankment to allow specified tidal amplitude (Lamberth and Haycock, 2001). The aims of the paper are: (1) to present an overview of sites subjected to natural or deliberate breaching of seawalls or summerdikes or regulated tidal exchange, (2) to introduce a method for evaluating restoration success, (3) to compare restoration success of different sites, (4) to determine which factors are likely to affect restoration success, and (5) to provide recommendations for future restoration schemes.

2. Selection and general description of study sites

A literature survey was conducted to identify restoration sites across north-west Europe. This resulted in a total of 89 sites (Appendix A). Sites that had no information on location, type of scheme, size or year of restoration, or sites for which the scheduled restoration had not been implemented, were not included in the analysis. This reduced their number to 70. The majority of the restoration sites (48) is located in the United Kingdom, in particular on the south-east coast (Fig. 1). However, they rarely exceed 100 ha in size and the total restoration area of these 48 sites (2007 ha) is lower than that of the ten German sites (2590 ha) (Fig. 2). For fifty percent of the sites, embankments were accidentally breached during storm surges, in particular in 1897 and 1953. The oldest accidentally breached embankment reported in the literature dates back to 1802. The first deliberate de-embankment was executed in 1991 (Northey Island, site Nr. 30b), after which between one and seven new de-embankments were initiated in north-west Europe each year (Fig. 3). Habitat creation/restoration is the most common reason for de-embankment, especially in the Netherlands, Belgium and Germany (Fig. 3). In many cases this habitat creation is driven by European legislation e.g., the Habitats Directive, which requires member states to designate special areas of conservation (Pethick, 2002). Flood defence is another major reason for de-embankment, as the re-creation of a saltmarsh in front of a sea defence is considered a cost-effective way of improving safety to the hinterland.

3. How to evaluate success?

3.1. Definition of success

There is much recent debate on the question of how to define restoration success. One approach is to determine whether or not the terms set in an agreement, contract or permit have been met (Kentula, 2000). Use of compliance success is appropriate whenever restoration targets are set beforehand. However, in many cases there are no clearly specified targets. In the present overview, for example, many sites had no clear pre-defined targets. Other possibilities for assessing success are to compare the ecological structure or functioning of a restored site with one or more reference sites (Thom et al., 2002; Edwards and Proffitt, 2003). However, the choice of the reference sites strongly affects the outcome of such a comparison (Kentula, 2000; Morgan and Short, 2002). Besides, comparing conditions with a natural reference system may not be realistic or appropriate because restoration may start on different substrate or different elevation (Thom, 2000), or because the reference site itself may be degraded. Historical reference has also been used for assessing success, in which case success criteria have often been based on the situation before the industrial revolution and before the application of artificial fertiliser (De Jonge and De Jong, 2002). However, when taking into account the increased human population with the accompanying levels of pollution, landscape fragmentation and species extinctions, a return to pre-industrial revolution ecosystems is hardly achievable.

Fig. 1. Location of salt-marsh restoration sites for which information on year of breach, type of scheme and area was available ($n = 70$). Different symbols are used for different countries. For sites names and detailed information see Appendix A. Lettering to indicate multiple sites at one location are omitted on the map for clarity.

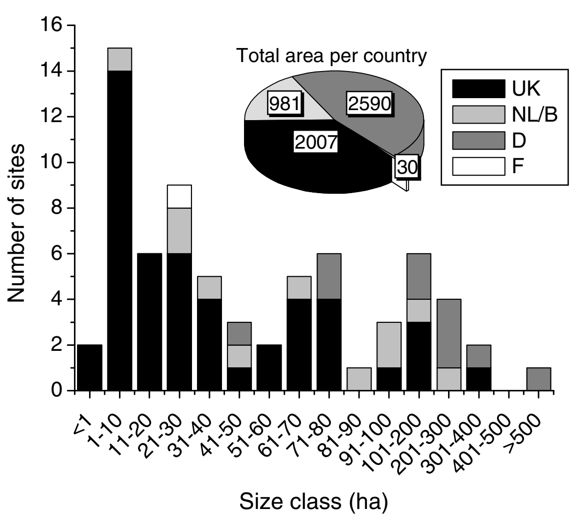

Fig. 2. Frequency distribution of sizes of sites ($n = 70$) and total area per country (UK: $n = 48$; NL/B: $n = 11$; D: $n = 10$; F: $n = 1$).

Part of the debate on how to define success focuses on the question of whether the aim should be the restoration of the structure of an ecosystem or its functioning. Zedler and Lindig-Cisneros (2000) defined structure as a condition at one point in time (e.g., species diversity) and function as a process that occurs over time (e.g., primary production), and concluded that structural measures are often (wrongly) used as substitutes for functioning. Zedler and Callaway (1999) further point out that the restoration of functionality often takes longer than the restoration of the plant communities themselves. Although we do not dispute the fact that successful restoration should include proper structure and functioning of the system, we focus on restoration of the structural component as the first and most important stage in salt-marsh restoration. By definition, coastal salt marshes are referred to as the vegetated part between land and sea, receiving frequent tidal inundation (Adam, 1990). Once the vegetation has established it can serve different functions, e.g., sediment trapping, nutrient cycling, dissipation of wave energy, spawning area for fish, feeding, breeding and resting area for birds etc. If these functions do not follow upon the restoration of the vegetation, it can be because the site is not accessible or because the habitat structure is not suitable (e.g., geese will not use a site if it is dominated by tall plants) (van der Wal et al., 2000). In the latter case, management strategies may be incorporated to improve the structure of the site (e.g., grazing or mowing of tall vegetation). It should be noted however, that different functions may require different structural components. Hence, in order for a

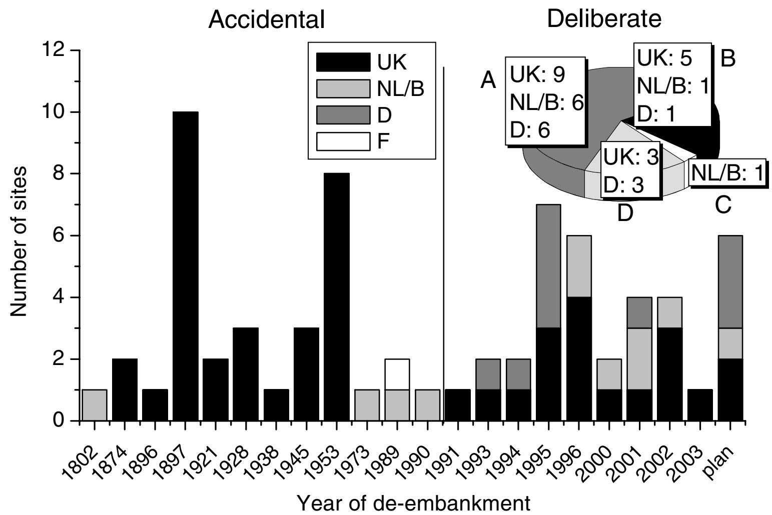

Fig. 3. Year of de-embankment and main reasons for deliberate de-embankment with the number of sites per country for each category (A: habitat creation or restoration; B: flood defence; C: gaining experience; D: unknown).

restoration site to serve as many different functions as possible, high structural diversity should be created. In the present paper we will focus on plant species diversity based on a well-defined target species list as a first step for measuring structural success in restoration sites.

3.2. Applying the species pool concept for measuring success

For community restoration it is important to know which species are part of the target community and how they can arrive in the target community. These questions can be addressed through the species pool concept, reviewed by Zobel et al. (1998). In their review, three species pools are distinguished, each at a different spatial scale: (i) the regional species pool: a set of species occurring in the region and capable of co-existing in the target community, (ii) the local species pool: a set of species occurring in the landscape surrounding a target community, (iii) the community species pool: a set of species present in the target community. Various abiotic and biotic processes will act as filters between the different pools and determine whether a species from the regional or local pool will actually arrive and establish in the target area. The actual determination of the species pools is still in its infancy, but a promising approach is to select species from the local or regional flora based on phytosociological similarity (Zobel et al., 1998). We have applied this approach to define a regional target species list for north-west European salt-marsh and brackish-water plant communities. This regional species pool should include all species that have the potential to establish in a salt-marsh restoration site of the region concerned if the site were suitable and accessible.

3.3. Determining the regional species pool for north-west European salt marshes

On the basis of a differential influence of climate and sea currents on the distribution of salt-marsh plants in north-west Europe, we have classified our study sites into two distinct biogeographical regions, following Dijkema et al. (1984). The two regions are: (i) the Central North-Atlantic, extending from Scotland and south Scandinavia to North France, and (ii) the Southern North-Atlantic, covering south and south-east England, Brittany, south-west France and north-west Spain (Fig. 4). In addition, the German Baltic shore is treated as a separate region because the salinity of the submerging water, tidal range and geomorphology are very different from the North-Atlantic region (Dijkema, 1990).

For each region, typical salt-marsh communities were identified from the extensive work on National Vegetation Classification surveys by Schaminée et al. (1998) for the Central North-Atlantic and Rodwell (2000) for the Southern North-Atlantic, and a paper by Krisch (1990) for the German Baltic region (Appendix B). Species were included in the target species list if they occurred in 61% or more of the phytosociological relevés of each salt-marsh community. This minimum percentage of occurrence ensured that all species characteristic of salt-marsh communities were included whereas non-typical species were excluded. No distinction has been made between different *Salicornia* species, because of difficulties in correctly identifying

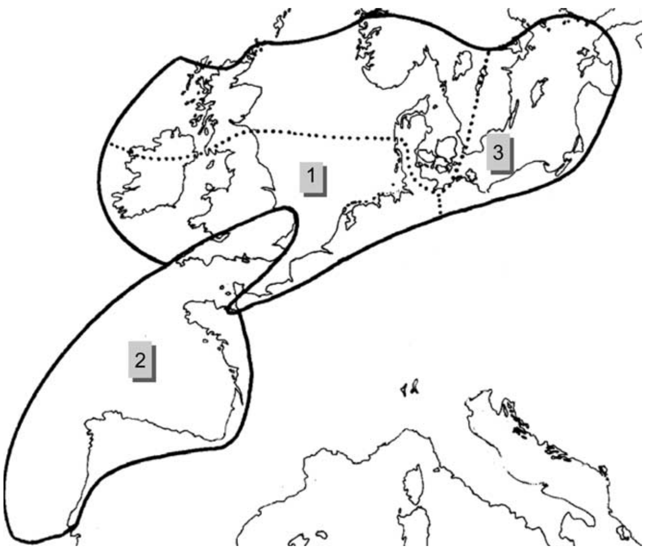

Fig. 4. Biogeographical regions of salt-marsh vegetation. 1, Central North-Atlantic; 2, Southern North-Atlantic; 3, German Baltic (adapted from Dijkema et al. (1984)).

these species in the field. The procedure for selecting target species for the regional species pool resulted in a total of 39 species for the Central North-Atlantic, 34 for the Southern North-Atlantic and 27 for the German Baltic region. The names of these species and a number indicating for which region the species is considered a target species are shown in the first two columns of Table 1.

4. Evaluating restoration success

For the evaluation and comparison of the success of different salt-marsh restoration projects we have used a saturation index, where the presence of all target plant species in a restoration site is expressed as a percentage of the total regional target species pool of the region concerned (i.e., 39 species for the Central North-Atlantic, 34 for the Southern North-Atlantic and 27 for the German Baltic region). We realise that this index does not take into account important drivers of diversity, such as size of the site, age and elevation range, which would have resulted in a more realistic, but also site specific evaluation. Instead, our intention is to evaluate and compare success of all the sites identified in this review on an idealistic concept where all regional target species have the potential to establish in a site, but may not actually do so because the site is unsuitable or inaccessible. Factors affecting the presence or absence of certain species are discussed later and this information can be used by site managers to determine which management options may be required to increase the chance of certain plant species establishing in the site.

Species lists were available for only 37 out of the 70 study sites (Table 1). The saturation index for the different sites ranged from 18% to 64% (Table 1). In comparison, the saturation index of 40 established marshes in the Wadden Sea region, ranged from 56% to 92% (Dijkema and Wolff, 1983). Restoration sites in the United Kingdom were the least diverse, with the majority of the sites having saturation indices below 30% (Fig. 5). Species that were absent from all restoration sites included *Spartina maritima*, *Poa subcoerulea*, *Puccinellia fasciculata*, *Carex serotina*, *Blysmus rufus*, *Oenanthe lachenalii*, *Ononis repens spinosa*, *Limonium binervosum*, *Frankenia laevis* and *Limonium bellidifolium*. Many of these species are characteristic of high-marsh and transition state communities. More research is needed to establish whether these species are nationally or regionally rare and missed as a result of insufficient sampling effort, or whether their absence is due to limited dispersal capabilities, abiotic or biotic constraints within the restoration sites. The most common species were *Salicornia* spp., *Suaeda maritima*, *Aster tripolium* and *Puccinellia maritima*, which were encountered in more than 80% of the sites (Fig. 6). These species often occurred in more than 61% of the plots in a particular restoration site and are characteristic of pioneer and low-marsh communities.

5. Factors affecting restoration success

5.1. Suitability

5.1.1. Surface elevation

In salt-marsh systems, elevation in relation to tidal inundation is generally accepted as the major abiotic factor governing the establishment and survival of halophytes at different zones within the range from mean high water neap (MHWN) to mean high water spring (MHWS) tide levels. In the present study we have examined the relationship between elevational range and restoration success by expressing the difference between maximum and minimum elevation recorded within a site as a percentage within the range from MHWN to MHWS. The results show that the elevational range is positively related to the saturation index ($R^2 = 0.37$, $P < 0.05$) (Fig. 7, UK sites only, MHWN and MHWS tide levels from Pye and French (1993)). Remarkably, many sites occupy less than 50% of the elevational range from MHWN to MHWS tide levels, hence these sites do not have the full restoration potential. Elevation has also been identified as the primary factor controlling species composition in restored salt marshes in the USA. Thom et al. (2002), for example, observed that their study site had subsided approximately one meter during the 70

Table 1
Target species of the regional species pool and frequency abundance (I = 1–20%, II = 21–40%, III = 41–60%, IV = 61–80%, V = 81–100%., P = present) of target species in 37 restoration sites with saturation index per site

Site number	**Region**	**62**	**40**	**13**	**20a**	**20b**	**8a**	**8b**	**16**	**29a**	**29b**	**21b**	**2**	**47**	**19**	**37**	**25**	**48**	**39**	**5**	**46a**	**46b**	**59**	**77a**	**61**	**66**	**76**	**32**	**44**	**67**	**36**	**30b**	**57**	**65**	**1a**	**58**	**22**	**50**
Years after breach		**193**	**119**	**119**	**96**	**96**	**96**	**96**	**96**	**96**	**96**	**72**	**72**	**55**	**48**	**48**	**40**	**40**	**40**	**40**	**40**	**40**	**23**	**13**	**8**	**8**	**7**	**7**	**6**	**5**	**4**	**3**	**3**	**3**	**2**	**1**	**1**	**1**
Number of quadrats		**107**	**5**	**4**	**7**	**5**	**9**	**10**	**6**	**11**	**12**	**6**	**7**	**8**	**10**	**6**	**11**	**8**	**12**	**5**	**6**	**4**	**26**	**10**	**48**	**12**	**?**	**620**	**7500**		**48**	**50**	**221**		**144**	**820**	**160**	**7**
Quadrat size (m2)																												**1**	**1**		**1**	**1**			**1**	**100**	**1**	
Spartina maritima	1,2																																					
Spartina anglica	1,2	I	I	I	II	I		I		I		I			I		II	I	II	I	I	I	I	I	I			I	I			I	I			I	II	
Salicornia spp.	1,2,3	I	II	III	II	IV	III	III	IV	III	III	I	II	II	I	IV	II	III	III	I	II	IV	III		I	I	P	IV	III	I	II	V	II	P	I	V	IV	
Suaeda maritima	1,2,3	II	II	III	II	I	II	III	II	III	III	III	III	II	III	III	II	II	II	III	II	I	V			II		II	I			IV	I	P	I	V	I	
Aster tripolium	1,2,3		II	III	II	I	II	III	II	III	III	II	II	I	II	III	II	I	II	I	I	III	V	I	IV	V	P	I	I	I	II	I	II	P		V	II	
Plantago maritima	1,2,3	I					I		I		I						I	I						I	I	I				II			I	P		II		
Triglochin maritima	1,2,3				I		I	I										I						II	I		P			II				P		I		
Spergularia media	1	I																					I		I	I							I	P		II		
Limonium vulgare	1,2	I	III		II	II	II	I	II	I	I	I	I	III		I	III		I		I	I			I			I	I			I	I					
Atriplex portulacoides	1,2	I	II	II	I	I	III	III	III	II	III	IV	II	III	I	I	II		I	III	II	I	III					II	I			I	I		I	I	II	
Glaux maritima	1,2,3	I															I	I					I	II	II	III	P						I	P		I		I
Poa subcoerulea	3																																					
Puccinellia maritima	1,2,3	I	IV	II	IV	V	III	IV	III	I	I	III	III	IV	IV	III	I	II	III	III	III	IV	V	I	IV	II	P	III	I		I	II	I	P	I	V	III	
Atriplex prostrata	1,2	II																I					IV	I	III	V		I	I		I	I	I	P	I	V		
Spergularia marina	1,2,3	I				I			I					I		I								I	I		P	I	I	I	I	I	I	P	I	V		II
Puccinellia distans	1,2,3																										P							P		I		II
Puccinellia fasciculata	1																																					
Hordeum marinum	1																																					
Juncus gerardi	1,2,3	II																						IV	I	I	P			III			II	P		I		III
Festuca rubra	1,2	I															I	II						I	I	V		I			I	I	I	P		I		III
Armeria maritima	2																																					
Carex extensa	1																																I	P				
Carex serotina	3																																					
Centaurium pulchellum	1																																I	P				
Blysmus rufus	1,3																																					
Seriphidium maritimum	1,2													I												II									I	I		
Elytrigia atherica	1,2	III					I										I						I		II	V		I	I		I	I	I		I			
Juncus maritimus	1,2,3	I																																				
Oenanthe lachenalii	1,2,3																																					
Samolus valerandi	3																										P											
Potentilla anserina	1,2,3	I																							I	II				I	I		II	P		V		III
Agrostis stolonifera	1,2,3	II																						V	IV	III		I		IV	III		II	P	I	V		V
Elytrigia repens	2,3																							I				I		II	I				I			
Leontodon autumnalis	2																																					
Trifolium fragiferum	1,3	I																							I		P			III			I	P		I		II

Site number	Region	62	40	13	20a	20b	8a	8b	16	29a	29b	21b	2	47	19	37	25	48	39	5	46a	46b	59	77a	61	66	76	32	44	67	36	30b	57	65	1a	58	22	50
Years after breach		193	119	119	96	96	96	96	96	96	96	72	72	55	48	48	40	40	40	40	40	40	23	13	8	8	7	7	6	5	4	3	3	3	2	1	1	1
Number of quadrats		107	5	4	7	5	9	10	6	11	12	6	7	8	10	6	11	8	12	5	6	4	26	10	48	12	?	620	7500		48	50	221		144	820	160	7
Quadrat size (m2)																												1	1		1	1			1	100	1	
Carex distans	1,3	I																															I	P				
Ononis repens spinosa	1																																					
Lotus corniculatus	1	I																																	I			
Trifolium repens	1,2	I																						I	I						I		I	P		I		IV
Sagina maritima	1	I																																				
Plantago coronopus	1	I																															I					
Cochlearia danica	1	I																																				
Festuca arundinacea	1	I																						II										P				I
Bulboschoenus maritimus	1,2,3																							III	IV		P			I			I	P		I		
Schoenoplectus tabernaemonti	1,2,3																										P						I	P				
Phragmites australis	3																													II								
Triglochin palustris	3																										P											
Eleocharis uniglumis	3																										P											
Juncus articulatus	3																													III								
Sarcocornia perennis	2																											I	I			I						
Inula crithmoides	2		I									I		I																								
Suaeda vera	2																											I										
Limonium binervosum	2																																					
Frankenia laevis	2																																					
Limonium bellidifolium	2																																					
Total	39,34,27	25	8	6	8	8	9	8	8	7	7	8	6	9	6	7	11	10	7	6	7	7	10	15	19	14	14	15	11	13	11	12	25	23	11	21	6	10
Saturation index	100	64	24	18	24	24	26	24	24	21	21	24	18	26	18	21	32	29	21	18	21	21	26	44	49	36	52	44	32	48	32	35	64	59	32	54	18	26
Reference:		1	2	2	2	2	2	2	2	2	2	2	2	2	2	2	2	2	2	2	2	2	3	5	4	6	7	8	10	9	16	11	12	13	14	15	16	17

Sites are listed in order of decreasing restoration time. The first column after the species list shows the regional species pool (1 = Central North-Atlantic, 2 = Southern North-Atlantic, 3 = German Baltic).

1. van Dort and Leusink (1998); 2. Burd et al. (1994); 3. van Duin et al. (1997); 4. Koppejan (2000); 5. Armel Dausse, pers. comm.; 6. Sabine Arens, pers. comm.; 7. Zander (2002); 8. Pers. obs.; 9. Bernhardt and Koch (2003); 10. Reading et al. (2002); 11. Dagley (1995); 12. van Gennip and Knotters (2002); 13. Främbs et al. (2000); 14. Diack (1998); 15. van Duin et al. (2003); 16. Angus Garbutt, pers comm.; 17. www.Groningerlandschap.nl/dollard/breebaartporjectvegetatie.htm.

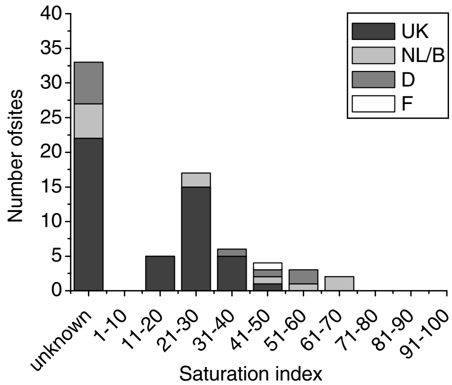

Fig. 5. Frequency distribution of saturation indices for 70 de-embankment sites.

years of embankment. As a result, restoration resulted in a different species composition than was anticipated from historical and nearby references. Other studies reveal that the rate at which vegetation develops in de-embanked sites is determined by the initial elevation (Cornu and Sadro, 2002; Williams and Orr, 2002) or that sites lower than 1.5 m below high water spring tides will fail to colonise with salt-marsh vegetation (Pethick, 2002). However, initially low elevation in itself may not be a problem if sedimentation rates are high enough. Data on sedimentation rates were available for 26 sites in our review. Because the techniques used for measuring sedimentation in these sites do not differentiate between net accretion and the effect of soil shrinkage and compaction, it is more appropriate to speak of surface elevation change, defined as the change in elevation relative to a sub-surface datum (Cahoon et al., 1995). Surface elevation change decreases linearly with the age of the restoration sites ($R^2 = 0.55$, $P < 0.0001$) (Fig. 8). Sites for which the technique used to measure surface elevation change could not be determined (Hauener Hooge, Nr. 66 and Sieperdachor, Nr. 61) were not included in the analysis. Blackshore Mill (Nr. 6) and Bulcamp Marsh

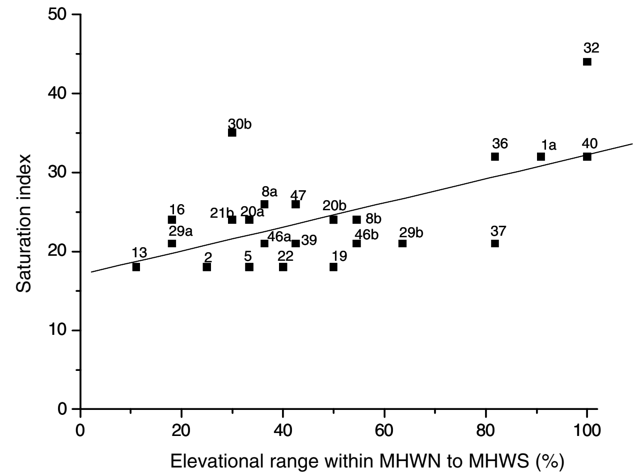

Fig. 7. Relationship between saturation index and percentage elevational range falling within mean high water neap (MHWN) to mean high water spring (MHWS) tide levels ($n = 24$, $P < 0.01$, $R^2 = 0.37$).

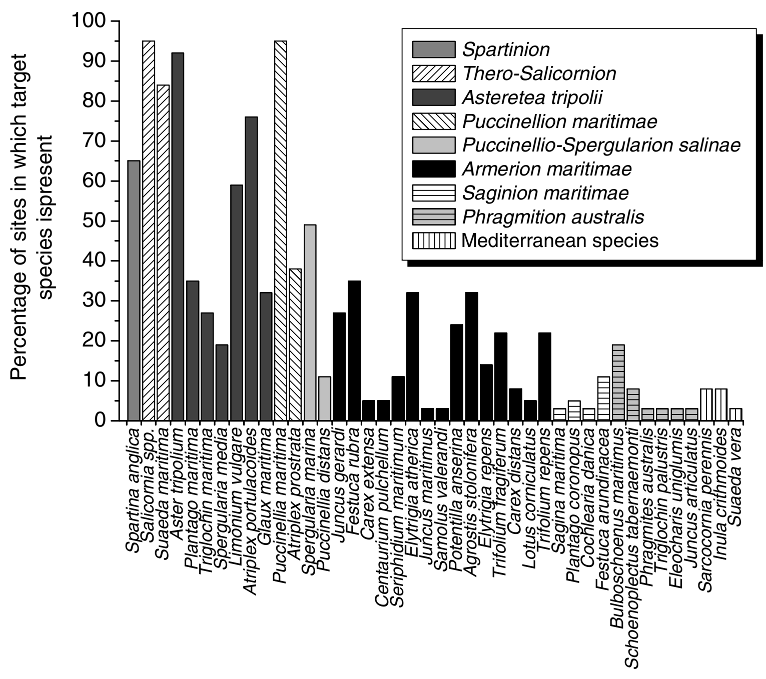

Fig. 6. Percentage of sites in which a target species occurs ($n = 37$). Species are arranged by their syntaxonomical class (Schaminée et al., 1998).

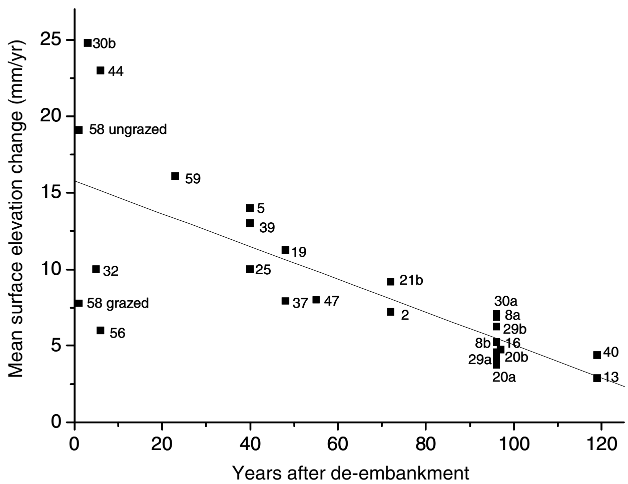

Fig. 8. Relationship between surface elevation change and restoration time ($n = 26$, $P < 0.0001$, $R^2 = 0.55$).

(Nr. 11) were also excluded from the analysis as sedimentation–erosion measurements were taken recently, whereas the sites were 53 and 45 years old, respectively. The results suggest that the elevation of the sites, which are likely to have subsided considerably during the period of embankment due to soil shrinkage and a lack of fresh sediment input, can increase rapidly during the first years after de-embankment. Over time, when pre-reclamation levels or levels similar to existing marshes and suitable for vegetation establishment are gained, the rate of surface elevation change is likely to decrease. Morgan and Short (2002) also observed a higher amount of deposited sediment in the younger sites compared to older ones, in a comparison of six salt-marsh restoration sites in New Hampshire, and the same observation at 15 re-flooded sites in San Francisco Bay formed the bases for a conceptual model of salt-marsh plain evolution with time since breaching (Williams and Orr, 2002). The decrease in sedimentation rates over time after de-embankment may be explained by the fact that once the substrate has become high enough for vegetation to colonise, accretion rates will decline due to less frequent flooding (Brown et al., 1999; Bakker et al., 2002) and because much sediment is trapped by vegetation and is therefore unavailable for the interior marsh (Adam, 1990; Schröder et al., 2002). *Puccinellia maritima*, has been identified as a key species in the process of trapping and stabilising sediment on European marshes (Andresen et al., 1990; Langlois et al., 2003).

For the sustainability of re-created and established salt marshes it is required that rates of surface elevation change are at least equal to local rates of relative sea-level rise (Reed et al., 1999). For 25 sites in the present study, surface elevation change is higher than relative sea-level rise, which is in the order of 1.0–3.0 mm/yr (Pye and French, 1993; van Duin et al., 1997). At Canvey point, (Nr. 13), rates of surface elevation change are lower than the rate of relative sea-level rise.

5.1.2. Size of restoration sites

It has been established that, for a variety of organisms and habitats, a linear relationship exists between the number of species and the size of the area (plotted on a log scale) (Begon et al., 1996). Therefore, the size of restoration sites may be an important determinant of success in restoration sites. Indeed, a significant relationship ($P < 0.05$) between the percentage of regional target species and size of the restoration area is observed for our study sites (Fig. 9), although the regression coefficient is low ($R^2 = 0.25$). The data in Fig. 9 suggest that restoration sites should be at least 30 ha in order to be able to harbour 50% or more of the target species. The best results are found for sites larger than 100 ha. It should be noted however, that the width of a site (i.e., the line perpendicular to the coastline) is likely to be more important than the length (i.e., the line parallel to the coastline), due to zonational processes leading to higher species diversity.

5.1.3. Soil salinity

Soil salinity is an important factor affecting the composition of salt-marsh vegetation. High salinities for example may prevent germination and seedling establishment (De Jonge and De Jong, 2002), whereas low salinities allow glycophytes to outcompete halophytes (Adam, 1990; Zedler and Callaway, 2001). Monitoring of soil salinity during salt-marsh restoration will thus be helpful for the evaluation of success. However,

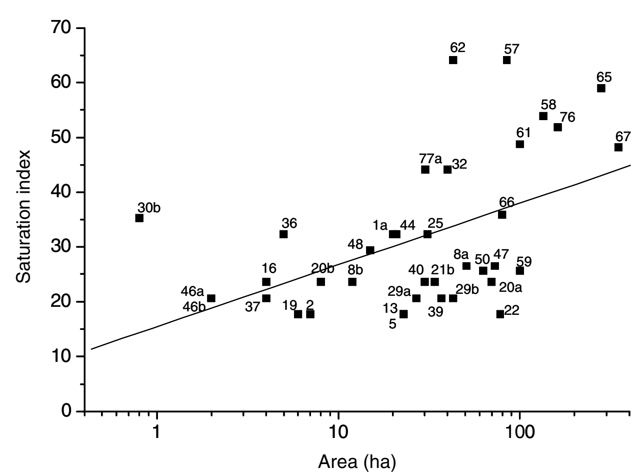

Fig. 9. Relationship between saturation index and size of restoration sites ($n = 37$, $P < 0.01$, $R^2 = 0.25$).

in this review only three sites were found to have measurements on soil salinity. At one of these sites (Noord Friesland Buitendijk Nr. 58), soil salinity of the de-embanked polder was compared to that of the fronting upper marsh. Results show that prior to breaching the summerdike, salinity levels in the summerpolder were at most 20% of those of the established marsh, whereas one year after the breach they had risen to 70%. Much lower levels were found at Berensch (Nr. 65) where soils had a 6–15‰ salinity, which was ascribed to restricted tidal flooding (Främbs et al., 2000). Together with temperature and rainfall, salinity of the incoming water will be a major determinant of soil salinity in de-embanked sites. In the Baltic sea for example, salinities of open water are between 5‰ and 10‰ (Dijkema, 1990). In this region, a 5–13‰ soil salinity was reported for the restoration site Ziesetal (Nr. 76).

5.2. Accessibility

5.2.1. Source area

A prerequisite for the successful restoration of salt-marsh communities is the availability of a target species source and the ability of the species to reach the target area. The best results may be expected when the target species are still present in the community species pool of the target area, which consists of the established vegetation and the soil seed bank (Zobel, 1997; Zobel et al., 1998). The contribution of the latter however, quickly declines with time after embankment, as many salt-marsh species do not build up a long-term persistent seed bank (Thompson et al., 1997; Wolters and Bakker, 2002). Unfortunately, the community species pool before de-embankment has rarely been assessed. In fact, only one site (Noord Friesland Buitendijks, site Nr. 58) in this review has quantitative information on both the established vegetation and the soil seed bank before de-embankment (Bakker et al., 2001; van Duin et al., 2003). This study shows that 49% of the species of the regional pool were already present in the community species pool before de-embankment, mainly as a result of high tides flooding over the low summerdike during storms in winter. Not surprisingly therefore, restoration of this site is proceeding rapidly with 54% of the target species establishing in the vegetation within one year after de-embankment (Table 2). However, in cases where embankments have functioned as the main sea defence fronting the restoration site, it is reasonable to assume that target species will be absent from the community species pool. Their natural establishment will therefore depend on dispersal from other areas, i.e., the local or regional species pool. It is generally assumed that the distance between the target area and a target species source will largely determine the chance of a species arriving in the target area. Thus, better results may be expected when an established salt marsh (i.e., local species pool) is directly adjacent to a de-embanked site. Indeed, a comparison between the presence of target species in the target area with the local pool, taking into account only species of the regional target list, shows that the composition of restoration sites and adjacent marshes closely resemble each other even within a short period of time (<5 years) (Table 2). An exception is polder Breebaart, (Site Nr. 50) where only 48% of the target species present in the local pool have established in the site within one year. This might be explained by the fact that the only opening through which tidal water can enter the site is a 1 m by 2 m wide sluice, which may form a barrier to successful seed dispersal. In some situations, the presence or absence of a local source of target species does not seem to be the main factor governing the establishment of a species in the target area. A comparison between Tollesbury (adjacent to established marsh) and Saltram (nearest marsh >16 km away) for example, shows that 26% and 32% of the target species have established in the site four years after de-embankment, respectively (Reading et al., 2002). On the other hand, Onaindia et al. (2001) concluded that 20 and 35 years after the col-

Table 2
Number of target species in Regional (R), Local (L) and Community (C) species pool, number of species shared between local and community species pool and percentage of target species (community vs regional, local vs regional and community vs local) for six sites in different regions

Site number & name	*t* (yrs)	R	L	C	Shared C,L	C/R (%)	L/R (%)	C/L (%)	References
61. Sieperdaschor	13	39	21	19	18	49	54	90	1
44. Tollesbury	6	34	10	11	8	32	29	110	2
67. Karrendorfer Wiesen	5	27	18	13	13	48	67	72	3; 4
57. Kroons polders	3	39	26	25	20	64	67	96	5
58. Noord Friesland Buitendijks	1	39	20	21	18	54	51	105	6; 7
50. Breebaart	1	39	21	10	8	26	54	48	8; 9

References: 1. Koppejan (2000); 2. Reading et al. (2002); 3. Bernhardt and Koch (2003); 4. Zimmermann (2001); 5. van Gennip and Knotters (2002); 6. van Duin et al. (2003); 7. Hommel and Horsthuis (2002); 8. www.Groningerlandschap.nl/dollard/breebaartporjectvegetatie.htm; 9. Vreeken-Buijs (2002).

lapse of a seawall, species diversity of two restoration sites was still low compared to a reference marsh (17 and 16 species versus 36, respectively), and suggested this may be due to the large distance (80 km) between the nearest established marsh and the restoration sites. Nevertheless, a few restoration sites harbour more target species than the local source, hence these species will have travelled over longer distances.

5.3. Management

The management policy for most restoration sites is to abandon all human intervention after de-embankment and leave the site to develop naturally. However, it is questionable whether this policy would result in the most successful restoration, i.e., in this case highest number of target species.

5.3.1. Construction and maintenance of drainage structures

At the start of the restoration, artificial creeks may be required to improve drainage and increase colonisation rates (Eertman et al., 2002). Creeks will be especially important in sites where embankment has resulted in an over-consolidated soil surface acting as an aquaclude that impedes subsurface drainage (Crooks et al., 2002). Moreover, creeks may assist in supplying sediment to the salt-marsh surface (Reed et al., 1999) and differential sediment deposition patterns related to distance from creeks may positively influence (plant) species richness and distribution (Zedler and Callaway, 2001). In artificially drained sedimentation fields for example, plant growth may start 20 cm lower in elevation than on natural island marshes (Bakker et al., 2002). In some restoration sites therefore, meandering creeks are dug deliberately to enhance colonisation rates. In the Sieperdaschor (Nr. 61), a new creek, which was dug five years after de-embankment, resulted in enhanced tidal intrusion to the site's interior, coupled with higher sedimentation rates and more rapid colonisation of bare mud (Eertman et al., 2002). In order to accommodate the high tidal amplitude, this creek started to meander spontaneously. In most de-embanked sites, however, drainage occurs predominantly through existing ditches (Appendix B) and a dendritic creek network as found on many natural marshes may never develop (Verbeek and Storm, 2001), especially when the initial elevation of the site is high. Williams and Orr (2002), for example, concluded that on sites that were raised to mature marsh level prior to de-embankment, tidal drainage channels had not developed after 24–29 years. In contrast, dendritic channel systems developed spontaneously on subsided sites during the build up of intertidal mudflat (Williams and Orr, 2002). Another factor affecting creek development is the composition of the soil subsurface. In south-east England for example, low quantities of calcium carbonate in the soil in combination with the transition from marine to fresh water hydrology, have resulted in the formation of an aquaclude (i.e., a layer of over-consolidated material acting as a barrier to water movement) (Crooks et al., 2002). In such cases, the construction of artificial creeks may be required to enhance restoration success. Apart from the role of creeks, drainage is also affected by the size of the opening in the embankment. Boumans et al. (2002) for example reported enhanced salt-marsh vegetation development when culverts were enlarged by ca. 1 m in diameter. Lowering of the elevation at which the culverts were placed did not increase success.

5.3.2. Grazing or mowing regimes

Management may also be required to prevent successional processes from reducing species diversity over time. A comparison between restoration time and saturation index for 37 study sites showed that with the exception of a 197-year-old site, the highest saturation index was observed for the youngest sites, with a rapid decrease setting in after 15 years of restoration (Fig. 10). Highest species richness also occurred at around 15 years of restoration time in a comparison of six constructed restoration sites in New Hampshire (USA) (Morgan and Short, 2002). On the basis of their trajectory model, these authors

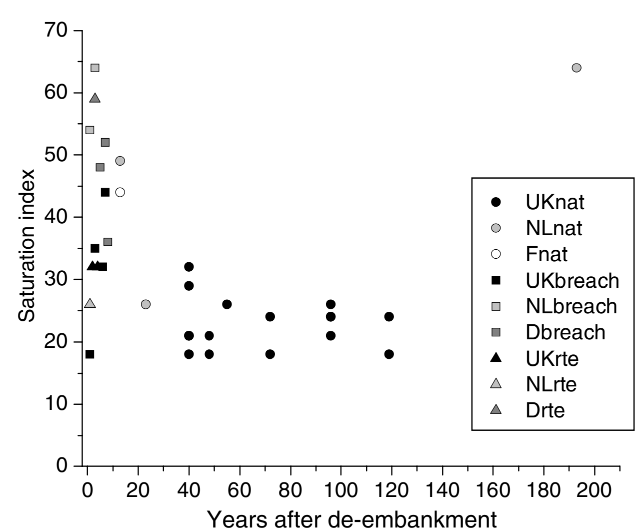

Fig. 10. Relationship between saturation index and restoration time. Nat, natural de-embankments; Breach, deliberate partial or complete removal of seawall; RTE, regulated tidal exchange by means of sluices or pipes.

suggest that the level of species diversity will be maintained over time. However, long-term experiments on barrier island and mainland marshes in Germany and the Netherlands, show that successional processes are likely to result in the dominance of a single or few tall growing species, such as *Atriplex portulacoides* at the low marsh and *Elytrigia atherica* at the high marsh (Andresen et al., 1990; Bos et al., 2002; Schröder et al., 2002; Bakker et al., 2003). Under brackish conditions, *Elytrigia repens* and *Phragmites australis* will expand if successional processes are not restrained (Dijkema, 1990; Esselink et al., 2000; Esselink et al., 2002). The dominance of these tall species results in the suppression or disappearance of species of shorter statue and loss of diversity. Species diversity may be maintained over time by the implementation of a grazing or mowing regime. In nine restoration sites grazing or mowing regimes have been implemented (seven of which have data on plant species abundances), and the effect of grazing has been studied in two of these. Results of one of these studies (Sieperdaschor, Nr. 61) show that nine years after de-embankment, the number of target species in lightly grazed plots is higher than in ungrazed plots (Bakker et al., 2002). At the cessation of grazing, the percentage cover of *Elytrigia atherica* and *Bolboschoenus maritimus* rapidly increased (Bakker et al., 2002) and in recent years, *Phragmites australis* has expanded rapidly in the ungrazed areas of this site (Eertman et al., 2002). *Elytrigia atherica* was also the dominant species in the ungrazed site Peazemerlannen (Nr. 59), 25 years after de-embankment (Bakker et al., 2002). According to Scherfose (1993), salt-marsh species that are likely to benefit from grazing are those with short-time strategies (i.e., annuals or biennials, low growing, early flowering, without storage organs and with stolons, rosettes or creeping shoots). This might then explain why some target species are rarely present in the restoration sites (Fig. 6), as the majority of sites are ungrazed. In fact, the only seven sites that have a grazing or mowing regime are also the top seven most successful sites on the bases of the saturation index.

Apart from the effect on the vegetation, grazing may also affect soil salinity and surface elevation change. This has been studied in one of the restoration sites (Noord Friesland Buitendijks, Nr. 58), where grazing resulted in up to two times higher soil salinity and a 40% lower rate of surface elevation change (van Duin et al., 2003). A negative influence of grazing on sedimentation rates was also reported by Andresen et al. (1990) and is most likely the result of reduced sward height and density of tillers (Esselink et al., 1998). Short turf with relatively high evapotranspiration was also suggested as the main cause for increased soil salinities in grazed plots (Bakker, 1985).

6. Recommendations for future restoration schemes

A major challenge in the restoration of salt marshes is to identify which factors are important in salt-marsh development. Past and future de-embankment schemes can contribute to this understanding, provided that key parameters and processes are being monitored. Paramount in future restoration cases is the need for clear targets in order to be able to evaluate restoration success. A possibility presented in this review is to identify target plant species from a regional species pool and to use species diversity within this restricted set of species to assess success. It should be realised however, that the presence of a species does provide information as to whether a salt-marsh community has formed. Therefore, collecting data on plant species abundance, preferably recorded in standard 2 m × 2 m permanent quadrats will be the next important step. In addition, species abundance should be recorded in transects covering the entire range from high to low elevation to allow the mapping of vegetation communities and their spatial distribution. Monitoring will not only allow the evaluation of success but also provide important feedback based on which management of the site can be adapted if necessary. A management option that most likely influences success is the construction of creeks to enhance tidal flooding of interior parts, increase sedimentation rates, improve drainage and enhance plant colonisation rates, species diversity and distribution ranges. Another option is the implementation of a grazing or mowing regime in order to create heterogeneity in the soil and vegetation and prevent dominance of a single species. A prerequisite for this type of management is that the sites are high enough for the establishment of vegetation communities suitable for grazing or mowing regimes. Experiments can be designed to test the effect of different management strategies or to study specific factors involved in salt-marsh development (e.g., seed dispersal, algae/invertebrate/plant relationships, nutrient availability etc.).

Acknowledgements

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Appendix A. Site number, name, location and sources of references. Underlined sites included in review (n=70), bold sites have information on target species in vegetation (n=37)

Site Nr.	Site name	Location	Country	Type[1]	Area (ha)	Breach[2]	Years embanked	Land use[3]	Main reason[4]	Number, type and dimension (in m) of breach (B), sluice (S) or pipe (P)	Drainage[5]	Elevation (m + sea level or ordnance datum)[6]	Tidal range (m)	SEC (mm/yr)[7]	Soil salinity	Management[8]	References
1a	**Abbots Hall A**	Blackwater estuary	UK	R	20	1996		A	2	2 P; 0.46 & 0.30 dia	III	-0.5-1.0 MHWN	4.7				1; 2; 3; 4; 5; 6;
1b	Abbots Hall B	Blackwater estuary	UK	B	115	2002		A	2	5 B			4.7				6
2	**Aldboro Point**	Colne estuary	UK	N	7	1921	25		1	6 B; 50	II	0.7-1.0 MHWN	4.5	7.22 a			7; 8
3	Alkborough	Humber estuary	UK	B	400	2004			2	B			6.4				6
4	Barkshore	Medway estuary	UK	N		1897	78		1	B							9
5	**Barrow Hill**	Colne estuary	UK	N	23	1953	154	P	1	3 B; 95	II,IV	0.8-1.4 MHWN	4.5	14 a			7; 8
6	Blackshore Mill	Blyth estuary	UK	N	<1	1953		A	1	B		-0.8 OD	2	13-16 e			10
7	Brancaster West	North Norfolk	UK	B	7.5	1996		F	3	B			6.5				6; 11
8a	**Brandy Hole A**	Crouch estuary	UK	N	51	1897	123	A	1	2 B; 135	II	0.4-0.8 MHWN	5	6.88 a			7; 8
8b	**Brandy Hole B**	Crouch estuary	UK	N	12	1897	123	A	1	2 B; 25	II	0.2-0.8 MHWN	5	5.21 a			7; 8
9a	Bridgemarsh Island A	Crouch estuary	UK	N	29	1928	114		1	1 B; 45			5				7; 8
9b	Bridgemarsh Island B	Crouch estuary	UK	N	69	1928	114		1	3 B; 105			5				7; 8
9c	Bridgemarsh Island C	Crouch estuary	UK	N	51	1928	114		1	2 B; 155			5				7; 8
10	Brue Pill	Weston Bay, Somerset	UK	B		1990			2	B			11.6				6
11	Bulcamp Marsh	Blyth estuary	UK	N	100	1945	141	A	1	B	II	-0.6 OD	2	-10 - 6.5 e			12
12	Burntwick	Medway estuary	UK	N	71.3	1897	27		1	B							9
13	**Canvey Point**	Thames estuary	UK	N	23	1874	34		1	entire; 1330	I	-0.5-0.1 MHWN	5.2	2.86 a			7; 8
14	Carnforth Marsh	Morecambe Bay	UK	R	51.2					5 S, 2 P			8.4				5
15	Chaldock Point	Chichester Harbour	UK	B		2000				B			4.4				4
16	**Clementsgreen Creek**	Crouch estuary	UK	N	4	1897	123	A	1	2 B; 45	II	0.4-0.6 MHWN	5	5.21 a			7; 8
17	Copperhouse	Medway estuary	UK	N	1.9	1897	34		1	2 B							9
18	Cotehele	River Tamar, Cornwall	UK	B	15	2002	152	A	2	B			4.2				6
19	**Ferry Lane**	Colne estuary	UK	N	6	1945	105	A,P	1	3 B; 50	IV	0.8-1.4 MHWN	4.5	11.25 a			7; 8
20a	**Fingringhoe Marsh A**	Colne estuary	UK	N	70	1897	57	P	1	4 B; 220	IV	0.8-1.2 MHWN	4.5	3.75 a			7; 8
20b	**Fingringhoe Marsh B**	Colne estuary	UK	N	8	1897	98	A	1	3 B; 35	II,IV	0.6-1.3 MHWN	4.5	4.58 a			7; 8
21a	Foulton Hall A	Hamford water	UK	N	66	1896	122	P	1	3 B; 125	II,III	0.6-0.8 MHWN	3.8	4.74 a			7; 8
21b	**Foulton Hall B**	Hamford water	UK	N	34	1921	147	P	1	1 B; 15	II,III	0.6-1.0 MHWN	3.8	9.17 a			7; 8
22	**Freiston**	The Wash	UK	B	78	2002	19	A	2	3 B	II,III	0.8-1.6 MHWN	6.4				47; 13
23	Gwent Levels	Bristol Channel N	UK	R	10.8					2 S; 1.05*1.05							5
24	Havergate Island	Deben estuary	UK	R	12	2000				1 S; 0.23			3.6				5; 4
25	**Hemley**	Deben estuary	UK	N	31	1953	155	P	1	1 B; 45			3.6	10 a			7; 8
26a	Horsey Island A	Hamford water	UK	N	5	1953	179	F	1	entire; 605	I	-0.2-0.7 MHWN	3.8				7; 8

(continued on next page)

Appendix A (*continued*)

Site Nr.	Site name	Location	Country	Type[1]	Area (ha)	Breach[2]	Years embanked	Land use[3]	Main reason[4]	Number, type and dimension (in m) of breach (B), sluice (S) or pipe (P)	Drainage[5]	Elevation (m + sea level or ordnance datum)[6]	Tidal range (m)	SEC (mm/yr)[7]	Soil salinity	Management[8]	References
26b	Horsey Island B	Hamford water	UK	R				A	2,3	1 S/P			3.8				14; 3
27	Humber	Humber estuary	UK	B	1000	Plan		A,I	2,3	various							48
28	Milfordhope	Medway estuary	UK	N		1908	10		1	B							9
29a	**North Fambridge A**	Crouch estuary	UK	N	27	1897	123	A	1	4 B; 155	II	0.4-0.6 MHWN	5	4.17 a			7; 8
29b	**North Fambridge B**	Crouch estuary	UK	N	43	1897	123	A	1	2 B; 105	II	0.2-0.9 MHWN	5	6.25 a			7; 8
30a	Northey Island A	Blackwater estuary	UK	N	79	1897	123	A,P	1	6 B; 870	II,IV	-0.1-0.4 MHWN	4.8	7.08 a			7; 8
30b	**Northey Island B**	Blackwater estuary	UK	B	0.8	1991	118	A,F	3	1 B; 20	I	0.7-1.6 MHWN	4.8	24.8 b			15; 3; 14
31	Oakham	Medway estuary	UK	N		1897	78		1	B							9
32	**Orplands**	Blackwater estuary	UK	B	40	1995	175	A,F	2	2 B; 50 & 40	III	0-2.5 MHWN	4.7	10 e			16; 1; 2; 3; 14
33	Pawlett Hams	River Parret, Somerset	UK	B	4.8	1994			3	B		2.7-3.7 MHWN	11.1				14
34	Porlock Marsh	Somerset	UK	N	101	1996		A,P	1	B			11.1				17; 4
35	Ryans Field	Hayle, Cornwall	UK	R	6					S			5.8				5
36	**Saltram**	River Plym, Devon	UK	R	5	1995	145	P	2	1 B (to 2.4m); 13m, 5 P; 0.6m	I	-0.1-0.9 MHWN	4.7				18; 19; 1; 3
37	**Sampson's Creek**	Blackwater estuary	UK	N	4	1945	105		1	1 B; 20	IV	0.2-1.2 MHWN	4.7	7.92 a			7; 8
38	Seal Sands	Tees estuary	UK	R	9	1993	19	Po	2	1 P; 1.05	III		4.6				20; 4; 5
39	**Skipper's Island**	Hamford water	UK	N	37	1953	113	P	1	2 B; 70	I	0.5-1.5 MHWN	3.8	13 a			7; 8
40	**Stone Marsh**	Hamford water	UK	N	30	1874	34	P	1	1 B; 20	IV	0.2-0.6 MHWN	3.8	4.37 a			7; 8
41	Thorngumbald / Paul's Strays	Humber estuary	UK	B	70	2003		A,P	2,3	2 B			6.4				6
42	Thornham Bay	Chichester Harbour	UK	B	6.9	1996		W		B	III		4.4				14; 4
43	Titchwell	North Norfolk	UK	R	36					? P; 0.6			6.5				5
44	**Tollesbury**	Blackwater estuary	UK	B	21	1995	150	A	3	1 B; 50	I	-0.6-1.5 MHWN	4.7	23 c			21; 19; 4; 3; 14; 8; 6
45	Trimley	Orwell estuary	UK	B	16	2001				B			3.6				4
46a	**Wallasea A**	Crouch estuary	UK	N	2	1953	179	A	1	3 B; 160	I	-0.2-0.4 MHWN	5				7; 8
46b	**Wallasea B**	Crouch estuary	UK	N	2	1953	179	A	1	2 B; 90	I	-1.0-0.6 MHWN	5				7; 8
46c	Wallasea C	Crouch estuary	UK	B	110	2005		A	2	B							49
47	**Walton Central**	Hamford water	UK	N	73	1938	64		1	1 B; 130	IV	0.5-0.9 MHWN	3.8	8 a			7; 8
48	**Woodbridge**	Deben estuary	UK	N	15	1953	155	A	1	1 B: 35			3.6				7; 8
49	Boonepolder/ O Zwakepolder	Westerschelde	NL	B	80	*1995			2	NA							22
50	**Breebaart**	Dollard estuary	NL	R	63	2000	21	A,P	2	1 S; 1*2	II,III		3				23
51	Dordtse Biesbosch	Haringvliet	NL	B					2	B							22
52	Everingepolder	Westerschelde	NL	B	235	*1995			2	NA							22
53	Groene Strand	Terschelling	NL	B	23	1996		D	2	S						G	23; 24

Site Nr.	Site name	Location	Country	Type[1]	Area (ha)	Breach[2]	Years embanked	Land use[3]	Main reason[4]	Number, type and dimension (in m) of breach (B), sluice (S) or pipe (P)	Drainage[5]	Elevation (m + sea level or ordnance datum)[6]	Tidal range (m)	SEC (mm/yr)[7]	Soil salinity	Management[8]	References
54	Hedwigepolder	Westerschelde	NL	B	320	*1995			2	NA							22
55	Hellegatpolder	Westerschelde	NL	B	125	*1995			2	NA							22
56	Holwerder summerpolder	Friesland	NL	R/B	37	1989/95	33	P	2	3 S; 1*2 / 1 B; 12	II,III	1.1 MSL	2	6 e			8; 24
57	**Kroons polders**	Vlieland	NL	B	85	1996	76	D	2	1 B; 10		0.82-1.26 OD		6.2-12.6 c		M	23; 25; 52; 53
58	**Noard Fryslân Bûtendyks**	Friesland	NL	B	135	2001	91	P	2	3 B; 60	II,III	0.4-0.9 MHT	2	7.78-19.1 c	10-70% of ref	G	26; 27; 23; 28
59	**Paezemerlannen**	Friesland	NL	N	100	1973/79	40	P	1	2 B; 500	II,IV	1.4 MSL	2.2	16.1 d			29; 8; 23; 30
60	Schelde	Schelde estuary	NL	R	10	2001		A,P	4	S							31
61	**Sieperdaschor**	Westerschelde	NL	N	100	1990	24	A,P	1	1 B; 15, 1 S; 1.5*1.2	II,III	2.6 MSL	5	5-30 e		G	32; 33; 8; 34; 30
62	**Verdronken Zwarte polder**	Zeeuws Vlaanderen	NL	N	43	1802	2	D	1	B						G/M	35; 36
63	Zwarte polder	Westerschelde	NL	B	65	*1995			2	NA							22
78	Ketenisse polder	Schelde estuary	B	B	30	2002		P	2	entire	I						37
64	Zeeschelde	Schelde estuary	B	R	300	plan		P	2,3	S							31
65	**Berensch (Spieka-Neufeld)**	Niedersachsen	D	R	280	1995	125	P	2	1 S; 1.3	II,III	0-1.0 MHT	3.2		6-15‰	G	38; 39
66	**Hauener Hooge**	Leybucht	D	B	80	1994	60	P	2,3	1 B; 100	II,III	0.1-0.6 MHT	2.8	20-87 e			40; 50
67	**Karrendorfer Wiesen**	Mecklenburg-Vorpommern	D	B	350	1993	83	A,P	2	entire; 5000	I	-0.7-3.0 MSL	0.02			G	41; 42
68	Langeoog	Niedersachsen	D	B	240	Plan	156	P	2	entire; 5500	II,III	0.2-0.5 MHWN	2.6				40
69	Luetetsburg	Niedersachsen	D	B	50	Plan	48		2	B		1.5-2.2 OD					43; 39
70	Munster polder	Niedersachsen	D		92	plan		A,I	2	undecided		1.4-2.1 OD					40
71	Neuensien polder	Rugen	D	B	80	2001				B							41
72	Preetz polder	Rugen	D	B	180	1995				B							41
74	Sundische Wiese	Mecklenburg-Vorpommern	D	B	943	2009	49	P	2	? B			0.02			G	44
75	Zickerniss-Niederung	Rugen	D	B	225	1995				B							41
76	**Ziesetal**	Mecklenburg-Vorpommern	D	B	162	1995/99	110	P	2	B / entire	II		0.02		5-13‰	G	45; 46
77a	**Baie des Veys A**	Normandie	F	N/R	30.2	1989	119	P	1,2	1 S; 1*1	II,III	1.9-2.5 OD	6				51
77b	Baie des Veys B	Normandie	F	R	900	*1995			2	NA							8
	Sites not known from start:																
	Pillmouth	River Torridge, Devon	UK	B		2000	>200	A	3	3 B	II		7		25150 µS		54

1 N = natural breach; B = deliberate breach; R = regulated tidal exchange.
2 * = plan for breach not carried out.
3 A = arable; P = pasture; F = freshwater grazing marsh; D = dune valley or beach plain;I = industry or commerce; W = waste ground; Po= pool.
4 1 = accidental; 2 = habitat creation/compensation;3 = flood defence;4 = gaining experience.
5 I= superficial;II= drainage ditches; III = artificial creeks;IV= natural creeks.
6 MHWN= mean high water neap tide; MHW = mean high water; MSL = mean sea level; OD= ordnance datum.
7 SEC = surface elevation change. Methods used: a = depth to agricultural layer; b = sedimentation plate ;c = sedimentation-erosion bar; d = repeated leveling; e = unknown.
8 G = grazed; M = mown.

Appendix B

Salt-marsh communities used for selecting regional target species for (1) the Central North-Atlantic (Schaminée et al., 1998) (2) the Southern North-Atlantic (Rodwell, 2000) and (3) the German Baltic (Krisch, 1990) regions

	NVC code		
Syntaxon	**1**	**2**	**3**
Spartinion			
Spartinetum maritimae	24Aa1		
Spartinetum townsendii	24Aa2	SM6	
Thero-Salicornion			
Salicornietum dolychostachya	25Aa1		
Salicornietum brachystachya	25Aa2	SM8	*Salicornia europaea* group
Suaedetum maritimae	25Aa3	SM9	
Puccinellion maritimae			
Puccinellietum maritimae	26Aa1	SM10,11,13*	*Puccinellia maritima* group
Plantagini Limonietum	26Aa2		
Halimionetum portulacoidis	26Aa3	SM8,14,26	
Puccinellio-Spergularion salinae			
Puccinelietum distansis	26Ab1	SM23	*Spergularia salina* group
Puccinellietum fasciculatae	26Ab2		
Puccinellietum capillaris	26Ab3		
Parapholido strigosae-Hordeetum marini	26Ab4		
Armerion maritimae			
Juncetum gerardi	26Ac1	SM16*	*Juncus gerardi* group
Armerio-Festucetum littoralis	26Ac2		
Junco-Caricetum extensae	26Ac3		
Blysmetum rufi	26Ac4	SM19*	*Blysmus rufus* group
Artemisietum maritimae	26Ac5	SM17	
Atriplici-Elytrigietum pungentis	26Ac6	SM24,25,26	
Oenanthe lachenalii-Juncetum maritimi	26Ac7	SM15,18	*Oenanthe lachenalii* group
Elymus repens community		SM28	*Agrostis stolonifera* group
Saginion maritimae			
Sagino maritimae-Cochlearietum danicae	27Aa1		
Phragmition australis			
Eleocharitetum uniglumis		(SM20)*	*Eleocharis uniglumis* group
Scirpetum tabernaemontani/maritimi	8bb2, 8bb3d	S20	*Aster tripolium* group
Mediterranean			
Arthrocnemum perenne stands		SM7	
Suaeda vera-Limonium binervosum		SM21	
Limonio vulgaris-Frankenietum laevis		SM22	

* Target species of the following communities: SM13e, SM16f, SM19 and SM20, were not included for the Southern North-Atlantic region as they occurred only in the Northern part of Great-Britain.

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