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# Restoration of intertidal habitats by the managed realignment of coastal defences, UK

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## Abstract

In the United Kingdom, coastal defence has been the primary driver of intertidal habitat restoration. Rising sea levels, coupled with the high cost of maintaining coastal defences, have led coastal managers to look for more cost effective and sustainable methods of coastal protection. Managed realignment, the landward retreat of coastal defences and subsequent tidal inundation of formally reclaimed land, has since the early 1990's, been increasingly used to fulfil these requirements.

Results from several managed realignment schemes have shown that with fairly minimal pre-treatment and management, by allowing tidal ingress through a simple relatively small breach, the landward realignment of coastal defences will quickly produce intertidal mudflats on low-lying agricultural land which are colonised by invertebrates and, given the appropriate elevation, saltmarsh plants. What is unclear, however, is the time scale needed to produce intertidal habitats that are equivalent to reference conditions or if equivalency can indeed ever be reached.

To date, most research effort has concentrated on monitoring the biological and physical development of realignment sites. This paper presents the results of a survey of saltmarshes that have developed on formally reclaimed land as a result an accidental breach in the embankment. These marshes are used as analogues for managed realignment and may give an indication of the future trajectories of current saltmarsh creation efforts.

Keywords: Managed realignment; Saltmarsh; Restoration; Chronosequence.

## Introduction

Replacing coastal habitats where they are eroded, inundated or otherwise impacted upon is particularly important given the high level of ecosystem service they provide. Saltmarsh creeks provide spawning and nursery areas for many fish species and the vegetation provides roosting, nesting and feeding sites for birds. In addition to the specialist flora and fauna directly associated with tidal saltmarshes they are areas of high productivity providing a source of organic matter and nutrients for adjacent marine habitats. Their biodiversity and functional value is recognized in law under the European Union Habitats Directive (CEG, 1992). The directive seeks to maintain 'no-net-loss' in total habitat area. The UK's Biodiversity Action Plan (BAP) commits the Government to develop strategies to conserve and, where possible, enhance biodiversity (UK Biodiversity group, 1999). Managed realignment (the setting back of coastal defences inland) is viewed as an important and viable technique in meeting BAP objectives for the creation of intertidal habitats.

In addition to their high biodiversity value it is widely accepted that coastal wetlands, and saltmarshes in particular, play an important part in ameliorating the effect of wave action on coastal defences (Moller *et al.*, 2001; Toft and Maddrell, 1995; Pethick, 1992). Moller *et al.* (1999) showed that wave attenuation over saltmarsh was 50% higher than over sand flat, even under similar water depths. As saltmarsh width decreases an almost linear increase in the height of the sea wall is necessitated to offer comparable protection, adding considerably to capital wall building and maintenance costs (Dixon *et al*., 1998; King and Lester, 1995). By setting back coastal defences and creating saltmarsh in the intervening area considerable savings could be made.

Since the early 1990's the managed realignment of coastal defences is increasingly being used in the United Kingdom as a cost effective and sustainable response to biodiversity loss and flood management. Results from several managed realignment schemes have shown that with fairly minimal pre-treatment and management by allowing tidal ingress through a simple, relatively small breach the landward realignment of coastal defences will quickly produce intertidal mudflats on low-lying agricultural land which are colonised by saltmarsh plants and invertebrates. In addition it has been shown that the tidal curve within the de-embanked area quickly reflects that of the adjacent marshes and wave attenuation is considerably reduced (Rawson *et al*., 2004), (although this is not reported for the majority of sites). What is unclear however is how long it will take for saltmarsh vegetation, representative of semi-natural communities to develop, if at all.

## Sites of historic sea defence failure

The UK has a long history of saltmarsh reclamation for agricultural use. Marshes were embanked, drained and used initially for grazing livestock and in many cases, when salinities had reduced, ploughed for the production of crops. The response of the fronting intertidal areas was to adapt to the change in shore profile with the seaward extension of the saltmarsh, which, over time would again be at an appropriate elevation for reclamation. This process was piecemeal, with no specific design criteria, and the speed of reclamation often reflected the economic climate of the period. Conversely, at certain periods through time large storm events led to breaches in embankments, some of which were not repaired, usually because it was uneconomical to do so. These historic breach sites provide an analogue for modern day managed realignment and present us with a chronosequence of saltmarsh development on formally enclosed land.

In the late 1980's, as the concept of managed realignment was evolving, Burd (1992; 1994) studied these historic breach sites in an attempt to determine the physical variables which may have determined the fate of the de-embanked marshes when they were re-flooded, together with the observed characteristics at the time of the survey. Results from the project were used to inform on the construction of sites for saltmarsh creation by managed realignment. Burd (1994) identified 23 historic breach sites in the English county of Essex that had been fully enclosed and then subsequently breached. The majority of sites were originally enclosed before 1774, with further sites being added in the period to 1840 and only 4 enclosed after 1840. Although many of the sites were breached a number of times in their history the storms of 1897 caused the final breach in 10 sites with three further sites being breached and repaired at this time. The 1921 storms accounted for several more of the final breaches, and the 1953 floods caused the most present round of permanent loss to the sea.

## A chronosequence survey of saltmarsh vegetation

Of the 23 historic breach sites identified by Burd (1994) in Essex, 20 were visited along with four managed realignment sites, in a survey to determine how closely the vegetation of those sites resembled that of adjacent semi-natural saltmarshes. The survey was carried out during the summer of 2004. Sites were rejected that did not have directly adjacent saltmarsh. In two cases the sites no longer contained any saltmarsh at all. This gave a total of 18 sites with adjacent reference saltmarsh over four estuaries or embayments (Hamford Water, the Colne, the Blackwater and the Crouch), including the managed realignment sites. This gave a space for time view of saltmarsh development from 2 to 107 years old (as of 2004).

Five 2x2m quadrats were located along a 100m transect at equivalent elevations within the de-embanked sites and adjacent saltmarsh. Transects were stratified into five 20m lengths, with one quadrat placed at random within each 20m length. This method ensured good coverage along the 100m transect but avoided any periodicity that might be found in the vegetation (*e.g.* ridges and runnels that reflected old agricultural systems). Elevation was used as a surrogate for tidal inundation to ensure that at each site vegetation communities on both the reference saltmarsh and de-embanked site received equivalent tidal inundation frequencies. Checks were made at several sites by watching the incoming tide to ensure this was in fact the case. Species presence and an estimate of percentage cover were recorded for each quadrat. Mean percentage covers for species recorded within each transect were calculated. The difference in percentage cover between the de-embanked sites and the reference marshes was calculated as "*historic –reference"*, where negative values indicate lower percentage covers in the de-embanked sites and positive values, a greater cover (Table I).

*Atriplex portulacoides* occurred at significantly lower mean percentage covers (-12.9%) in the de-embanked sites than the reference marshes along with *Spergularia media* and *Triglochin maritima*. Conversely, *Spartina anglica* occurred at significantly higher percentage covers (+10.6%) within the de-embanked sites than the reference marshes.

Table I. Mean percentage cover (s.e.) for species recorded in quadrats within the historic sites (and managed realignment sites) and adjacent reference marshes

| | Reference marsh | s.e. | Historic/ MR sites | s.e. | Hist/MR – Ref | s.e. | p_diff |
|---|---|---|---|---|---|---|---|
| *Armeria maritima* | 0.0 | (0.0) | 0.0 | (0.0) | 0.0 | (0.0) | 0.22 |
| *Aster tripolium* | 8.8 | (3.0) | 9.6 | (2.7) | 0.8 | (1.9) | 0.67 |
| *Atriplex portulacoides* | 28.5 | (4.1) | 15.6 | (4.7) | -12.9 | (4.6) | 0.01 |
| *Atriplex prostrata* | 0.0 | (0.0) | 0.0 | (0.0) | 0.0 | (0.0) | 0.16 |
| *Bostrychia scorpioides* | 1.8 | (0.6) | 1.8 | (1.1) | 0.0 | (1.2) | 0.98 |
| *Cochlearia anglica* | 0.2 | (0.1) | 0.1 | (0.1) | -0.1 | (0.1) | 0.44 |
| *Limonium vulgare* | 10.3 | (3.0) | 6.2 | (2.6) | -4.1 | (4.2) | 0.33 |
| *Plantago maritima* | 0.3 | (0.2) | 0.0 | (0.0) | -0.3 | (0.2) | 0.13 |
| *Puccinellia maritima* | 29.4 | (3.9) | 31.6 | (5.7) | 2.1 | (4.8) | 0.66 |
| *Salicornia agg.* | 7.2 | (2.2) | 11.6 | (3.3) | 4.3 | (2.5) | 0.10 |
| *Sarcocornia perennis* | 0.8 | (0.2) | 0.4 | (0.1) | -0.5 | (0.2) | 0.06 |
| *Spartina anglica* | 2.8 | (1.5) | 13.4 | (4.6) | 10.6 | (3.6) | 0.01 |
| *Spartina maritima* | 0.01 | (0.0) | 0.0 | (0.0) | 0.0 | (0.0) | 0.40 |
| *Spergularia marina* | 0.01 | (0.0) | 0.0 | (0.0) | 0.0 | (0.0) | 0.33 |
| *Spergularia media* | 1.0 | (0.3) | 0.5 | (0.2) | -0.5 | (0.2) | 0.04 |
| *Suaeda maritima* | 1.9 | (0.5) | 3.7 | (1.2) | 1.8 | (1.2) | 0.15 |
| *Triglochin maritima* | 6.2 | (1.7) | 0.7 | (0.6) | -5.5 | (1.8) | 0.01 |
| Algal | 0.5 | (0.5) | 0.4 | (0.4) | -0.1 | (0.1) | 0.33 |
| Bare mud | 6.6 | (2.1) | 8.8 | (2.0) | 2.2 | (2.2) | 0.34 |
| Water | 0.2 | (0.1) | 0.0 | (0.0) | -0.2 | (0.1) | 0.19 |

The successful establishment and spread of *Spartina* throughout the UK during the twentieth century was largely due to the species perennial life history and the existence of a zone of mud flat formally unoccupied by saltmarsh plants – a vacant niche (Gray *et al.*, 1990). The sparsely vegetated mudflats in the early phases of saltmarsh development would provide ideal conditions for the invasion of *Spartina* within the de-embanked sites, at least by the time of the 1953 floods by which time the species was wide spread.

Rauss (2003) describes the invasion of *Spartina* in the Bay de Veys in northern France from when it was first recorded in 1906 (also France's first record) through an invasive stage to becoming the dominant species by 1963. Rauss (2003) reports the present situation where *Spartina* is now confined to the pioneer zone and the saltmarshes of the Bay de Veys are again dominated by typical *Atriplex portulacoides/Puccinellia maritima* communities. This transition from invasion through stabilisation and regressive phases, where *Spartina* is now a stable component of the community, comes as a result of interspecific competition between saltmarsh plants. It remains to be seen whether the saltmarshes that have developed within the de-embanked sites in the UK can follow the same trajectory over time or if a lack of adequate drainage or sediment supply produces relatively static vegetation.

The lower mean percentage cover of *Atriplex portulacoides* may be a reflection of a higher water content in the soils of the de-embanked marshes, of which *Spartina* is more tolerant. There is a significant inverse relationship between water saturation in the root zone and abundance of *Atriplex portulacoides* (Crooks *et al*., 2002). Watts *et al*. (2003) found that newly accreted marine sediments at the Tollesbury realignment site on the Blackwater Estuary in Essex were characterised by high water content, low bulk density and low resistance to erosion. This may be due, in part, to the formation in reclaimed agricultural soils of an over consolidated horizon with low hydraulic conductivity, forming an aquaclude or barrier to water that restricts sub-surface drainage within the developing marsh sediments (Crooks, 1999).

Figure 1 shows the difference in cover values for *Suaeda maritima* between de-embanked (historic plus managed realignment sites) and reference marshes over time. As would be expected for a pioneer species cover values are highest in the de-embanked sites initially and over time, the cover falls. Conversely, the perennial species *Limonium vulgare* has considerably lower cover values in the early years in the de-embanked sites and shows as increase in cover over time (Figure 2).

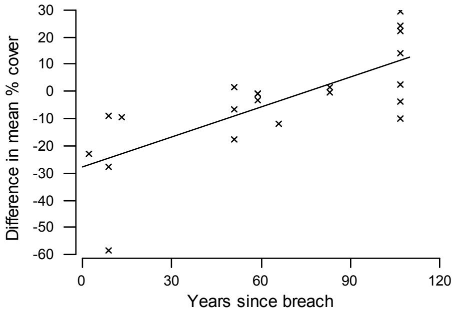


*Fig. 1. Difference in percentage cover (historic – reference) over time for Suaeda maritima, with fitted linear regression (p=<0.05).*

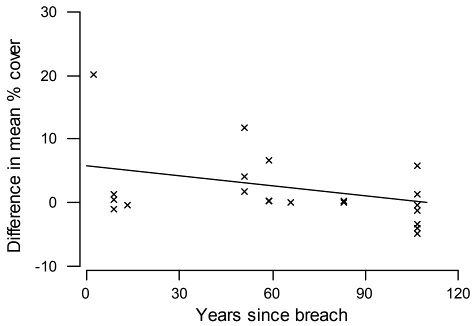


*Fig. 2. Difference in percentage cover (historic – reference) over time for Limonium vulgare, with fitted linear regression (p=<0.001).*

In addition to differences in mean percentage cover between de-embanked and reference marshes, there were also differences in the number of species recorded with the de-embanked sites showing lower species richness than their adjacent reference marshes

(Fig. 3). The mean number of species recorded in the reference marshes was 10.1 with a mean of 7.9 species recorded in the de-embanked sites. It is estimated that the fitted linear regression line will pass through 0 at 126 years. That is to say it will take an average of 126 years for species richness within the de-embanked sites to accumulate an equivalent number of species to that of their adjacent reference marshes (it may not be the same species however).

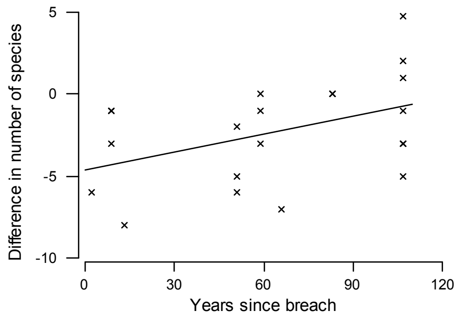


*Fig. 3. Difference in species richness (historic – reference) over time, with fitted linear regression, p=<0.05. spp_hist - ref = - 4.57 + 0.0362 years since breach.*

## Choosing the right reference to measure restoration success

There has been considerable debate on how to define success in tidal wetland creation (Zedler, 2001). The choice of reference to measure the success of a scheme will strongly affect the outcome.

In the county of Essex, saltmarsh losses have been rapid with 974ha being lost between 1973 and 1998 (Cooper *et al*., 2001), mainly caused by human activities and a continuous rise of high and extreme water levels (Van der Wal and Pye, 2004). By 1998 there were only 2878ha of saltmarsh remaining. The majority of marshes are cliffed on the seaward edge and backed by sea walls giving a truncated saltmarsh profile. This is reflected in the limited number of mid and upper marsh species recorded in the chronosequence survey (Table I). The truncated nature of the Essex saltmarshes may make it difficult to identify reference conditions by which success can be measured, particularly in situations where the elevation of de-embanked sites are outside the range of those observed in the reference sites. Wolters *et al*. (in press) have suggested a diversity score to measure success, where plant species present within the restoration site are given as a proportion of those present in a regional target species pool. In this way success can be measured as a continuum towards the goal of 100% fit with the target species pool in addition to allowing comparisons between sites. At present many of the schemes in the UK are unlikely to reach a complete fit, due to their small size and lack of elevational range. Measures of success must be realistic and adapted to take into account the physical characteristics of individual sites.

Even though there may be problems using existing saltmarshes as a bench mark for reference conditions, it may still be the most appropriate template to measure the development of a site in the absence of other quantifiable data. Reference conditions must however, be chosen to reflect the variation inherent in natural processes. Caution should be taken not to use 'one-off' surveys to identify target conditions; rather a long term and wide scale approach to monitoring should be used. Monitoring of saltmarshes adjacent to the Tollesbury managed realignment site, in Essex has shown that there has been a large change in the vegetation composition. Between 1994 and 2001 there was a 25% loss of *Atriplex portulacoides* leading to a complete change in vegetation community classification in 28% of permanent plots. Using the plant data collected from 1994 or 2001 as a baseline would have very different outcomes by which the success of a scheme could be measured against. It is therefore important that methodologies used to identify reference conditions are robust enough to reflect the variation in hydrodynamics, ecology and geomorphology found in the natural environment.

The contemporary view of restoration ecology emphasises process and function and the modern paradigm is the restoration not of a species assemblage or community but of a functioning ecosystem which can evolve and change. Measures of success should take account of this and include, for example, nutrient exchanges between created marsh and the adjacent estuary, the ability to adapt to disturbance (natural and/or anthropogenic) and the contribution of a scheme to overall estuarine processes.

## Implications for saltmarsh creation

Experience over the last decade has shown that, where elevations are suitable, managed realignment can quickly produce intertidal flats that are colonised by saltmarsh vegetation. Realignment sites low in the tidal frame have developed large stands of *Salicornia* spp., rare in the south east of England where saltmarshes are predominantly cliffed, excluding pioneer communities. Nationally scarce plants such as *Inula crithmoides*, *Suaeda vera* and *Spartina maritima* have all been recorded within realignment sites and transitional species such as *Trifolium squamosum* and *Bupleurum tenuissimum* (also nationally scarce) have been recorded growing on the now abandoned embankments. Where sites are at present too low for the development of saltmarsh, mudflats have been colonised by intertidal invertebrates providing additional feeding areas for wading birds (Atkinson *et al.,* 2004). Fish have also been recorded using realignment sites for feeding and refuge.

The results of the chronosequence survey should not be interpreted as the inevitable failure of saltmarsh creation schemes in matching reference conditions, rather a cautionary note that habitat creation efforts rarely create an exact replica of a semi-natural system. Neither should saltmarsh developed on managed realignment sites be seen as a like-for-like replacement for losses elsewhere. Emphasis should continue to be placed on the protection and sustainable management of existing marshes.

Created saltmarshes should ideally function within the normal variation found in semi-natural marshes and retain key features (Atkinson, 2004). It is essential that pre-breach monitoring of existing habitats, at a local or regional scale, is part of any intertidal habitat restoration scheme to take into account variation so that specific and realistic success criteria can be prescribed. Post breach monitoring should be designed to measure such success criteria. The survey of the historic breach sites has shown that differences between existing saltmarshes and those developed on previously agricultural land can exist for long periods. It is therefore difficult to assign time scales for saltmarsh restoration. Success should be measured on a continuum towards a desired goal. In this way, whilst the achievement of the goal may be beyond the life span of most projects, the trajectory of a site towards that goal may be measured.

A longer term, geographically wider, more flexible approach to coastal zone management has been developed in the UK whereby both intertidal habitats and coastal defence can both be accommodated. As Morris *et al*. (2004) emphasise, the benefits of managed realignment schemes should not be looked at in isolation but in the broader context of coastal management. Ultimately the success of any managed realignment should be measured in terms of the contribution the scheme makes to coastal processes at the landscape scale.

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