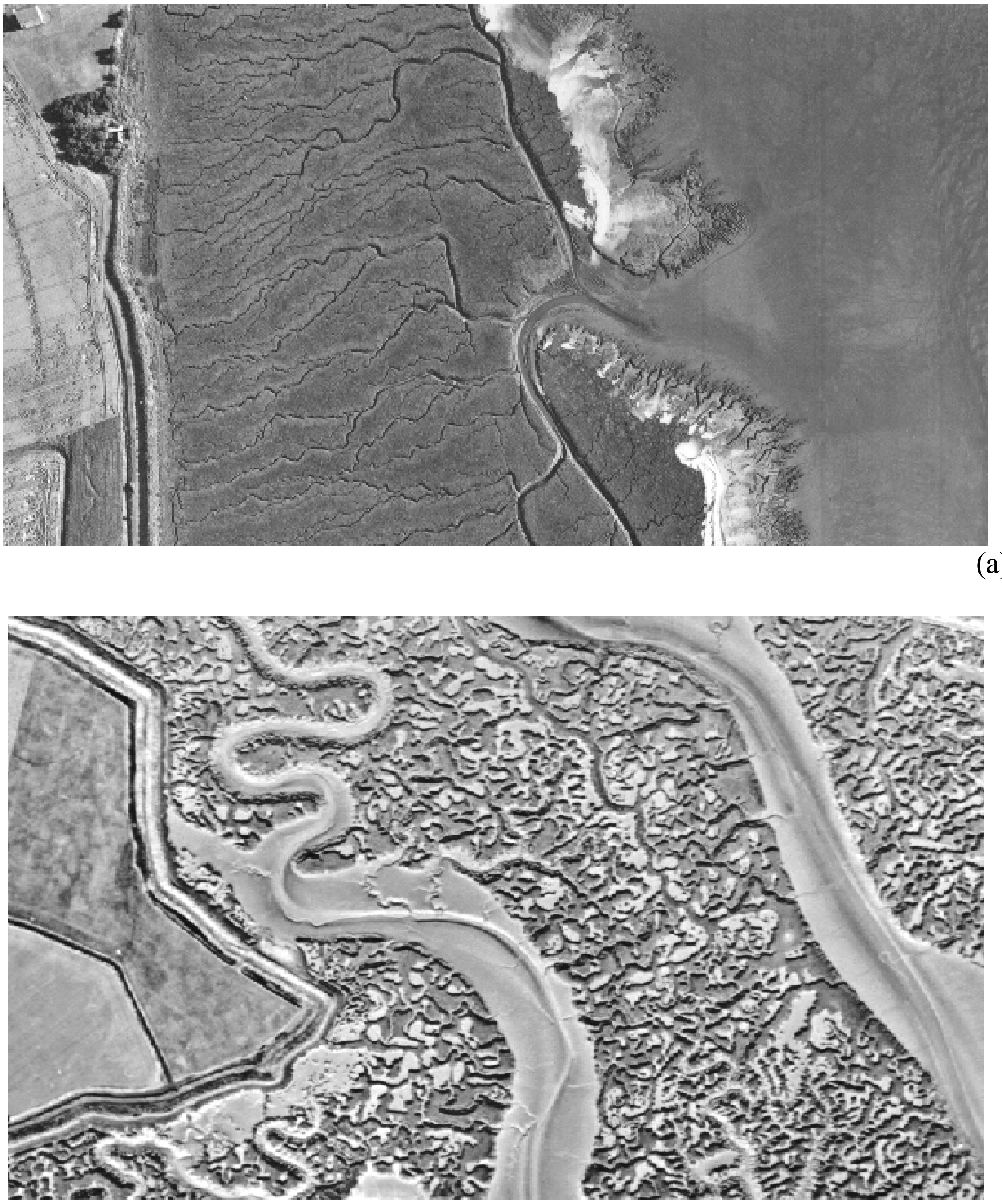
agricultural land use. At these sites, such as North and South Fambridge, Bridgmarsh Island and parts of Northey Island, where the sediment thickness are greater, a complex system of anastomising (connecting) creeks are superimposing upon the rectilinear network (Figure 3.4). Natural marshes in the Severn Estuary are characterised by relatively very low creek densities and the formation of stepped series of cliffed margins forming terrace sequences of marshes of varying ages (Figure 3.1). Natural unmanaged-retreat sites within this region, some of which may date back to the 16th century (Allen 2000), possess this regionally-typical 'tabular', very low density creek network system morphology (Figure 3.5).

A study by Crooks and Pye (2000) found that, while all the marshes in these south-west and south-east regions consisted of similar muds, there were considerable differences in terms of the density and other related geotechnical properties of the sediments. In the Severn Estuary, the marshes terraces were found to contain well drained, high density and high (undrained shear) strength sediments that were very resistant to erosive forces. Over in south-east England, by contrast, marshes of similar age were found to be very poorly drained with low density and low strength sediments. These sediments were thus relatively less resistant to erosion by waves and the local tidal system. What is more, on sampled naturally-restored marsh sites the underlying former land-claimed marsh was found to be heavily consolidated and to form an aquaclude (ie a barrier to water), thus increasing the drainage problems within the developing marsh. These naturally regenerated marshes possessed the lowest of all densities and strengths and it would be expected that they would be more susceptible to erosion. Subsequently, a recent study by Crooks *et al* (in press) on the natural marsh at St Peters Flat on the Dengie Peninsula found soil drainage to effect the species composition of the vegetation, acting against species intolerant of poor drainage (eg *Halimione portulacoides*) and favouring those more tolerant of moisture and anaerobic soil conditions (eg *Puccinellia maritima*). It may therefore be the case that the poor drainage on restored marshes in this region has an influence, not only on sediment erodibility and extent of bare mudflat but also upon the saltmarsh vegetation communities.



(a)

(b)

**Figure 3-3** Range of natural saltmarsh morphologies found in south-east England

Open coast marshes, such as that at St Peters Flat on the Dengie peninsula (a) are typified by more erosion resistant sediments and lower creek densities that inner estuary marshes, such as Old Hall/Tollesbury marsh (b). In terms of restoration the location of a site within a landscape will therefore influence the form of the marsh which will develop. How these different creek structures influence the ecological functioning (fisheries support, nutrient recycling etc.) of each marsh is not known. Photographs courtesy of the Environment Agency (Dimensions of each image is 1400 m by 800 m. North to top.)

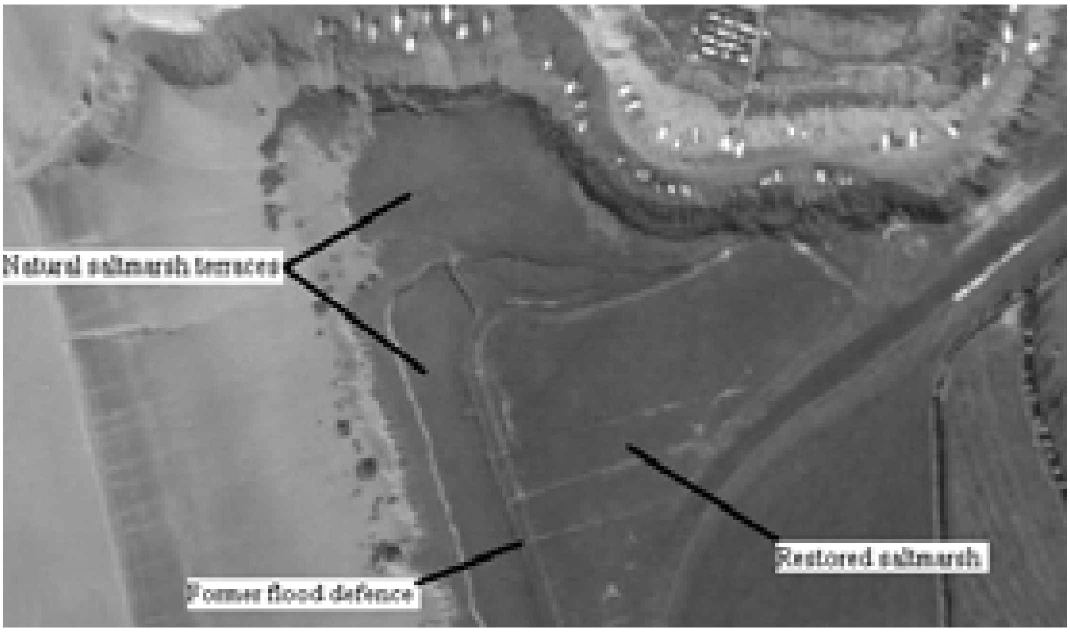
(a)

(b)

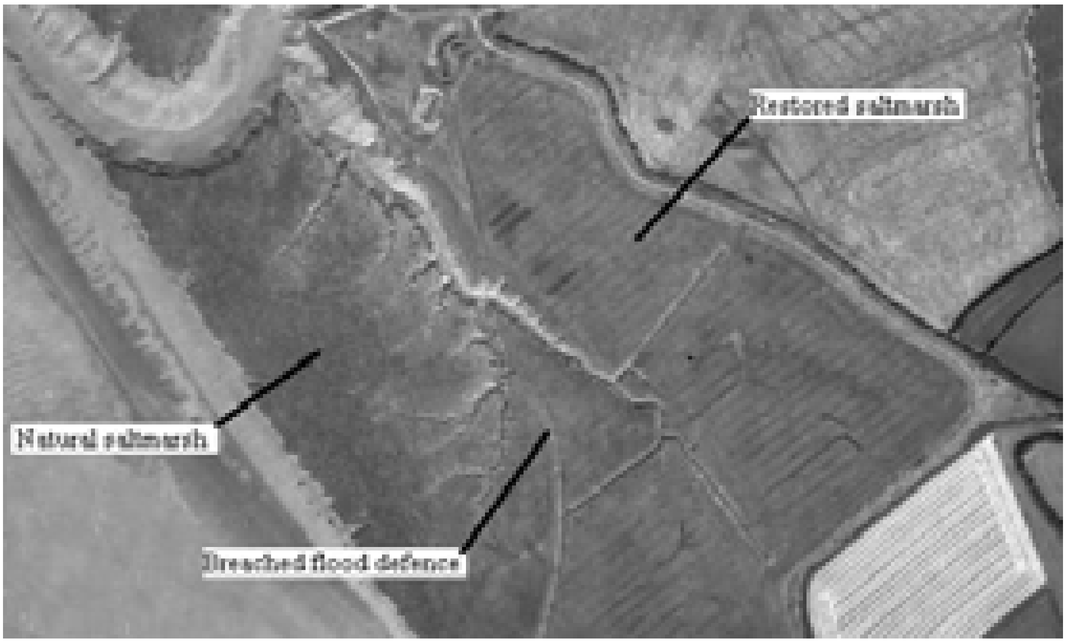
**Figure 3-4** North Fambridge (a) and Northey Island (b) saltmarshes naturally regenerated during the storm surge of 1897

These sites provide an example of how breach retreat, estuarine fringe saltmarshes in Essex in south-east England might develop if no management was undertaken. At North Fambridge the creek network of the regenerated saltmarsh (top half of picture) have adopted the topography of the former agricultural drains and grips still visible on the freshwater marshes (lower half of picture). At Northey Island poor drainage has aided in the formation of a dense creek network with a high ratio of bare mudflat to vegetated saltmarsh surface.

Breach retreat provides a means to create saltmarsh the form of which is sensitive to previous and current land management. A saltmarsh may take several decades to develop from a mudflat, the form, function and ecology of which is not possible to predict in the early stages of the project. (Dimensions of each image is 1400 m by 800 m. North to top.) Photographs courtesy of the Environment Agency.




(a)




(b)

**Figure 3-5** Naturally regenerated marshes of the Severn Estuary

Oldbury Pill (a) and River Parrett east bank (b) provide two examples of unmanaged saltmarsh restorations in the Severn Estuary region of south-west Britain. These sites provide an indication of how a saltmarsh will develop following a realignment of flood defence in this region. While the age of the River Parrett site is currently unknown the relocation of mediaeval seawalls at Oldbury Pill has been traced to a time prior to the 19$^{th}$ century (Allen, 2000). Here in the Severn Estuary, where tidal ranges are very high both natural and restored saltmarsh sediments consolidate relatively rapidly upon deposition within the high intertidal zone. Because of this the saltmarsh form with a relatively low creek density. Unmanaged restoration of saltmarshes in this area appears to produce saltmarsh with the same geomorphological characteristics and the natural marshes though the functional value of these marshes has yet to be determined. (Dimensions of each image is 1,400 m by 800 m. North to top). Photographs courtesy of English Nature and the Coutryside Council of Wales.

### 3.4.2 Managed realignment: geomorphological development flowing bank breach

Because all UK sites are very young (most are less than six years old) there is limited data from which conclusions can be drawn on the geomorphological development of these sites. In all cases sedimentation has taken place, though with local erosion near the breach as the inlet widens naturally to accommodate the tidal prism of the site. This sedimentation has been associated with halophyte (salt-tolerant plants) establishment mostly in the higher margins of the sites and elsewhere internally at suitable elevations.

In Essex, there is some early evidence that if poor drainage develops on a managed realignment site there is a knock on effect on the establishment of primary colonisation by saltmarsh plants. At the Tollesbury site water-laden sediments have been associated with growth mats of the algae *Enteropmorpha*, which in turn, has been linked to the poorer than expected establishment of *Salicornia* spp. (Reading *et al* 2000). At Orplands although soil water content has not been monitored, measurements indicate that while the natural reference marsh possesses oxygenated surface sediments, those within the restoration site are anoxic (Environment Agency 1999a). This may reflect poor drainage of lower intertidal sediments in possible association with bacterial degradation of plant matter from the underlying soil surface. In both these cases further monitoring is required beyond the five-year programmes to track how soil drainage and oxygen status will change.

Although managed realignment is planned for The Wash (ie North Sea Camp), there are currently no examples in the UK of an experimental restoration of intertidal habitat in sandy conditions. There are examples, however, within the Netherlands from which some early conclusions can be drawn. The Netherlands is a low-lying country, more than half of which lies below sea-level. In its history there has been a constant struggle to maintain extensive Mediaeval flood defences. In the macrotidal Scheldt Estuary, the most southerly of the Netherlands, Sieperda Polder marsh (which forms part of the larger Saeftinge marsh) was land-claimed for arable agriculture and cattle grazing during 1966. Twenty-four years later, in a severe storm of 1990 the protecting summer dike was breached and the Sieperda polder was returned to the intertidal zone. A primary channel was dug to enhance drainage.

Topographical survey Sieperda polder have been limited but, since breaching, surface sedimentation has progressed, landward artificial drainage ditches are infilling with sediments and the main drainage channel is developing a less artificial, more natural form (Figure 3.6). Vegetation coverage reflects the distribution of sediments as muds accumulating in more sheltered areas, possess plant species more tolerant of less well-drained conditions (Eertman in press).

Generally, it would appear that sandy marshes respond more readily to restoration actions than muddy marshes. This may be because sand-rich marshes drain through the sediments and do not suffer consolidation and waterlogging problems. The ecology of sandy marshes is different from muddy marshes and so their ‘functional value’ will also be different.


(a)


(b)

**Figure 3-6** Restoration of a sandy intertidal marsh, Sieperda Polder

Central to figures (a) and (b) is a shallow, wide, channel which after excavation in 1990 is developing a form which appears to be similar to many UK natural sandy systems (such as in The Wash or the Solway Firth). The main creek is adapting to the hydraulic conditions while concurrently the redundant borrow pits and drainage channels are in filling. Sandy systems, in which the sediments are supported by sand grains, rather than clay particles, are less prone to consolidation when drained for agriculture and, on reflooding, should respond more rapidly than muddy systems in terms of accretion, reestablishment of a creek drainage system and vegetation colonisation. The functional values of sandy and muddy coastal systems have yet to be established. Photographs courtesy of R.H.M. Eertman.

### 3.4.3 Dredge material: geomorphological development following placement

The use of hydraulic pipelines to place dredge material on intertidal sites has been widely used in the United States and elsewhere over the past 30 years. These sites have provided a wealthy data set to determine whether the use of dredge material replicates the saltmarshes and mudflats that have been lost (see Streever 2000; Zedler 2000). By and large, while saltmarsh vegetation establishment has commonly occurred on marshes created at the correct elevation, the placement of high water content slurry within a containment area in the high intertidal zone has been found to result in deposits with geomorphological characteristics very different to those of natural marshes. Common features of these higher marshes are failure to develop a surface slope and the formation of a highly consolidated deposit into which the weak tidal currents, found in the high intertidal zone, are unable to cut creek networks (PWA 1994; D. Cahoon pers. comm.; W.J. Streever pers. comm.). Because of this, the ecological value of the marshes in terms of functions supported by the creek systems are lost (Zedler 2000).

Of any coastal system in the United States it is those of San Francisco Bay and southern California that contain saltmarshes which might be considered to be geomorphologically comparable to those of Britain, particularly southern England. In San Francisco Bay, Phillip Williams and Associates (PWA 1994) determined from aerial photographs a number of creek characteristics (drainage density, bifurcation ratio, length ratio and the sinuosity) for two natural marshes and two composed of dredged material. The analysis found on natural marshes (Corte Madera Ecological Reserve and Lameister Tract) and on the lower portion of Muzzi Marsh seaward of the dredge placement, networks of sinuous creek (locally called slough) channels developed. By contrast, on high-elevation marshes restored using dredged material, such as the upper portion of Muzzi Marsh or on the Upper Portion of Faber tract (Figure 3.7), creek channels did not develop, even after two decades of tidal inundation.

Along the east coast of the United States, Shafer and Streever compared 14 east coast dredge material marshes with nearby natural *Spartina*, marshes. Pair-wise comparisons were made on the basis of: (1) edge:area ratio; (2) relative exposure index; (3) elevation profiles; (4) elevation of *Spartina alterniflora*, (5) soil organic carbon content; (6) soil silt-clay content; and (7) below ground plant biomass. It was found from elevation profiles that artificial structures, such as the berms used in the construction of dredge marsh sites to confine the placed slurry, lead to differences between dredge material marshes and natural marshes. The elevation at which *Spartina alterniflora* occurred in dredged material marshes was not significantly different from those of natural marshes. No difference was found either with regards to soil organic carbon and silt content. However, below-ground biomass of dredged material marshes was significantly lower than that of natural marshes. From a geomorphological perspective it was found that, based on the quantification of edge to area ratio, dredged material marshes had fewer ponds and depressions than natural marshes. An analysis of creek structure was not undertaken.

In considering these results, it must be borne in mind that the saltmarshes on the east coast of the United States are found in more modest tidal conditions (those in which spring tide ranges do not generally exceed 2 m and 4 m) than the UK (typically spring tide ranges are much greater than 5 m, possibly up to 14 m) and that *Spartina* marshes are ecologically different from marshes of Northwest Europe being peat- rather than sediment-based. Nevertheless there are some morphological characteristics that differ between natural marshes and dredge material marshes that are likely to occur in similar marshes in the UK. Shafer and Streever (2000) conclude that, if an objective of marsh construction is to replicate natural marsh geomorphology, methods to increase the amount of interconnected marsh edge need to be developed and, secondly, methods of effectively describing and summarising geomorphological characteristics need to be further development.

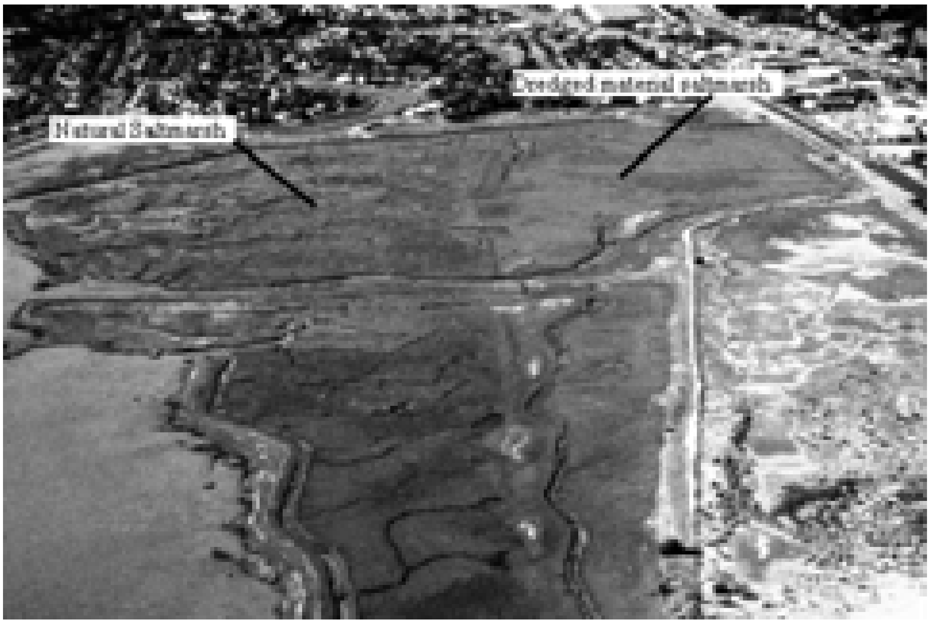
Within the south-east of England the use of dredged material to create intertidal mudflat or saltmarsh for ‘soft’ flood defence purposes is being tested at a number of sites. These sites, created in 1998, include north Shotley Marsh, Trimley Marsh, Horsey Island, Cob Marsh, Old Hall Point, Tollesbury Wick and Wallasea Ness. They are recognised to have a limited life of 20-50 years, sacrificially giving up sediment over this time frame (Environment Agency 1999b). The sediment is derived from channel excavations through Harwich Haven to the port of Felixstowe.

Post-placement monitoring of changing topography and vegetation cover is in progress at these sites, though it is too early to determine what geomorphological developments will take place. Several of the dredged material mudflats/marshes (Wallasea Ness, Tollesbury Wick and Cobb Marsh) consist almost exclusively (>99%) of sediments of sand grade or coarser. Old Hall Point, Trimley and North Shotley and Horsey Island marshes have, however, been constructed with significant amounts of silts and clays (66%-74%) and are more similar to the local natural marshes. The success in creating saltmarsh of flood defence value to protect designated freshwater wetlands behind seawalls on these experimental sites is likely to lead to further recharge works taking place.

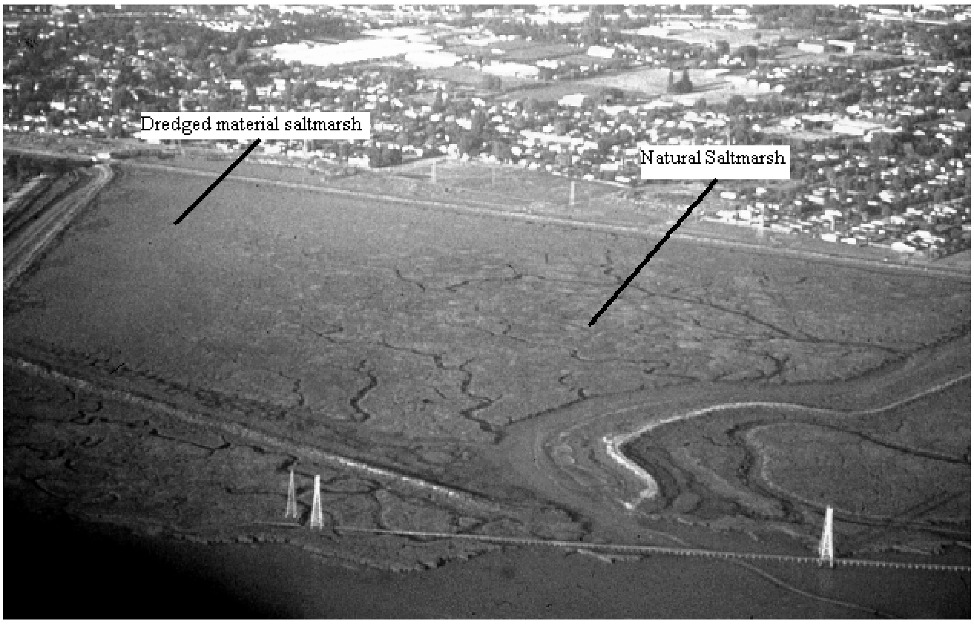
The trials at North Shotley, Trimley and Horsey Island are interesting as here experimental techniques are being tested in order to use high-density slurry (bulk wet weight: c.1.3–1.5 $Mg/m^3$), to create a marsh surface of variable topography and hence improve drainage and ecological diversity. The material is collected using suction-dredging but without the addition of further water to liquidate the slurry before piping into the collection area. Early indications are that this denser material consolidates to accentuate slightly the slope characteristics of the underlying surface (because in lower areas where a greater thickness of material accumulate greater differential surface lowering occurs). At Horsey Island, where the marsh has been created on a natural surface with some relief, there is some evidence of a developing proto-creek network (Dick Allen, Harwich Harbour Authority pers. comm.). If this technique does foster natural creek development then it is a significant advance in the use of dredged material for the restoration of saltmarshes. Moreover, there is anecdotal evidence (Dick Allen, pers. comm.) that benthic invertebrates can survive this dredge process as, at Horsey Island, *Hydrobia* were seen to be moving on the surface of the slurry immediately after piping and placement. A structured scientific investigation is required to assess the value of this ‘high-density slurry’ technique.

In November 2000, the Trimley Marsh site was extended to include an area where a creek network had previously occurred. Channels were excavated into the underlying material and high-density dredge material was deposited in February 2001. Trials are underway to determine whether this engineered approach will enhance or quicken creek system development.

Nearby at North Shotley, the surface lies below the level suitable for saltmarsh development and a mudflat has developed. Here, the gravel bund which protects the lower margin of the site has rolled landward under wave action and provides possible suitable conditions for a second phase of recharge at a higher elevation over the inner region of the site.




(a)




(b)

**Figure 3-7** Saltmarsh restoration with dredged material, San Francisco Bay

Muzzi Marsh (a) and Faber Marsh (b) are morphologically representative of many marshes created with dredged material. Placement of dredge material in the high intertidal zone increases the rate of vegetation colonisation but the formed saltmarsh is often devoid of a creek network. At the margins of Faber Marsh (centre right of picture) low elevation dredged material placement fostered creek development as subsequent natural sedimentation took place. Thus, dredged material placement at lower intertidal elevations may support subsequent saltmarsh development. Creation of mudflats with dredged material is more technically problematic because of their dynamic nature. A placed mudflat may erode if exposed to excessive wave energy or, if not, will naturally evolving into a saltmarsh. Photographs courtesy of PWA-Ltd.

## 3.5 Conclusions

The features that make natural coastal wetlands (saltmarshes and intertidal mudflats) unique are the same features that present a significant challenge to those attempting to create such wetlands. Their geomorphology is complex; they are biologically diverse; and they are vulnerable to sea-level rise. Geomorphological features, such as surface topography and creek density, largely determine the whole range of other ecological interactions that take place on and within a marsh. These linkages between habitat form and function are very poorly understood, even for natural intertidal habitats, and so predicting how a specific habitat will develop on restoration, beyond a broad generalisation, is subject to great uncertainty.

Nevertheless, unmanaged retreat sites show us that it is possible to restore intertidal flat and saltmarshes given time and suitable environmental conditions (tides, sediment supply etc). These restored habitats may, however, have different physical characteristics and environmental functions to those of nearby natural marshes. This is not to say that because they are different that they are necessarily of poorer environmental quality. Unmanaged retreat marshes in Essex have, for example, a higher creek density than natural marshes, and as such, may have higher value in terms of support of fisheries. This may be at the expense of other functional values. For example, these unmanaged retreat sites tend to have a high proportion of cover of Sea Purslane *Halimione portulacoides* which makes those areas unsuitable for grazing waterfowl. The science and techniques to assess these differences are not yet available.

What is clear is that time is a major component in restoring intertidal habitats and that if a marsh of ‘natural’ form is required then it is preferential to allow a natural succession from low intertidal through primary marsh to mature marsh. The actions to restore saltmarsh and mudflat in Essex for flood defence purposes in Essex shows that the rate of this process can be enhanced by engineering techniques (such as placement of dredged material, planting of vegetation and digging of channels) but this places artificial constraints on natural development. As a consequence these marshes are often very different in form, and probably function, from natural marshes. The evidence from the United States is that, when dredged material is placed too highly in the intertidal frame, marshes form without a creek network and the functions associated with them are lost. If engineering works are used to create a creek network then the restored form should be based upon natural hydraulic laws, which relate creek dimensions to features such as the size of the marsh area, and seek to replicate the form of natural habitats.

# 4. Success of mitigation and compensation schemes at creating suitable invertebrate habitat

## 4.1 Introduction

Many of the nationally and internationally important populations of waterfowl are attracted to intertidal habitats by the availability of invertebrate prey. Birds selectively feed in areas where their rate of energy return is highest, and thus wintering waterfowl densities closely mirror the densities of their invertebrate prey (Weber & Haig 2000; Meire 1996; Piersma *et al* 1993; Yates *et al* 1993; and see chapter 2). The preferred habitats and prey items of some of the more abundant species of waterfowl are shown in Tables 4.1 and 4.2. As well as their importance as bird food, invertebrates are themselves a component of biodiversity and may play an important role in biogeochemical cycling. Establishment of a "normal" community of invertebrates, with similar abundance, diversity and species composition to nearby "natural" sites is a potential indicator of the functional integrity of a created site.

## 4.2 Theoretical background

When a new area of intertidal habitat is created adjacent to existing similar habitats, most species of invertebrates will be able to disperse freely into the newly created area. However, the time taken for populations to become established will be affected by the life history biology and habits of the species concerned. Mobile species such as the crustacean *Corophium volutator* regularly leave the sediment and swim in the water column, so will be readily able to swim in to the created habitat. Generation times for this species are shorter than one year and reproduction takes place for much of the summer, so if conditions are suitable we would expect this important prey species to become established quite rapidly. By contrast, other important prey species are not mobile, breed during a short period of the year, and are relatively long lived. Establishment of populations of these species with a normal population structure will take considerably longer. For example, larger individuals of the bivalve species *Scrobicularia plana*, *Mya arenaria* and the cockle *Cerastoderma edule* will only be present several years after habitat creation, as adults are essentially non-mobile, colonisation of new areas can take place only by the settling of larvae from the plankton, which takes place only once a year, and individuals take several years to grow to the size at which they are eaten by waterfowl. A similar model has been used by Newell *et al* (1998) in a discussion of the ecological effects of dredging. The longest potential delays will occur for species that lack a planktonic larva and have limited powers of dispersal. These species are often small, but may provide food for smaller shorebirds such as knot and grey plover (see Table 4.2). This is not simply a theoretical possibility. Levin *et al* (1996) observed delayed colonisation of a created saltmarsh site in North Carolina by taxa that lacked a planktonic larva, and almost complete absence of oligochaetes even after four years. She attributed this to their lack of any dispersive phase. She suggested that these difficulties might be overcome by "seeding" of the created habitats with sediment from existing similar locations.

Clearly any estimates of time required for colonisation that are based on a knowledge of species' population biology are minimum estimates, as they assume that sediments are suitable for colonisation immediately after habitat creation. It will take rather longer for invertebrate populations to establish if sediments are initially unsuitable. For example, invertebrate colonisation of a new area of mudflat at Seal Sands in the Tees estuary, took longer than these theoretical minimum times possibly due to the over-compaction and substrates during construction and protection from wave action (Evans *et al* 1998, 2000).

In the next section, we review the available data from the monitoring of invertebrate populations at created sites in the UK and elsewhere, focussing particularly on whether invertebrate populations at created sites are similar to those at nearby control sites and, if so, the time taken for this equivalence to be achieved.

## 4.3 Summary of monitoring data

### 4.3.1 Summary of monitoring data

As shown in Tables 1.1–1.3, invertebrate data have been collected only at a minority of habitat creation sites. Sites where invertebrate monitoring data are available are summarised in Table 4.3. The quality of data varies substantially, but the best studies include several natural reference sites and several created habitats or replicate samples from different areas of a single created habitat (see Box 4.1 for an example). In the UK, the most comprehensive invertebrate monitoring data are for a mudflat created by restoring exchange of salt water into a previously reclaimed area on Seal Sands (Evans *et al* 1998, 2000) and at the Tollesbury managed retreat site (Reading 1996, 1999), although the Tollesbury study has reference sites that are rather more homogeneous in sediment characteristics than those on the retreat site. We will return to the issue of choice of control sites in our recommendation for monitoring. In view of this sparsity of data for UK sites, we base our discussion largely on data from other parts of the world.

**Box 4.1 An example of a well executed invertebrate monitoring programme**

Talley and Levin (1999) studied five natural *Salicornia* marshes in southern California, and nearby created marshes in four of the five areas. Their results led them to suggest that faunal recovery in a created *Salicornia* marsh could take 10 or more years. They also noted that there was considerable variation in the species composition of macrofauna between different bays, so that it was essential that natural reference sites were close to the created marsh being studied.

This study has a number of desirable features.

- There are replicate created and reference marshes, with a number of samples taken from each; allowing comparisons to be made within and between areas.
- The authors were careful to restrict their comparison between similar created and natural marshes, so *Salicornia* marshes were chosen as natural reference sites and tidal elevation was controlled for in sampling both the created and natural marsh;
- Sediments have been sieved through a 0.3 mm sieve, which is fine enough to sample essentially all macrofauna;
- invertebrates were identified mostly to species level;
- species abundance data were analysed using the best available statistical methods
- The report appears in the peer reviewed literature, meaning that it has been through a more rigorous quality control system than un-refereed “grey literature”

In some cases, colonisation by invertebrates has been relatively rapid (eg Ray 2000) in other cases, it has taken several years for invertebrates to recolonise (eg Evans *et al* 1998, 2000; Levin *et al* 1996; Moy & Levin 1991; Talley & Levin 1999). In the extreme, the invertebrate community at constructed sites remains different from that at natural sites 15 or even 25 years after construction (Sacco *et al* 1987; Minello & Webb 1997; Craft 1999). For saltmarsh sites, the time taken for invertebrates to recolonise can be particularly long. This phenomenon is paralleled by the vegetation communities found on some unmanaged retreat sites which are still different 100 years since inundation (Burd 1994) and the failure of saltmarsh vegetation to develop on some apparently suitable unmanaged retreat sites (French *et al* 1999; Hughes 2001). These long delays in the development of normal invertebrate populations may reflect the lower exposure to disturbance by waves and currents of sediments on saltmarshes as compared with those lower in the intertidal zone. In the more dynamic environment of the lower intertidal, sediments are resuspended and deposited more frequently, so will more rapidly "forget" the consequences of processes that occurred during habitat creation. It is perhaps significant that ingress of seawater to the Seal Sands site is through a sluice (Evans *et al* 1998, 2000), so this mid-intertidal site is largely protected from wave action, although compaction of the sediments by earth moving equipment is another possible explanation for delayed invertebrate recruitment (Evans *et al* 1998). However, there have been only a very small number of studies of created mudflats and it would not be safe to assume that the development of invertebrate communities will always occur reasonably rapidly on mudflats until considerably more data are available on such sites.

The failure of communities to converge may also result in part from differences in sediment characteristics between the created and natural reference marshes. Dredged sediment may have rather different grain size to sediments that are normally deposited on the upper intertidal (eg West *et al* 2000). Even when the grain size is comparable, sediments on created marshes may contain much lower organic carbon concentrations than natural marshes (eg Lindau & Hossner 1981; Langis *et al* 1991; Craft *et al* 1999).

Finally, marshes that result from managed or unmanaged retreat, or are created by excavating upland areas, may have sediments that are overconsolidated in comparison with natural marsh sediments. Crooks and Pye (2000) discuss the detailed mechanisms that can lead to overconsolidation of soils on retreat sites. Such very consolidated sediments are a very poor habitat for invertebrates, as shown by the low invertebrate abundances on the foreshore at North Shotley prior to mud placement (Posford Duvivier 2000, 2001) and low diversity and abundance of invertebrates in the eroding consolidated sediment that is often found in the intertidal zone of the Humber Estuary (Grant pers. obs.). These overconsolidated sediments are often poorly drained, which can favour the growth of algae rather than the growth of saltmarsh plants (Crooks & Pye, 2000). If overconsolidation is the key reason why saltmarsh plant communities have often not developed on unmanaged retreat sites after several decades, then it is possible that a similar reason may explain the lack of convergence of macrofaunal communities.

Some causes of sediment unsuitability, such as sediment anoxia, may be temporary, and animals will be able to colonise after an initial time lag. Other factors may make sediments unsuitable for colonisation indefinitely. If sediments differ in grain size from those on nearby reference sites, then species composition will be different and if the sediments are coarser grained, then the invertebrate biomass will often be lower and created sites may support lower abundances of waterfowl, although if sufficient fine grained sediment is available nearby, the fine material will be transported to the created site. If the created habitat consists of very consolidated muds (see Section 3.4), the sediments may be too firm to allow invertebrates to burrow. This often occurs at eroding intertidal sites, including some of those in Essex (c.f. Posford Duvivier 2000, 2001, and comments on North Shotley mud placement in Table 4.3). Invertebrate populations are likely to remain low at such sites until sufficient depth of unconsolidated sediment has accumulated.

As noted previously, much attention has been focussed on the creation of saltmarshes, and there has been rather little interest in creating mudflats or monitoring the development of invertebrate communities in mudflats resulting from accidental seawall breaches. However, there is some evidence that colonisation of mudflats by invertebrates can take place rather faster than saltmarshes. After experimental defaunation of large areas on Balgzand, in the Dutch Waddensea, most species recolonised within 12 months, with seasonally breeding species settling into the experimental treatments during the first breeding season (Beukema *et al* 1999). At two sites in Maine, control and created sites showed similar diversity, abundance and species composition after two years (Ray *et al* 2000). However, in the experimental defaunation studies at Balgzand, it was three to four years before biomass recovered to normal values. At Seal Sands in the Tees Estuary, *Nereis* did not become abundant for three years after flooding, and *Macoma* was still rare after seven years (Evans *et al* 1998, 2000) and at Orplands, bivalves have not colonised the managed retreat site after four years, despite being common on the adjacent intertidal. Larger individuals and bivalves in particular constitute important prey items for a number of species of shorebird, so their absence is likely to have some adverse effects on the suitability of the sites as bird feeding areas.

### 4.3.2 Implications of results of monitoring studies

In some studies, the invertebrate communities of created sites have converged quite rapidly to become similar to those at natural reference sites. However, in a number of studies this convergence takes several years and in other cases the invertebrate communities have very different characteristics from control sites many years after creation. As predicted in the introduction, it can be several years before the full size range of large bodied invertebrates occurs (eg LaSalle *et al* 1991). As invertebrate food resources are a major determinant of bird distribution patterns, it is likely that these differences in invertebrate community composition will be reflected in differences in waterfowl abundances. The more detailed implications of this for bird populations are drawn out in Chapter 6. There are a number of cases where convergence of invertebrate communities has taken about five years to occur, but we do not understand why convergence frequently takes this long. Ideally, creation of new habitats in mitigation for the loss of others would therefore take place at least five years before development.

Failure of invertebrate communities to converge may reflect some unknown flaw in the created habitats (over-consolidation of sediments is a possibility in some cases) or it may simply reflect that environmental conditions at the created habitat are “normal” but are not identical to those at the reference sites. The most likely differences would be in sediment grain size or tidal elevation. Monitoring programmes need to be able to distinguish between differences that reflect differences in sediment grain size, tidal height etc, and differences that reflect incomplete establishment of a normally functioning community. Few of the published or unpublished monitoring studies have collected sufficiently detailed data on tidal height and sediment grain size to make any assessment of the importance of this. In any case, we do not yet have a full understanding of how sediment properties determine the invertebrate communities which occur. In broad terms, we can identify that some species are characteristic of sandy sediments while others are characteristic of muds. Moving beyond these broad brush generalisations to a more detailed ability to predict the species which will occur at a created site and their relative abundances is severely hindered by the fact that relationships between environmental factors and the invertebrate communities that occur at a particular site are not well understood, and in particular by the fact that the most frequently collected measures of sediment characteristics, such as mean grain size, are relatively poor predictors of community composition (Snelgrove & Butman 1994; Newell *et al* 1998; Seiderer Newell 1999).

The fact that colonisation can take several years indicates that environments may be unsuitable for reasons that are not apparent to a human observer. It is thus important to guard against being too complacent in claiming that replacement intertidal habitats can be readily created.

### 4.3.3 Implications for monitoring programme design

The full assessment of invertebrate colonisation of a created site requires quantitative sampling of invertebrate populations at several locations in the created wetland, and several locations at nearby natural sites. Natural variability between sites and over time mean that sampling a single natural reference site will rarely be adequate, and in some cases similar reference sites may not be available (Ray 2000; West *et al* 2000 and see Grant & Millward 1997 for a discussion of the data interpretation problems that this creates). In the case of intertidal mud placement schemes, pre-placement invertebrate sampling are a helpful addition. For example, the Tollesbury managed retreat site provides one of the best available invertebrate monitoring data from a UK site, but uses only a single natural marsh control site. In this example, the heterogeneity of sediments between the sampling sites in the retreat area is acknowledged to contribute to the greater diversity of invertebrates in the retreat site as compared with the control site (Reading *et al* 1999). Keeping track of other environmental variables is important too. The report on three years of monitoring at Tollesbury (Reading *et al* 1999) notes that invertebrate colonisation was more rapid at lower lying, wetter, sites, but the data are not included to allow this to be readily assessed.

The larger individuals of invertebrate species that are important bird prey items may occur at densities that are too low to be adequately sampled using the small area cores that are commonly used in ecological studies of estuarine mud flats. The ideal sampling programme would include both the taking of core samples to quantify the abundance of smaller individuals and either sieving of larger core samples through a coarse meshed sieve or methods such as the raking of sediment surface to sample larger organisms such as bivalves (eg LaSalle 1991; Ray *et al* 1999, 2000). Because bird predation of many invertebrates is strongly size selective, and only the largest prey are accessible to birds, it would be extremely valuable if the size distribution of individuals within the population is quantified in addition to the numbers of individuals present. It is essential that each sample is accompanied by characterisation of tidal height and sediment properties (mechanical, grain size, organic carbon content) to allow the influence of these to be factored out from the comparison between control and created sites. Created habitats are usually located within estuaries where the tidal regime is rather different from that on the open coast, and salt water is frequently allowed onto sites via a pipe or sluice rather than a large breach in the sea wall, so the tidal regime within a created site can be very different from that in the adjacent estuary. It is, in consequence, very important that tidal influence is reported in terms of number of inundations per year rather than simply giving heights above OD (or its equivalent outside the UK).

If good quantitative data are available on invertebrate abundance, then sophisticated statistical methods are available to make a comparison of community composition between control and created sites (see Talley & Levin 1999 for an example). There are a number of such examples from North America. For example, (Ray *et al* 2000) carried out a quantitative comparison between two created mud flats and adjacent control areas in Maine. This paper represents a peer-reviewed summary of work reported in more detail in the grey literature (Ray *et al* 1999), and this more detailed study includes data on the abundance and size structure of larger infaunal invertebrates such as *Mya arenaria* and *Nereis virens.* Unfortunately, examples of good monitoring are depressingly few in the UK. Exceptions are the monitoring of a re-created intertidal mud flat on Seal Sands (Evans *et al* 1998, 2000) and the Tollesbury managed retreat site.

## 4.4 Conclusions

At the beginning of this section we made some predictions based on the life history biology of marine invertebrates. The monitoring data support the predictions of this model in some cases. When species colonise largely by settlement of juveniles from the plankton, they can re-establish at a site during the next breeding season (late spring and summer). Abundance and biomass values usually take between two and four years to recover, and it may take even longer before the largest individuals of long lived species are found. Information on the faunal colonisation of subtidal environments disturbed by dredging gives a similar picture to that from created intertidal habitats (see Box 4.2). However, there are cases, particularly, but not exclusively, on saltmarshes where infaunal recovery takes much longer than this. This presumably results from the condition of the sediments being initially unsuitable for some invertebrate species. Until the reasons for these delays are clarified, it is not possible to claim with complete confidence that a mitigation scheme will be successful in terms of the invertebrate communities that become established there, and therefore in terms of the supply of food available to waterfowl. Clearly the time-lags for invertebrate colonisation support the suggestion that any new habitat should be created in advance of it being first required as a replacement habitat by waterbirds. In some cases, the abundance and diversity of infauna are not equivalent to those of nearby reference sites after more than a decade. If this delay is due to persistent differences in environmental conditions, it may be avoided by matching the environmental conditions at the created site more closely to those of the site for which it is intended to substitute. However, until we understand the reasons for these long delays of convergence of the ecology of created sites to that at natural reference sites better we cannot claim that it is possible to readily create equivalent habitats.

**Box 4.2 Lessons from studies on dredged subtidal sediments**

In view of the paucity of data on mudflats, we have also sought to draw lessons from other literature on the colonisation or recolonisation of marine sediments in other contexts. There are some analogies between a created intertidal flat and areas of subtidal sediment that have been disturbed by dredging or aggregate extraction (although areas of aggregate extraction usually involve coarser sediments than many of the intertidal sites that are important to waterbirds). A factor that may limit the applicability of such studies to habitat creation is that dredging often selectively removes coarser sediment, so some of the changes in invertebrate populations may be due to changes in sediment grain size in the remaining material. Nevertheless, monitoring studies at dredged subtidal sites indicate that patterns of recovery are broadly in line with our theoretical predictions. Van Dalfsen *et al* (2000) reported rapid colonisation by opportunistic species at study sites in Denmark and the Netherlands, with full recovery taking between two and four years. They also reported that the period of recovery was perhaps longer than this at a site in the Mediterranean. At a site near Dieppe in the English Channel, species richness recovered after 16 months, but density and biomass were respectively 40% and 25% lower after 28 months (Desprez 2000). At a site in the Bay of Blanes, Spain, Sarda *et al* (2000), reported that biomass returned to normal after dredging within two years but that some longer-lived species took longer than this to recover. In a review of the impact of dredging, Newell *et al* (1998) suggest that fauna take between two to four years after dredging to recover. They suggest that faunal recovery in muds may only take six to eight months, but this included sites subject to frequent disturbance in which tidal currents may transport juveniles. They also note that in one of the intertidal sites studied by van de Veer *et al* (1985) no faunal recovery had taken place after 16 years.

**Table 4-1** Summary of habitat preferences of waterfowl on British estuaries

(Sources: [1]Rehfisch *et al* 1997; [2]McCulloch & Clark 1991; [3]Garter & Ebbinge 1997; [4]Austin *et al* 1996; [5]Percival & Evans 1997; [6]Yates *et al* 1993; [7]Yates & Goss-Custard 1991; [8]Prater 1981; [9]Cramp & Simmons 1983). Blank cells indicates no data available or no significant relationships found

| Species | Sediment | Vegetation | Topography | Exposure |
|---|---|---|---|---|
| **Brent Goose** | | *Zostera* beds [5]<br>Saltmarsh [3] | | |
| **Shelduck** | Mud<br>Muddy sediment [4] | | Low tidal range[4]<br>Narrow Estuary [4] | Low fetch [4] |
| **Mallard** | Mud/sand sediment [4] | Low saltmarsh area [4] | | Low swell [4] |
| **Oystercatcher** | Mud/sand sediment [4]<br>Fine sand, low silt sediment content [6]<br>Greater sand area [7] | Saltmarsh [4] | Wide estuary [4]<br>Wide shore [7] | High swell [4]<br>High fetch [4] |
| **Ringed Plover** | Muddy sediment [1]<br>Sandy areas [9] | | Low tidal range [4] | Low swell [4] |
| **Grey Plover** | Mud/sand sediment [4]<br>Low silt sediment [6]<br>High mud area [7]<br>Muddy sediment [8] | | Wide shore [7] | |
| **Knot** | Greater area of mud [4,7]<br>Low area of fine sand, high area of silt sediment [6,8] | Saltmarsh [4] | Wide channels [4] | |
| **Dunlin** | Muddy sediment [2, 4,8]<br>Little sand [1]<br>Low coarse sand and high silt sediment [6]<br>Greater areas of mud [7] | | Narrow shore [1,4]<br>High tidal range [4]<br>Narrow estuary [4] | Low swell [4]<br>Low fetch [4] |
| **Curlew** | Muddy sediment [4,8]<br>Little sand [1]<br>Low fine sand sediment [6]<br>Greater area of mud [7]<br>Inland fields [8] | | Narrow estuary [4] | |
| **Redshank** | Muddy sediment with little sand [1,8]<br>Low coarse sand, high clay sediment [6]<br>High area of mud[7] | Saltmarsh [8] | Low tidal range [4]<br>Narrow estuary [4]<br>Wide shore [7] | |
| **Turnstone** | Muddy sediment [4]<br>Rocky shores [9] | | | |

**Table 4-2** The diet of waterfowl in winter (taken from Field *et al* 1998)

Main food items are shown in capital letters. (Sources: Cramp & Simmons 1977; Cramp & Simmons 1983; Campbell 1946; dit Durell & Kelly 1990; dit Durell, Goss-Custard & Caldow 1993; Goss-Custard 1969; Goss-Custard & Jones 1976; Goss-Custard, Jones & Newbery 1977; Harris 1979; Moreira 1994; Olney 1965; Owen 1973; Percival & Evans 1997; Perez-Hurtado, Goss-Custard & Garcia 1997; Pienkowski 1983; Prater 1972; Prater 1981; Rands & Barkham 1981; Whitfield 1990; Worral 1984)

| | | **Brent Goose** | **Shelduck** |
|---|---|---|---|
| **Plants** | Algae | ***ENTEROMORPHA*** *Cladophera*<br>***ULVA*** Brown and red algae | *Enteromorpha*<br>*Vaucheria* |
| | Gramineae & Cyperaceae | *Puccinellia* Wheat<br>*Festuca* Barley<br>*Spartina* | Seeds:<br>*Scirpus* |
| | Other families | ***ZOSTERA*** *Aster*<br>*Salicornia* *Triglochin* | Seeds:<br>*Sueda* *Salicornia*<br>*Atriplex* |
| Invertebrates | Molluscs | | ***HYDROBIA*** *Littorina*<br>*Cardium* *Skenea*<br>*Macoma* *Paludina*<br>*Mytilus* *Tellina*<br>*Montacuta* *Nucula*<br>*Cingula* *Mya*<br>*Buccinum* *Theodoxus* |
| | Crustaceans | | Small crabs Sandhoppers<br>Shrimps *Artemia*<br>Prawns *Corophium* |
| | Annelids | | Nereis |
| | Arthropods | | Orthoptera Chironomidae larvae<br>*Carabus nitens* |

| | | Oystercatcher | Ringed Plover | Grey Plover |
|---|---|---|---|---|
| **Plants** | Algae | | | |
| | Gramineae Cyperaceae | | | |
| | Other families | | | |
| **Invertebrates** | Molluscs | ***CARDIUM*** *Patella*<br>***MYTILUS*** *Tellina*<br>***MACOMA*** *Scrobicularia*<br>*Littorina* *Mya*<br>*Nucella lapillus* | *Macoma*<br>*Hydrobia*<br>*Littorina* | ***HYDROBIA*** *Mya*<br>***MACOMA*** *Littorina*<br>*Cardium* Dreissenidae<br>*Mytilus* |
| | Crustaceans | *Carcinus*<br>*Crangon*<br>*Amphipods* | Corophium Eurydice<br>*Bathyporeia* Shrimps | ***CARCINUS*** *Retusa*<br>*Upogebia* *Corophium*<br>*Cleistosoma* *Crangon* |
| | Annelids | *Nereis*<br>*Arenicola* | ***NEREIS*** *Scopolos*<br>***NOTOMASTUS*** *Phyllodoce*<br>*Arenicola* Small oligochaetes | ***NEREIS*** ***LANICE***<br>***NOTOMASTUS*** *Phyllodoce*<br>*ARENICOLA* |
| | Arthropods | | Insect adults and larvae | |

| | | Knot | Dunlin | Curlew |
|---|---|---|---|---|
| **Plants** | Algae | *Enteromorpha* | | *Ulva* |
| | Gramineae Cyperaceae | Seeds:<br>Various | Seeds:<br>*Scirpus* | Various grasses and cereals |
| | Other families | | *Zostera*<br><br>Seeds:<br>*Ruppia* *Najas* | Mosses<br>*Equisetum*<br>*Rubus* berries |
| **Invertebrates** | Molluscs | ***MACOMA*** *Zua*<br>***CARDIUM*** *Limosa*<br>***HYDROBIA*** *Mya*<br>*Tellina* *Homalogyra*<br>*Littorina* *Retusa*<br>*Mytilus* *Rissoa*<br>*Paludina* | ***HYDROBIA*** *Tellina*<br>***LITTORINA*** *Mytilus*<br>***RISSOA***<br>*Retusa*<br>***THEODOXUS*** *Cardium*<br>***MACOMA*** *Scrobicularia* | ***SCROBICULARIA*** *Mya*<br>***MACOMA*** *Cardium*<br>*Mytilus* *Hydrobia* |
| | Crustaceans | *Carcinus* *Balanus*<br>*Corophium* *Crangon*<br>*Gammarus* | Amphipods Mysidacea<br>*Carcinus* *Cladocera*<br>*Crangon* *Artemia salina* | ***CARCINUS*** *Gammarus*<br>*Crangon* *Bathyporeia*<br>*Corophium* *Orchestia* |
| | Annelids | *Nereis*<br>Oligochaetes | ***NEREIS*** *Scolopolos*<br>***NEPHTYS*** *Arenicola* | ***NEREIS*** ***LANICE***<br>***CIRRIFORMIA*** *Arenicola* |
| | Arthropods | Insect adults and larvae | Insect adults and larvae | Insects inland |
| | Other | Hydrozoa Small starfish | | |

| | | Redshank | | Turnstone | |
|---|---|---|---|---|---|
| **Plants** | Algae | | | Various | |
| | Gramineae Cyperaceae | | | *Carex* | *Kobresia* |
| | Other families | | | Seeds:<br>*Juncus*<br>*Polygonum* | <br>*Pedicularis* |
| | **Other** | | | **Mosses** | |
| Invertebrates | Molluscs | ***MACOMA***<br>***HYDROBIA***<br>*Scrobicularia*<br>*Littorina* | *Cardium*<br>*Tellina*<br>*Mytilus* | ***LITTORINA***<br>***MYTILUS***<br>***MACOMA***<br>*Cardium*<br>*Patella* | *Hydrobia*<br>*Theodoxus*<br>*Lymnaea*<br>*Calliostoma*<br>*Lepidochitona* |
| | Crustaceans | ***COROPHIUM***<br>***CRANGON*** | ***CARCINUS***<br>*Gammarus* | ***GAMMARUS***<br>***BALANUS***<br>***CARCINUS*** | *Eupagurus*<br>*Talitrus*<br>*Hyale* |
| | Annelids | ***NEREIS*** | ***NEPHTYS*** | *Nereis* | *Lumbricillus* |
| | Arthropods | | | Insect adults and larvae | |
| | Other | | | *Ophiuroidea* | *Psammechinus miliaris* |

**Table 4-3** Summary of invertebrate monitoring data.

Data on fish usage are also included, as a number of studies have monitored site use by both fish and mobile crustacea such as shrimps

| Site and construction method | Nature of invertebrate data | Conclusions | Reference |
|---|---|---|---|
| **Mudflats/tidal channels – UK** | | | |
| North Shotley Mud placement | Cores sampled at four sites, pre-placement and every 3 months for 2 years afterwards, plus adjacent control sites. Pre- and post-placement surveys used different methods. Post-survey cores rather small (5 cm diameter) | Initial reduction in species due to smothering, but increase thereafter. Placement "has increased the value of the intertidal;" probably because foreshore prior to the mud placement consisted of consolidated mud. | Posford Duvivier 2000 See also Coastal and Estuarine Research Unit (2000) and Posford Duvivier (2001). |
| Essex Foreshore recharge works, 1998-2002 (Horsey Island, Cobmarsh Island, Old Hall Point, Tollesbury Wick, Wallasea Ness) | 5 replicate 10 cm diameter cores sampled before and after mud placement on upper, middle and low shore and an adjacent control site at each of the five sites. | Report simply presents summary data. At Horsey Island, much greater abundance and diversity at control site. At Trimley Marsh, control site much lower in both diversity and abundance. Others intermediate between these two extremes. | Environment Agency 1999 |
| Seal Sands, Tees Estuary. Managed realignment, with water entry via sluice. | 8 cm diameter cores sieved at 0.5 mm at up to 21 sites in restored area and reference sites on adjacent mudflats. | *Nereis* did not become abundant until 3 years after flooding; *Hydrobia* density lower than on adjacent mudflat after 4 years. *Macoma* still rare after 7 years but *Nereis* and *Hydrobia* more common on created mudflat by 2000 | Evans *et al* 1998, 2000 |
| See also Orplands and Tollesbury managed retreat sites under saltmarsh heading below. | | | |
| Mudflats/tidal channels – non-UK | | | |
| Balgzand, Dutch Wadden Sea (experimental defaunation rather than habitat creation). | Defaunation of 120 $m^2$ patches by covering sediment with mats of synthetic material for 3 months. Repeated 11 times, and samples taken from within plots and from undisturbed reference sites 5m away. 0.1 $m^2$ samples sieved at 1 mm, 0.01 $m^2$ samples sieved at 0.5 mm. | Species number largely recovered after 6-12 months, and number of individuals after 12 months, as most species settle from plankton in summer. Recovery of biomass took 3-4 years. | Beukema *et al* 1999 |

| Site and construction method | Nature of invertebrate data | Conclusions | Reference |
|---|---|---|---|
| Sheep Island, Jonesport, Maine, USA. Intertidal dredge spoil placement | 7.5 cm cores, sieved at 0.5 mm, plus some sampling of larger macrofauna. Comparison with nearby natural reference site, although there were differences in sediment texture between the wo. | Infaunal diversity, abundance and species composition similar after 2 years. Some species more common on constructed area | Ray 2000 |
| Beals Island, Jonesport, Maine, USA. Intertidal dredge spoil placement | As above | Infaunal diversity, abundance and species composition similar after 2 years. Some species more common on constructed area. | Ray 2000 |
| Pamlico Estuary, North Carolina. Excavated creek and adjacent planted marsh. | 0.02 $m^2$ core samples taken in shallow water in one marsh creek created in 1980-81 (tidal influence allowed in 1983) and four nearby natural creeks. Quarterly sampling 1985-1995 | Little difference between created and natural creeks, although authors state that "numbers of species ….were initially lower than the natural creeks" | West *et al* 2000 |
| Chehalis River, Grays Harbor, Washington State. Slough (tidal creek) created by excavation. | Determination of growth rates in juvenile salmon from rings in otoliths and examination of gut contents. Comparison between natural and created creek. | No differences in growth rates, but some differences in taxonomic composition of gut contents and salmon from the created creek had emptier stomachs. | Miller & Simenstad 1997 |
| Barn Island, Connecticut. Tidal exchange restored to area that had been converted to freshwater marsh by excluding tidal flow (Sinicrope *et al* 1990) | Examination of gut content of Mummichogs, *Fundulus heteroclitus*. Restored marsh area compared with two control areas. | During the summer fish in the restored marsh ate less food per unit body mass than those in other marsh areas. | Allen *et al* 1994 |
| Lincoln Avenue wetland system, Puyallup River Estuary, Tacoma, Washington. Excavated landfill and construction of dyke (equivalent to Gog-Le-Hi-Te Wetland in Simenstad and Thom 1996) | Examination of gut contents in juvenile salmon. No natural reference site available. | Fish forage successfully in the created area, but in the absence of a reference site "equivalency remains untested" | Shreffler *et al* 1992 |
| Mission Bay, San Diego. Planted marsh on dredge spoil | Fish use assessed using minnow traps located in creeks. Created marsh compared with nearby natural reference marsh | Three years after marsh creation, species richness and dominance similar between created and reference marsh, but abundance lower and populations biased towards large individuals on the created marsh. | Talley 2000 |

| Site and construction method | Nature of invertebrate data | Conclusions | Reference |
|---|---|---|---|
| Hiroshima Bay, Japan, Sediment placement | 7 constructed flats and 3 natural reference sites. 25 cm square cores sieved at 1 mm | No clear differences in biomass between natural and constructed sites 6-23 years since creation. Two constructed sites had very high biomass of bivalves, but these were artificially introduced. | Lee *et al* 1998. |
| Saltmarshes – UK | | | |
| Orplands, Essex, managed realignment (part of this site has not been colonised by plants, so could be considered as a mudflat site). | Yearly monitoring, up to 4 years post-breach. 1 mm sieve. Fauna grouped into polychaetes, crustacea and *Hydrobia* 11 locations on retreat site and 3 on nearby natural reference marsh | Counts generally very low, even in reference sites. "Polychaete worms and…. *Hydrobia* have become well established" [by 4 years] "Bivalves do not appear to have colonised the site despite the presence of substantial populations of several common estuarine species on the intertidal mudflats just beyond the old seawall" | Environment Agency 1999 |
| Tollesbury, Essex, managed realignment by breaching seawall (part of this site has not been colonised by plants, so could be considered as a mudflat site) | 9 replicate 10 cm diameter cores from seven areas within the realignment and one reference area on adjacent existing marsh site. Size of Polychaetes, bivalves and gastropods measured in addition to counts of all species. | After 2 months, 14 species in realignment area, 13 species in marsh with 10 in common. Densities in natural marsh much higher (report states a factor of 2-10, but some differences are much greater than this). Largest individuals of *Hydrobia ulvae* and *Nereis diversicolor* occurred on the natural marsh. | Reading 1996 |
| | | After 3 years, 19 species found in realignment site, as compared with 11-13 in natural marsh. "probably reflects the greater diversity of sediment types occurring within the study area". Three species at higher density in the marsh; seven at higher densities in the realignment area; two species at equal densities in both. Most intertidal invertebrates in the realignment area were concentrated at sites that remained wet at low water. *Hydrobia* larger on realignment area, but *Nereis* smaller. | Reading *et al* 1999 |

| Site and construction method | Nature of invertebrate data | Conclusions | Reference |
|---|---|---|---|
| Saltmarshes – non-UK | | | |
| Port Marsh, Newport River, North Carolina, USA. Excavation of dry dredge spoil, followed by planting. | Infauna sampled using 4.8 cm diameter cores, sieved at 0.3 mm. Sampling at increasing time intervals up to 52 months after creation. Epifauna sampled using 0.125 $m^2$ quadrats. Comparison with nearby natural reference marsh. | Initial colonisation by opportunistic polychaetes; Species richness increased during four year period, and dominance by initial colonists decreased, but equivalence not attained even after four years. | Levin *et al* 1996. |
| Seven created marshes, North Carolina, 1-17 years after creation. Includes Uncle Henry's marsh; Harker's Island, Oregon Inlet, and the three sites discussed belowy | 3 cm core, sieved at 250 µm<br>Comparison with adjacent natural reference marshes | Same species present in both, but densities in created marshes about half those in natural reference marshes | Sacco 1989, in Moy and Levin, 1991 |
| | | In all but one cases, total densities were lower on created marshes than on the paired reference marshes, although the pairwise difference was significant in only one case. For individual taxa, 16 of these pairwise comparisons showed a significant difference, with 13 of these being higher on the natural marsh | Sacco *et al* 1994 |
| Pine Knoll Shores, North Carolina. Marsh planted to control shoreline erosion. Construction date given as 1976. | Sampled in 1986. 3 cm core, sieved at 250 µm<br>Comparison with adjacent natural reference marshes | After 10 years, density slightly higher on created marsh | Sacco *et al* 1994 |
| Craft *et al* give construction date as 1974 | Sampled in 1995 and 1996. 3 cm cores, sieved through 0.25mm mesh. Comparison with nearby natural reference marsh | After 25 years, infaunal density was 70% higher than natural reference marsh. Number of taxa significantly higher on the constructed marsh and some differences in the abundance of individual taxa and functional groups | Craft, Reader *et al* 1997 |
| Dills Creek, Newport River, North Carolina. Marsh planted on excavated upland. | Infauna collected using 4.7 cm cores sieved at 300 and 63 µm.<br>Usage by the fish *Fundulus heteroclitus* assessed using pit traps.<br>Comparison with two adjacent natural marshes, with transects sampled at three tidal levels on each. | Infaunal communities on planted marsh functionally distinct after 3 years. *Fundulus* catches more than a factor of 10 lower on created marsh. | Moy & Levin, 1991 |

| Site and construction method | Nature of invertebrate data | Conclusions | Reference |
|---|---|---|---|
| Snows Cut marsh, North Carolina. Marsh established in 1970 on dredge spoil island | 70.9 $cm^2$ cores, sieved through 1mm mesh. Created marsh compared with nearby reference site | Marked differences in abundance, biomass, species composition and diversity between created and reference marsh after 2 years. Biomass low on planted marsh but higher on bare spoil than in natural marsh. | Cammen 1976 |
| | 3 cm core, sieved at 250 µm<br>Comparison with adjacent natural reference marshes | Densities comparable with natural reference marsh 15 years after creation, but community composition still different | Sacco *et al* 1987, quoted in Moy & Levin 1991; Sacco *et al* 1994 |
| | Sampled in 1995 and 1996. 3 cm cores, sieved through 0.25mm mesh. Comparison with nearby natural reference marsh | After 25 years, infaunal density was 330% higher than natural reference marsh. Number of taxa not significantly different, but some differences in the abundance of individual taxa and functional groups | Craft *et al* 1999 |
| | Sampled in 1998 | | Alphin & Posey 2000 |
| Drum Inlet, North Carolina | 70.9 $cm^2$ cores, sieved through 1mm mesh. Comparison with natural reference marsh | Little difference in abundance, biomass and diversity between created and natural reference marsh after 1 year, but community composition rather different | Cammen 1976 |
| Ashley River and Wappoo Creek, Charleston, South Carolina (disturbance from pipeline construction, rather than habitat creation) | Large invertebrates estimated visually; pit traps used to sample macrofauna; nets to capture natant macrofauna. Comparison between area excavated to construct pipeline and adjacent undisturbed areas. | Several invertebrate species, including *Littorina irrorata* and *Geukensia demissa*, eliminated during construction, with little recovery after 3-4 years. Some differences between treatments in other invertebrate date, although species numbers were similar and number of individuals was higher on disturbed sites in some cases. | Knott *et al* 1997 |
| Winyah Bay, South Carolina. Natural colonisation by *Spartina* of unconfined dredge material placed on intertidal. | 9.1 cm diameter corer sieved at 0.5 mm for smaller macrofauna, larger macrofauna hand sampled from 1 $m^2$ plot. Examined 4 and 8- year old zones. No natural marsh control sites. Qualitative comparisons made with descriptions of fauna from natural marshes elsewhere. | Macrofauna numbers more than 4 times greater in older area than in younger area, and large bodied molluscs occurred only in the older site. | LaSalle *et al* 1991 |
| Winyah Bay cont. | Vegetation and infauna. Infauna sampled with 12 cm diameter cores, sieved at 0.5 mm. No natural marsh reference sites. | Created marsh sites differ, with little sign of convergence of infauna over time. Suggest that fauna takes 4-5 years to stabilise, | Posey *et al* 1997<br>Alphin & Posey 2000 |

| Site and construction method | Nature of invertebrate data | Conclusions | Reference |
|---|---|---|---|
| Sweetwater Marsh, San Diego Bay, California | Fish collected from 10 m length of creek using nets, over eight year period. Compared 4 natural and 4 constructed creeks. | No differences in diversity and density. *Fundulus parvipinnis* occurred at higher densities in constructed channels. | Williams & Zedler 1999 |
| Sweetwater Marsh National Wildlife Refuge, San Diego Bay | Litter bags deployed on the marsh to sample invertebrates. Compared natural and constructed marsh | Densities lower (by a factor of about 3) on created marsh. Most common species occurred in both marshes, but some occurred almost entirely in the natural marsh | Scatolini & Zedler 1996 |
| Paradise Creek Marsh, San Diego Bay, Southern California. Managed realignment, opening new tidal channel to former marsh area separated from saline influence by development (Langis *et al* 1991) | 4.8 cm diameter corer, sieved at 0.3 mm. Comparison between nearby natural reference marsh and four other natural marsh sites. | Macrofaunal densities higher in created marsh after 10 years. Some differences in community composition | Talley & Levin 1999 |
| Northern Wildlife Preserve, Mission Bay, Southern California. Grading to tidal elevations plus creation of channel. | As above | Macrofaunal densities lower in created marsh after 16 months. Some differences in community composition | Talley & Levin 1999 |
| Upper Newport Bay, Southern California. "Created from fill" | As above | Macrofaunal densities higher in created marsh after 6 years. Some differences in community composition | Talley & Levin 1999 |
| Anaheim Bay, Southern California. Artificial island | As above | Macrofaunal densities higher in created marsh after 5 years. Some differences in community composition | Talley & Levin 1999 |

| Site and construction method | Nature of invertebrate data | Conclusions | Reference |
|---|---|---|---|
| Gog-Le-Hi-Te Wetland, Puyallup River Estuary, Tacoma, Washington. Excavated landfill and construction of dyke to create low salinity tidal, with both marsh and mudflat. Equivalent to Lincoln Avenue site of Schreffler *et al* 1992. | Benthic and epibenthic invertebrates sampled twice a year. 0.05 $m^2$ grab or 5.1 cm core sieved at 0.5 mm. No natural reference marsh sites available. Some comparison possible with nearby natural mudflats. | Number of taxa in epifaunal samples approached an asymptote in 3-5 years, but infaunal abundance and taxonomic richness remained relatively depressed over 7 year period. | Simenstad & Thom 1996 |
| Three *Spartina alterniflora* marshes planted on dredge spoil. Galveston Bay, Texas | Mobile and epi-fauna in 2.6 $m^2$ enclosure, animals clinging to vegetation and infauna from 10 cm diameter core, sieved through 0.5 mm mesh. Comparison with three natural reference marshes | After 2 to 5 years, transplanted marshes had lower densities of the shrimps *Palaemonetes pugio* and *Penaeus aztecus*, perhaps because sediment organic matter concentrations were lower. Infaunal diversity slightly lower in transplanted marsh, but no differences in infaunal or fish densities. | Minello & Zimmerman 1992 |
| 10 created *Spartina alterniflora* marshes Galveston Bay, Texas. 7 on dredge material, two planted on natural shoreline and one excavated upland. | Methods as in Minello and Zimmerman, 1992. Compared with 5 natural reference marshes | After 3-15 years *Palaemonetes pugio* densities not significantly different between types, but animals smaller on created marshes. *Palaemonetes vulgaris, Penaeus setiferus, Penaeus aztecus* and *Callinectes sapidus* significantly lower on created marshes. Fish densities also lower on created marshes, but diversities did not differ. These differences may, in part, reflect differences in tidal height, as the created marshes were more variable in elevation than the natural marshes. Some infaunal data, showing slightly higher densities on natural marshes. | Minello & Webb 1997 |
| Sarah's Creek, Gloucester Point, Virginia. Marsh planed on excavated upland | 6 inch benthic grab, samples sieved through 0.5 mm screen. Created site compared with two nearby natural marshes | After 5 years, Little difference between constructed marsh and natural marshes. Constructed marsh used significantly more by *Fundulus heteroclitus* | Havens *et al* 1995 |

| Site and construction method | Nature of invertebrate data | Conclusions | Reference |
|---|---|---|---|
| Barn Island, Connecticut. Tidal flushing restored to previously impounded area | Quantitative data only on the snail *Melampus bidentatus* and the ribbed mussel *Geukensia demissa.* Restored site compared with natural reference site nearby. | *Melampus* at higher density on natural marsh, but snails smaller so difference in biomass is less marked. | Fell 1991; Peck *et al* 1994 |
| Lower Fraser River, British Columbia, Canada. 7 planted marshes and one colonised naturally. | Sites compared with 8 natural reference sites and 5 unvegetated disrupted sites. Invertebrates sampled with epibenthic sled, fitted with 44 µm mesh. | Invertebrate densities on planted sites the same as on reference sites on average, 2-7 years after sampling. | Levings & Nishimura 1997 |

# 5. Success of mitigation and compensation schemes at creating suitable waterbird habitat

## 5.1 Introduction

Despite waterbirds being an important component of coastal ecosystems, few intertidal habitat restoration schemes have specifically targeted this group, unless specialist or endangered species are involved. In the majority of cases, saltmarshes or mudflats are created for economic (eg flood defence) or some unspecific biological purpose. This may be adequate if the goal is to produce an intertidal landscape, but waterbirds and other animals and plants often have very specific ecological requirements which may not be catered for in restored areas. Many of the restoration schemes reviewed here have not supported the full range of functions found on natural marshes and many, as yet, tend to support only a fraction of the waterbird assemblage found in surrounding 'natural' areas. Experience from previous restoration schemes has also indicated that it is necessary to restore habitats with particular species in mind, rather than restore ‘blind’ with the idea that any new intertidal habitat is beneficial – this may be so, but this attitude discourages experimentation, the setting of clear measurable goals against which the success of the scheme can be measured and may not further our understanding of the restoration process.

Evidence from the US also points towards the concept that, although a habitat that will be used by waterbirds can be created, the outcome is not always predictable. For example, the assemblages on restored sites may be less diverse and include a higher proportion of generalist species compared with natural areas. Restoration targeted at specific species or groups may therefore be problematical, as notably indicated by the Light-footed Clapper Rail *Rallus longirostris* example (Box 5.1). If the science of restoration is to be advanced it is essential that an experimental approach is taken and that sites are monitored adequately post-creation.

Where mitigation schemes aim to replace a specific area of habitat, it is extremely important to understand the functions the habitat sustains. Intertidal muds and saltmarsh are extremely variable in terms of the biological resources contained within. Invertebrates and, consequently, their waterbird predators are patchily distributed both spatially (ie across an estuary) and temporally (ie annual and seasonal variations). There are large species differences between species groups. For example, bivalves such as Blue Mussels *Mytilus edulis* tend to form stable beds which remain in the same place for many years. Hence, Oystercatcher distribution on an estuary may vary little from year to year. However the prey species of Redshank, such as *Corophium volutator*, can show very large differences in abundance spatially and temporally. Mitigation schemes should therefore take into account the current functions that the habitat supports through pre-monitoring and aim to provide a replacement habitat that is similar as possible to the habitat that is lost and spans the natural variation found in that habitat.

As very few of the mitigation studies, which are relevant to this study, have had clear goals and targets for success, we have set our own criteria. We review the bird monitoring data from existing saltmarsh and mudflat creation schemes (whether created for mitigation or not) in terms of the speed of colonisation and the waterbird assemblages that use restored sites, and define success depending on whether these differ from those found on natural sites. Where assemblages on restored and natural sites differ, we highlight the reasons why. In the

UK, when making these comparisons it must be borne in mind that the majority of estuaries have undergone land claim in the past and few can be said to have a natural form. For UK sites, we also assess whether the monitoring on restored sites is adequate and suggest a monitoring protocol for future schemes.

## 5.2 Availability of data in the peer-reviewed and grey literature

In Europe, where monitoring for mitigation purposes has not been a statutory requirement, the literature on bird usage of restored sites is very scarce, although several monitoring projects at newly-created sites are in progress in the UK (Tables 1.1, 1.2, 5.1). Detailed analysis of the data obtained from two of these monitoring schemes (Orplands and Tollesbury) have been analysed and appear in Appendix 1 and data from sediment recharge sites are briefly discussed in this chapter. The mudflat created at Seal Sands in 1993 provides the longest-running data set for interpreting trends on a created site. Other bird monitoring schemes, such as those at Porlock Marsh, a natural breach site, or at Havergate, a managed retreat site, are in their infancy and few conclusions can yet be drawn.

**Box 5.1 An example of restoring sites for Clapper Rails *Rallus longirostris***

Zedler (1993) undertook a detailed autecological study of the Clapper Rail to ensure that restored *Spartina* marshes met mitigation targets set for re-establishment of suitable habitat. Comparisons were also made between constructed and natural habitats.

As the rails nest in the intertidal area, it is important that the *Spartina* plants are greater than 60 cm high so that the birds can weave their nests in the canopy. This allows the nest to float upward, but not away, when inundated by the tide. The density of stems is also an important factor.

On restored marshes where the birds were not present, the *Spartina* was insufficiently high and dense. High interannual and spatial variability also highlighted the need for long periods of assessment post-restoration and a large database of reference wetlands.

This problem, caused in part by lower nutrient levels and/or the use of coarse dredge material on restored site has led to further experimentation and manipulation to determine whether fertilisation can increase canopy height. Although this is the case (Boyer & Zedler 1998), it is preferable to use fine sediments, which retain nutrients more than coarse dredge material, in the first instance. The implication is that marshes which have been restored with coarse material will never support Clapper Rails without frequent intervention.

Despite the large number of sites, monitoring at saltmarsh mitigation schemes in the United States concentrates on vegetation and fish populations. Notable exceptions include those which include endangered or specialist saltmarsh species (eg Havens *et al* 1995; Clapper Rail example in Box 5.1). We could find no US examples, which have specifically monitored birds on a created or restored mudflat.

It is difficult, if not impossible, to draw any firm conclusions as to whether previous schemes have been fully successful or not. Most projects do compare restored and reference areas with the implication that restored areas should support densities of waterbirds in the normal range for natural areas and we have used that criterion to determine whether a scheme has been successful or not.

## 5.3 Use of restored or created mudflats by waterbirds

### 5.3.1 Creation of new areas of intertidal mud

The immediate conclusion from studies at Teesmouth (Box 5.2) and the managed retreat sites at Orplands and Tollesbury (see detailed analysis of monitoring data in Appendix 1) is that waterbird assemblages quickly establish on newly-created intertidal habitats. At all three sites, the waterbird assemblage underwent large changes during the first year or two after creation. Changes in subsequent winters were smaller although still occurring at each of the sites five to seven years after creation. The evidence suggests that the waterfowl assemblage is likely to continue to slowly evolve in the future towards an established assemblage.

**Box 5.2 Creation of a new intertidal mudflat at Seal Sands, Teesmouth (Evans *et al* 1998, 2001)**

A created mudflat at Seal Sands provides the best studied example from the UK of how invertebrates and waterfowl colonise a newly created area. The intertidal area was created in 1993. The previously existing, almost freshwater, pool was separated from the main estuary by a porous slag wall which had been constructed in 1970. The pool was originally brackish but by 1993 had become almost impervious to saltwater incursion due to limited infilling with soil and clay. The aim of the project was to create a supplementary feeding area by replacing the pool with intertidal mud. This was undertaken by draining the pool and digging a central channel at the lowest tide level, sloping upwards towards the sides. This design allowed a maximum amount of edge to be exposed at all states of tide. The freshwater was pumped out and works took place to create the area in 1993. The tide was allowed in via a pipe and sluice. However, the pipe diameter was insufficient to allow full tidal inundation and tidal range was restricted and further engineering works were necessary during 1994. This delay actually may have been beneficial to shorebirds using the area as it meant that low- to mid- tidal level mud was exposed for longer compared with the outside estuary.

**Invertebrates**

Invertebrate colonisation was rapid and the benthic fauna has developed since creation. *Hydrobia* colonised in 1994 and slowly increased, possibly due to the low organic content in the sediment, to reach very high densities by 1999 in areas of algal cover. The highest densities of *Hydrobia* are found on fine-grained, organically enriched sediments with high bacterial populations (Newell 1965). *Nereis* colonised in 1995 and were abundant in the site from autumn 1996 onwards. However, *Corophium* failed to colonise until 1996 due to poor over-winter survival; sediment compaction during the creation phase may have been the cause of the poor survival.

**Birds**

As expected, most bird usage took place during the state of tide when Seal Sands were covered by the tide. Redshank, in particular, but also other species, used the area during passage periods when energetic demands are high. Curlew and Redshank also roost in large numbers on the site. The main conclusions were:

- Bird usage was highest during periods when intertidal flats are covered, two to three hours before high tide.
- Redshank usage has been consistently high. Other species use the area during migration periods.
- During periods when Seal Sands were covered, higher densities of Redshank and Dunlin and similar densities of Curlew were found in the created area compared with Seal Sands.
- Despite abundant prey, Grey Plover and Bar-tailed Godwit rarely used the area; Knot were absent, as were Dunlin and Ringed Plover except during passage periods.
- The reasons for some species not using the site may have been due to the enclosed nature of the site which may have an increased perceived predation risk.
- For the successful recolonisation of three of the main prey species, *Corophium, Nereis* and *Hydrobia*, requires a lead in time. Evans (1998) suggested at least three years but additional monitoring data indicated that further changes in the bird assemblage have occurred since the first report. Although invertebrate densities take three or more years to colonise newly-created areas, this study indicates that if the created mudflat is large and open enough, new areas can support densities of feeding birds as high as those on natural intertidal mudflats.

### 5.3.2 Speed of development and differences between waterbird assemblages on created and natural areas of intertidal mud

Although some shorebirds and wildfowl used all three sites, the assemblages using the created habitats were different to those using the surrounding natural areas. The reasons for this included a combination of a less diverse range of sediment types compared with the surrounding estuary and, at Seal Sands, a high degree of enclosure, which prevented some species which prefer open coasts from fully exploiting the area.

At the low-lying Orplands A site, the species assemblage was similar to that in the surrounding mudflats after only two years, but showed less inter-annual variation indicating that a component of the 'natural' assemblage was missing. Species such as Oystercatcher and Turnstone, present on the larger Blackwater Estuary, were also missing. At Seal Sands, however, even after seven years the waterfowl assemblages were different to the surrounding estuary. At Tollesbury, the assemblage was also different to the estuary as a whole both in terms of species and the timing of usage. For example large numbers of Redshank use the Blackwater estuary from early autumn onwards but do not use the retreat site until mid-late winter, a time when food supplies may be low. This suggests that the retreat area may not be a preferred foraging area but used only when food supplies are low elsewhere. However, the lack of monitoring on control sites makes it difficult to determine the exact cause for these differences. Waterfowl assemblages vary both between and within individual estuaries and so the comparison of the whole estuary versus an individual part of that estuary may not be valid.

The speed of development of the waterbird assemblage at the two Essex sites followed that predicted from the changes in benthic fauna. At Orplands and Tollesbury, the build up of fine muddy sediments led to the assemblage being initially dominated by Redshank, Dunlin and Grey Plover, which feed predominantly on *Hydrobia* and *Nereis.* These were the two main invertebrate species to colonise the sites during the first few years following the breach. Knot were initially absent from both Tollesbury and Orplands but their colonisation coincided with the appearance of *Macoma* on the sites, a preferred prey item, approximately three years after creation. Species such as Oystercatcher and Turnstone, which feed on molluscs and crustaceans, were very scarce in the sites during winter although common on the surrounding estuary. In the short-term, it is unlikely that the retreat sites will develop in such a way that populations of these two species would be supported. The large molluscs, preferred by Oystercatchers, were virtually absent from the Orplands sites at least five years after the breach and it may take some years for these to colonise, if they ever do. Experimental introduction of these species may speed up the colonisation process.

At Teesmouth, species such as Grey Plover, Bar-tailed Godwit, Dunlin, Knot and Ringed Plover were rare or absent, despite abundant prey populations in the created site. The degree of enclosure of the site was thought to be responsible. This highlights the importance that perceived predation risk can play in determining where birds feed. Cresswell (1994) showed the habitat choice of Redshank on a Scottish estuary was a balance between food profitability and predation. Redshank predation by Sparrowhawks was over four times higher on a saltmarsh area (where food densities were higher), compared with an adjacent intertidal mudflat. Juveniles were excluded by adults from feeding on the intertidal area and fed on the saltmarsh. They experienced 95% winter mortality compared with 11% for adults. This highlights that the design of any mitigation sites may affect predation rates.

### 5.3.3 Differences in seasonal use of natural and created mudflats

The monthly pattern of use of the three created sites has tended to be different to that of the surrounding estuary. At Tollesbury, the site for which the best data are available, Oystercatchers tended to use the site only in spring and summer (albeit in very small numbers), whereas numbers on the whole estuary were highest in winter. Redshank numbers were highest in the retreat site during the December to February period whereas, on the estuary as a whole, numbers were approximately stable from October to March. Knot usage was similar on both the retreat site and the estuary as a whole. These patterns indicate that some species either did not use the site in winter, partially used it or used it in a similar pattern to the estuary as a whole. At Seal Sands, the created mudflat was used more during migration periods, when the energetic demands of the waterbirds were likely to be higher.

The reasons for the different patterns of usage between species are unknown, but the pattern shown by Redshank is typical of a species that makes use of a site as a supplementary feeding area during times when food resources are likely to be at their lowest and energetic demands are high. This highlights the benefit of creating habitats that extend feeding time once the mudflats have been covered.

### 5.3.4 Can man-made mudflats support ‘natural’ populations of waterfowl?

Although a small sample, these three areas highlight several important points.

First, although each were largely successful in creating a habitat that is used by some intertidal feeding waterfowl, this was only possible as geomorphic conditions in the areas were suitable for the accretion of fine muddy sediments which were colonised by waterbird prey. Had this not been the case, colonisation of the underlying over-consolidated sediments by benthic invertebrates would have taken place at a much slower rate.

Second, waterbird assemblages on the created sites are less diverse in terms of species or show less variation compared with the adjacent estuary. They will therefore only be capable of supporting a fraction of the estuary assemblage. This therefore implies that, unless a large amount of effort is put into engineering the creation scheme so that it produces more diverse habitats, it may not achieve success in replacing habitat typical of that being lost, and hence the assemblage that was lost.

Third, some other physical factor such as the degree of enclosure or disturbance reduces the usage of the site that may be expected due to the food supply available. We have not addressed these important issues directly in this review but the design of a newly created area is an important consideration during the planning stage.

Fourth, for reasons we do not fully understand, the seasonal pattern of usage may vary between created sites and natural sites as well as between species.

Some of the above issues can be overcome if created mudflat sites are placed in areas with suitable geomorphic conditions and with careful engineering to ensure a diversity of habitats. These studies suggest that it is possible to create mudflats for waterfowl, but a lead-in time of at least five and probably 10 years would be needed.

#### 5.3.5 Sites at which sediment recharge has taken place

Recharge of existing sediments has been practised at many sites in south-east England and bird monitoring has been carried out at some of these sites. The monitoring took place from summer 1998 at approximately six month intervals. Each site was not covered by each survey which makes detailed comparisons extremely difficult.

Several of the recharge sites created from 1998 onwards (Wallasea Ness, Tollesbury Wick, Packing Marsh, Pewet Island and Cobb Marsh) consist of sediments of sand grade or coarser. Trimley, North Shotley and Horsey Island marshes have, however, been constructed with significant amounts of silts and clays and are more similar to the local natural marshes.

The sites, which have been recharged with coarse sediment, have been extremely successful in attracting large numbers of breeding Little Terns, Oystercatcher and Ringed Plover (Environment Agency, unpublished data). During 1999, two sites in the Blackwater supported approximately 100 pairs of Little Tern. During the winter, these sites were also used as roosting sites by Oystercatchers, Cormorants and other waterfowl but very few feeding birds were observed.

The sites were only visited once per winter, at different states of tide and some sites were not visited in some years. Since the sites have been created, two winters' data has been collected during visits in October/November 1998 and December 1999 at various times between 3.25 hours before and 2.5 hours after high water. Surveys also varied in length and it is thus impossible to draw any firm conclusions from these monitoring data as to the value of these sites for wintering birds. The sites which have been recharged with fine sediments have been used by waterfowl to some extent. Brent Geese were observed at the Shotely site in 1999 and small numbers of shorebirds were observed, including Ringed Plovers, Oystercatchers, Grey Plovers, Dunlin, Redshanks and Turnstone. Only two shorebirds (Grey Plover and Oystercatcher) and no geese used the site in October 1998. Unfortunately, Horsey Island was not counted in 1999 and so no comparison with 1998 is possible, when 15 shorebirds of five species were observed. It is far too early to draw any conclusions from these studies as most sites are less than three years old.

Although not specifically a recharge scheme, the evidence from the Duddon pipeline construction indicates that disturbance of the intertidal sediments reduced invertebrate densities and diversity which led to a reduction in shorebird numbers. However, both invertebrates and birds recovered and within one to two years populations of shorebirds on sandy and muddy shores were similar to pre-construction numbers. However, the removal of stones and their associated mussels led to a reduction in the number of Oystercatchers on the rocky shore which did not recover. Added to this, invertebrates at some sample points did not revert back to pre-construction assemblages and reached alternative stable states, thus highlighting again that restoration trajectories may not be predictable (Tittley 1997).

## 5.4 Usage of restored saltmarsh habitat by waterbirds

In contrast to mudflats, most studies investigating the use of created saltmarshes by birds are from the United States, with a few examples from Europe. This is for several reasons. First, in the US, there is an emphasis on the productivity and economic value of saltmarshes, rather than for the species they contain as is often the case in Europe. US legislation also requires wetlands to be restored or replaced if areas are lost, thus spawning an industry in the successful recreation of saltmarsh habitats and mitigation banking. Marshes in the US, unlike Europe, support bird species that are solely reliant on saltmarshes, several of which have a threatened status. These US studies, although not directly applicable to European systems, are useful in that they illustrate reasons why bird assemblages may differ between restored and natural marshes.

In Europe, saltmarshes support internationally important populations of grazing waterfowl although they are not solely dependent on it. In addition saltmarshes provide important roosting area for waterfowl, support large numbers of breeding Redshank and Oystercatcher and are also important for wintering passerines such as Twite, Snow Bunting and Shore Lark.

The US studies have concluded that, in terms of bird usage, functional equivalence of man-made marshes with natural marshes may or may not occur. In most cases macrofauna (including birds) colonise quickly and the assemblage reaches maturity in a short space of time, often less than three years (eg Simenstad & Thom 1996). At the Gog-le-hi-te wetland, the taxa richness of epibenthic organisms, fishes and density of fishes all approached asymptotic trajectories (ie stability) within three to five years of restoration, but the numbers of birds using the site continued to increase over the seven year duration of the study. Despite the rapid responses by fish, invertebrates and birds apparently contradicting arguments that the restoration, creation and enhancement of estuarine marshes is problematic and proceeds by trial and error (Zedler 1988; Moy & Levin 1991), measurements of other ecological functions have indicated that the wetland was in an early stage of maturity. Few predictable trajectories of community development were seen and few indicated system maturity. For example, the organic content, chlorophyll/phaeophytin pigments and the infauna taxa richness and density increased slowly or remained relatively depressed over the same three to five years of monitoring. *Carex* production showed a gradual progression towards reference marsh levels. Simenstad and Thom (1996) also point out that many ‘functional trajectories’ are unpredictable and, due to the short-term nature of monitoring projects, it is impossible to understand why trajectories either converge, diverge, fluctuate or achieve an alternative stable state to reference areas.

Differences in habitat are often cited as reasons why bird assemblages are different between restored and reference marshes. Havens *et al* (1995) showed that a constructed marsh in Sarah's Creek in Virginia supported far fewer Marsh Wrens *Cistothorus palustris* than natural marshes as the band of *Spartina* on the restored marsh was too thin. The stem density and height of *Spartina* on some restored marshes in southern California was unsuitable for Clapper Rails (Zedler 1993). Also, at Sarah's Creek, the higher length of open water/marsh interface in restored sites caused a higher usage by shorebirds whereas the lack of a mature salt bush community (*Iva frutescens* and *Baccharis halmifolia*) led to a lower usage of the restored marsh by passerines.

One frequent difference between restored and natural marshes in both the UK and US, is the consolidated nature of the sediments, lack of natural creek systems, smooth topography and poor drainage of restored and created systems. In Galveston, Texas, species richness and diversity was higher in the natural marshes due to the presence of migratory waterfowl, wintering shorebirds and saltmarsh specialists such as rails and marsh sparrows (Melvin & Webb 1998). The assemblage on constructed sites was dominated by gulls and terns, which nested on the unvegetated berms surrounding these sites. The main conclusion from this study was that created saltmarshes provided bird habitat, however not necessarily for the same species assemblage as natural saltmarshes. The reasons for the differences were thought to be due to the nature of the sites. All of the created marshes were on smooth, gently sloping shorelines exposed to wave action and contained flat monocultures of *Spartina* with few openings. Ponds and tidal flats were rare. Natural marshes tended to have more marsh edge and open water. Melvin and Webb postulate that created marshes supported fewer shorebirds and rails because they were at overall higher elevation, had less edge habitat, deep and steep-sided channels and taller and denser *Spartina*. Peaks and troughs in bird abundance on natural saltmarshes were strongly related to seasonal migration chronology, whereas those in restored areas did not. This indicated that natural marshes provided habitat that was not available in nearby created saltmarshes.

Similar problems occur on some of the naturally-regenerated marshes in south-east England (Table 1.2). The English population of Twite *Carduelis flavirostris* winters exclusively on saltmarshes and feeds on some restored saltmarshes. Atkinson (1998) compared usage of naturally restored saltmarshes by Twite with surrounding areas. Twite feed on the seeds of *Salicornia* spp. and the pioneer communities dominated by this species were absent from many of the restored marshes. The sites tended to be flat, highly dissected and poorly drained, except around creek edges. Consequently, the vegetation communities were dominated by a rank mix of *Puccinellia maritima* and *Halimione portulacoides*. The restored sites supported different, less diverse, vegetation communities than the surrounding natural marshes even though some of the restored sites were more than 100 years old. It is unlikely that they would ever reach a state where they would be colonised by Twite.

In contrast to the above studies, Brawley *et al* (1998) conclude that a site at the Barn Island Wildlife Management Area in Connecticut is reaching equivalence with surrounding natural marsh. Tidal exchange was restored in 1978 to an area known as Impoundment One and summer bird usage was monitored in 1993 and 1994. Species diversity and abundance were high in Impoundment One, the restored site, and large populations of two marsh specialists were observed. In comparison with 16 other estuarine areas, Impoundment One was ranked second in terms of avian abundance and diversity. Restored areas may therefore not always support poorer avian communities.

## 5.5 Implications for the monitoring of new habitat creation schemes

The monitoring studies described here encompass a broad quality spectrum ranging from those from which only general observations can be made, to those that allowed detailed statistical analysis. In some cases, pair-wise comparisons were made with surrounding natural habitats but, in many cases, lacked the statistical power to determine differences. For future monitoring schemes, it is important not to fall into the trap of 'mindless monitoring', a term used by Joy Zedler. Monitoring should be used as a means to determine whether certain criteria have been met and not because it is deemed necessary to be seen to be counting something.

At mitigation sites, there is an emphasis on recreating areas which are indistinguishable from natural areas. Many such projects take years to develop and, as such, permit several years of pre-monitoring to take place. This is important for any well-designed scientific study as it allows the abundance and annual variation of waterbirds using the area to be estimated before habitat loss and indicates the key features of the site which are likely to be lost (eg any feeding or roosting areas). However, further general studies on feeding birds and their relationship with their food resources are also desirable as this improves the ability to (a) specifically identify the effects of habitat change on the birds usage of the area and (b) provide a benchmark against which the success or failure of the scheme can be judged.

Pre-monitoring also allows protocols to be developed on the basis of the statistical power they have to detect differences. Statistical power is an essential tool for mitigation schemes. In basic terms, it determines the probability of finding a difference between two sets of samples based on the variation in the two samples and the sample size. This type of analysis allows an optimum sampling strategy to be developed. Poorly designed studies would not be able to detect real differences.

Assessments of whether restored and natural sites are comparable often include pair-wise comparisons between the restored, created or enhanced wetland with surrounding reference areas (Simenstad & Thom 1996). These monitoring studies tend to be short-term and concentrate on the structure of communities (eg numbers of particular species) rather than long-term studies of processes. The short-term nature of monitoring, typically less than five years, seriously hampers our understanding of how wetlands can be restored effectively for the long-term (Zedler 1988). As shown in previous chapters, easily measurable criteria such as the density or species composition of plants and animals can reach near-comparable status to reference marshes after only three years but complex ecological interactions do not necessarily follow that time scale. These arguments are further developed in Chapters 7 and 8 but most monitoring schemes are usually inadequate to identify an endpoint of ecosystem or community maturity (eg the restoration of trophic linkages between benthic prey and fish utilising the marsh reported by Moy & Levin 1991).