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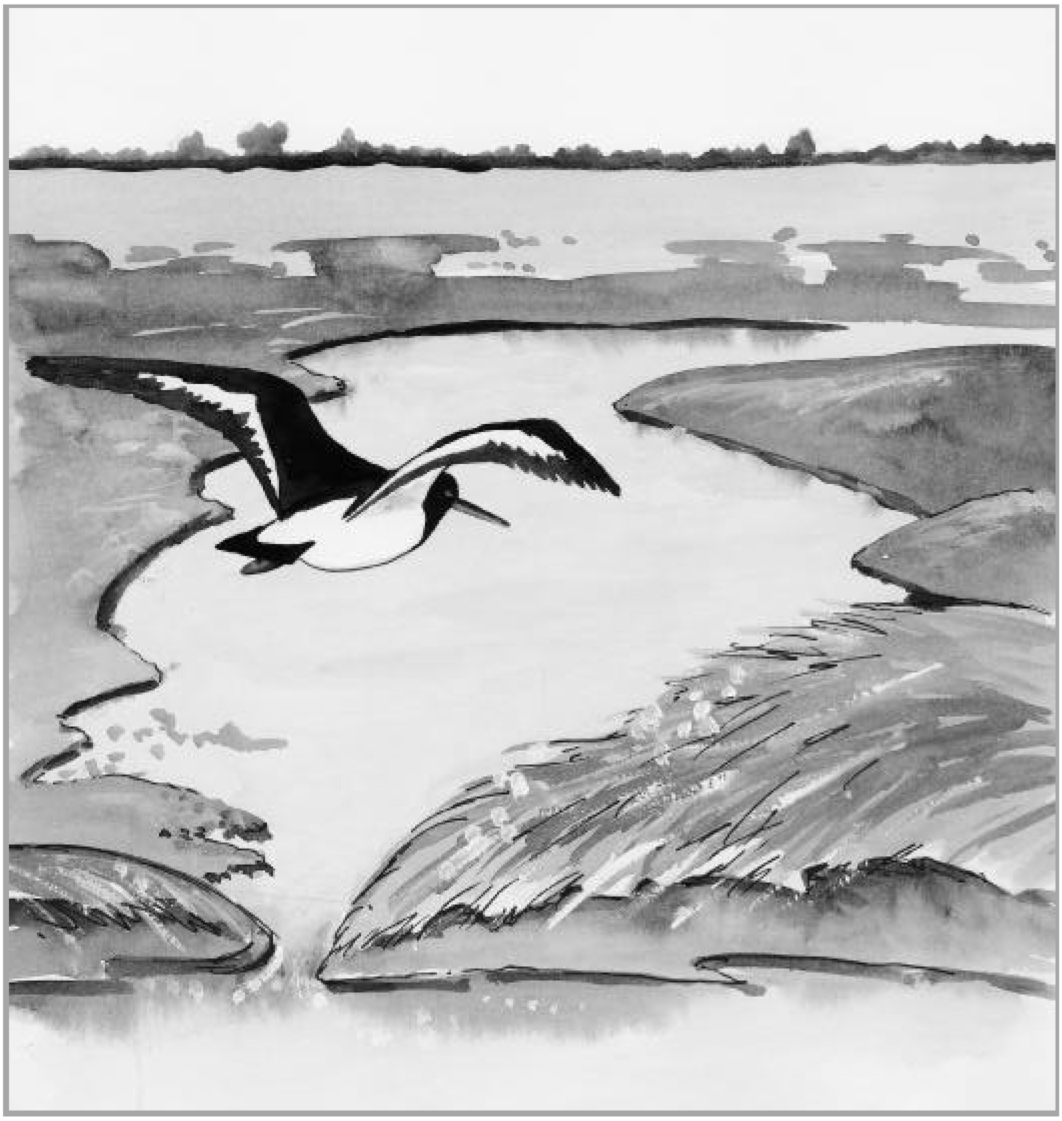

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English Nature Research Reports

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The success of creation and restoration schemes in producing intertidal habitat suitable for waterbirds

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Executive Summary

1. One of the greatest threats facing coastal waterbirds in the United Kingdom is the loss or degradation of coastal habitats through development or, in the medium to long term, loss of habitat due to climatic change and sea-level rise.

2. Although estimates of the impacts of habitat loss on waterbird populations may be difficult to predict exactly because of the role of density-dependent factors, habitat loss or change has been shown to impact locally by reducing both the abundance of waterbirds using a site and also at the population level by changing mortality and productivity rates. Predictive behaviour-based mathematical models, which are built around understanding of a species relationship with its environment have successfully predicted the population changes observed in the field and also allow the impact of novel situations on changes in population size to be assessed.

3. Under European law, appropriate compensation must be provided when a Natura 2000 site is adversely affected by development. Provision of compensation involves the creation or restoration of habitat of environmental value at least equivalent to that of the displaced habitat.

4. Intertidal habitats pose special problems for restoration because (i) they are topographically and ecologically complex, (ii) they support many species of animals, some of which require specific habitats and linkages to other terrestrial or marine habitats, and (iii) they exist and evolve within dynamic coastal settings, subject to changing tidal levels, salinities and long term forcing factors associated with sea-level rise and climate change.

5. Currently within the UK, the science of coastal habitat creation and restoration is poorly understood. This report collates information from the literature from around the world on projects where new intertidal habitats have been created or restored through the use of managed retreat (setting back of sea walls), sediment recharge and enhanced sedimentation. Specifically, it details the methods used and the stability and long-term geomorphological sustainability of the new habitats and the time scales at which marine invertebrate and waterbird populations colonise and then develop in the new habitats. As very few projects detail specific success criteria or perform adequate monitoring we provide guidelines as to how success may be measured and also how monitoring programmes for invertebrates and birds may be carried out.

6. There are few examples of newly restored intertidal habitat in the UK. Consequently there is limited monitoring data from which to draw conclusions on the success of domestic restoration actions. There are, however, a number of examples of historic natural breaches in flood defence and unmanaged restoration on intertidal habitat from which lessons can be drawn. There are also a number of managed restoration actions in other nations but caution must be exercised in drawing direct comparisons with these because of ecological and physical differences between coastal settings.

7. When restoring habitats, it is necessary to take an approach based around restoring ecological functions rather than concentrating upon individual attributes. From the available literature it is clear that, given suitable hydrodynamic and geomorphological

conditions, it is possible to recreate some types of of mudflat and saltmarsh within a relatively short time period (less than five years), though the exact form and function of "mature" restored habitat is near impossible to predict from the outset. The functions supported by estuaries and intertidal habitats often develop at different rates. Vegetation, invertebrate and bird fauna often respond relatively quickly (within a few years) whereas geo-chemical cycling and the restoration of nutrient flow between terrestrial and marine habitats take longer. Experience from the United States indicates that the outcome is often uncertain and unpredictable at the time of restoration and therefore any overall strategy for reaching restoration targets must be thought of in terms of risk at spatial and temporal levels.

8. The reestablishment of intertidal habitat involves the landward relocation of flood defences, the breaching or removal of former outer defences or measures to enhance sedimentation on the foreshore. No one technique is applicable in all coastal situations and each comes with pro and cons. Realignment involving breaching of outer defences enables the reestablishment of intertidal habitat within sheltered conditions but does not address larger estuarine morphological concerns. Bank retreat, the full landward retreat of outer flood defences, enables a more natural estuarine form to develop but may leave the site vulnerable to wave erosion. Enhanced sedimentation within wood fenced containment fields have been tried in a number of open foreshore settings but have been found to have beneficial outcomes only when natural conditions were naturally favourable in the first place.

9. The placement of dredged material either within managed realignment sites or on the open foreshore offers the opportunity to raise the intertidal surface elevation to a level suitable for mudflat or saltmarsh development. There are concerns that this approach does not allow the form of created intertidal habitat to develop features, such as creeks, which would normally develop as a marsh evolves slowly from a mudflat. Early attempts to place dredged material in the intertidal zones often result in the over-consolidation of sediments and poor creek development which, although useful for coastal and flood defence, are not suitable for the restoration of naturally diverse plant and invertebrate communities. New techniques, being developed in the south-east of England involve the placement of high-density slurry to create a marsh (or mudflat) surface of variable topography. Preliminary results indicate that the slurry may adopt the form of the underlying marsh surface and may encourage the development of a proto-creek network. If this were to persist, this would be a major step forward in saltmarsh restoration. Detailed research has yet to be undertaken.

10. Invertebrate communities will colonise suitable intertidal habitats if a source of potential colonisers is available. The rate of colonisation depends on the availability of source colonisers and the life-cycle of individual organisms. Species that are mobile, have a short generation time and a planktonic larval phase are likely to establish relatively quickly. However, bivalve species such as *Mya* spp and *Scrobicularia* spp may take a few years to colonise and grow to a size that are suitable for waterbirds. Evidence from empirical studies shows that although invertebrates colonise relatively quickly, species composition may be different from surrounding areas, even 10-15 years after colonisation.

11. Birds are mobile and quickly adapt to new habitats. Empirical studies show that colonisation is rapid and at Orplands and Tollesbury Wick, two managed retreat sites

in the UK, there was a high degree of bird usage within the first two years although there were differences in the species composition and temporal usage of the sites compared with the surrounding estuary. In other studies, differences between created and natural sites have manifested themselves with a preponderance of generalist, rather than specialist, species on restored sites. All but one of the studies reported here considered that the avifauna supported on restored areas was different to surrounding natural areas. This implies, that in many cases, created intertidal habitat is not supporting the full range of functions found in 'natural' habitats.

12. In the majority of studies reported here, the design of monitoring schemes and the definition of success criteria have been inadequate to determine whether a created or restored wetland has reached its intended target. Monitoring and assessment is an important component in the mitigation/compensation process and, within the UK, there are no agreed protocols for intertidal habitat restoration. In developing such protocols, mechanisms to account for functioning at the wider landscape level (i.e. linkages between the habitat on site and those elsewhere in the coastal area), beyond individual site specific and compliance issues should be sought.

13. There are large gaps in the knowledge about intertidal habitat restoration in the UK. These include the efficacy of the methods used to create areas, how to measure functional equivalence in a manner that is rapid and cost-effective and also the human-use values that are put on intertidal habitats. An experimental approach and an adaptable management framework, with regular assessment of the monitoring data, is essential for any large-scale compensatory project.

14. For compensation projects, it may be wise to demand that habitat to be lost is recreated in advance of that loss. In this way the ‘value’ of the restored or created habitat may be directly compared with the natural habitat to be lost and measures to ensure equitable replacement can be undertaken.

15. The creation of new habitats as part of compensation for damage to Natura 2000 sites provides the opportunity to recreate historically lost habitats. Many UK flood plains have undergone large-scale development and freshwater-transitional and brackish-water habitats have largely been removed from these areas. Reinstatement of these habitats will improve the linkages between terrestrial and marine habitats and is likely to improve the likelihood of success of compensatory measures.

1. Introduction

In the United Kingdom, increasing industrial and recreational usage of coastal areas has led to loss or degradation of intertidal habitat. These habitats, which include mudflats, saltmarshes and brackish coastal lagoons are amongst the most important in the UK in terms of conservation and support large concentrations of breeding, passage and wintering waterfowl. Intertidal habitats are also of economic importance for flood defence and because they support a range of estuarine functions of economic significance, such as spawning or nursery areas for fish, wildfowling, recreation, grazing of livestock and general tourism interest.

Although the net trend around the world has been towards destruction of wetlands, significant efforts are now being focused on the voluntary restoration and creation of these habitats. Part of the interest in wetland creation and restoration stems from the fact a major part of the wetland resource and their associated ecological and social values have been lost (Box 1.1). There are also legal obligations set by Government policies to address past and ongoing loss of these valuable habitats. In the United States 'no net loss' of all wetlands was set as a national goal in 1990 under the Bush Senior administration. Under this approach it was not anticipated that there would be a complete cessation of wetland loss but that any wetland displaced for economic and political reasons would be replaced by a created or restored wetland of equal environmental value. In Europe, the Directive on the Conservation of Natural Habitats and of Wild Fauna and Flora (92/43/EEC) (known as the Habitats Directive, DG XII 2000) has initiated a similar though not identical approach. Under this Directive, all habitats, including wetlands, designated within Natura 2000 sites are provided with a high level of protection. Where an overriding public interest case is made habitats within this network of sites may be developed, but habitats of equal value have to be recreated as compensation. Both approaches have chosen to recognise ecosystem 'functions' as the baseline currency by which wetland values and the success of restoration efforts should be measured.

In the United Kingdom and European Union, there are currently no formal guidelines for best practice in the restoration or creation of new habitats which specifically relate to waterbirds, or recommendations for determining the success of the mitigation/compensation process. Despite a large body of work in the United States, the science of intertidal habitat creation or restoration is poorly developed in North-west Europe. This review, which concentrates on the creation or restoration of habitats specifically for waterbird conservation, summarises information collected about sediments, invertebrates and birds from creation or restoration schemes in NW Europe and, where relevant, draws upon the experience from the United States. First, we summarise the theoretical and actual effects of intertidal habitat loss on bird populations and then detail the methods used, the persistence of new habitats and the speed of colonisation and long-term trends in invertebrate and bird numbers. Finally we discuss how to determine the success of mitigation or compensation schemes, that are specifically related to the creation of new habitats for waterbirds.

Within the UK, the science of intertidal habitat creation is in its infancy and there are few examples of comprehensive monitoring of sites that have been created or restored (Table 1.1). As there are so few examples of intentional retreat, sites at which intertidal habitat has been created accidentally due to unintentional breaches in sea walls as a result of neglect or storm damage, can be valuable analogues (Table 1.2). Sites where such 'unmanaged retreat' has occurred are termed ‘naturally regenerated marshes’. These tend to be more varied in size, age and other characteristics than sites where intertidal habitat has been intentionally created. They provide useful ‘natural’ experiments and inferences about the restoration process can be drawn from these sites.

Box 1.1 Ecosystem functions and values to human society of saltmarsh and intertidal flats (after Short *et al* 2000)

Function	Value
Primary production	Support of food webs, fisheries and wildlife
Canopy structure	Habitat, refuge, nursery and settlement; support of fisheries
Organic matter accumulation	Support of food webs, counter sea-level rise, sequestration of carbon
Seed production / vegetative expansion	Maintenance of plant community and biodiversity
Sediment filtration and trapping	Counter sea level rise, improve water quality and support fisheries
Epibenthic and benthic production	Support of food web, fisheries and wildlife
Nutrient and contaminant filtration	Improvement of water quality, removal of pathogens and support of fisheries
Nutrient regeneration and recycling	Support of primary production and fisheries
Organic export	Support of estuarine, offshore food web and fisheries
Wave and current energy dampening	Protect backmarsh areas (and sea defences, if present) from erosion and flood
Self-sustaining ecosystem	Recreation, aesthetics, open space, education, landscape level biodiversity and historical value

1.1 Scope of the review

1.1.1 This report collates information from the literature on projects where new intertidal habitats have been created or where existing intertidal habitats have been enhanced. We specifically examine:

- The success or otherwise of the creation of new intertidal habitats in previously terrestrial locations (with particular reference to creeks).
- The success or otherwise of the enhancement of existing mudflats through recharge, in particular by direct capping with marine sediments.
- Details of materials, construction techniques and time scales.
- Details of the stability of sediments and long-term geomorphologic sustainability of newly created and recharged habitats.
- Time taken for re-colonisation by intertidal invertebrates and information on changes in their species composition and abundance on created/enhanced intertidal habitats as they develop.
- Time taken for the establishment of feeding bird assemblages on recently created or enhanced intertidal habitats and information on their densities and species composition, with comparative information from adjacent control areas of similar size and nature.

1.1.2 To the extent to which the available information allow, we evaluate the success of replacement or enhanced habitats in accommodating displaced birds in cases where intertidal habitat creation or enhancement has been undertaken specifically to provide compensation for the loss of bird foraging areas.

1.1.3 We have also collated from published and unpublished sources, any pre- and post-development bird monitoring data that can be used to quantify the likely consequences of intertidal habitat loss in terms of changes to bird numbers present on affected sites, where possible including several years of pre-development baseline information followed by a run of years of detailed bird counts during and following construction works.

1.1.4 On the basis of this information we seek to provide best practice guidelines for future works including the development of a sound waterfowl monitoring strategy.

1.2 Previous and current studies

After some three decades of restoration efforts in the United States, there is a growing body of literature reviewing the creation and restoration of wetlands. These began with general publications in the early 1990s (Kuster & Kentula 1990; Marble 1992; Kentula *et al* 1992; National Research Council 1992). A number of reviews specific to certain wetland types have been undertaken. Of most relevance to this study are those that have concentrated on efforts to create and restore tidal wetlands, eg Broome *et al* (1988); Zedler (1988); Broome (1990); Shisler (1990); Thayer (1990); Zedler (1996) and Weinstein & Kreeger (2000). A recent special issue of *Wetlands Ecology and Management* (Streever 2000) focused upon the beneficial use of dredge material for the restoration of US saltmarshes and mudflats. A special issue of the journal *Restoration Ecology* focuses on international experience of saltmarsh restoration by dike breaching (Simonstad 2001). Unvegetated mudflats are not classed as wetlands under S.404 of the US Clean Water Act and thus intertidal habitat creation work in the US has focussed on the creation of saltmarshes (Posford Duvivier Environment 1991). As a consequence, mudflat creation schemes in the US have usually been motivated by a desire to dispose of dredged material rather than by nature conservation concerns.

Despite the abundance of literature and expertise in saltmarsh creation in the US, extreme caution should be used in using the results of overseas studies to predict the likely success of mitigation schemes in the UK as there are fundamental differences in the character of intertidal habitats between the two regions. The UK coastline has a rich diversity of intertidal habitats characterised by sediments that vary in texture from sands and gravels through to cohesive mudflats. Tidal ranges are moderate to large (3-12 m average tidal range) and sediments are largely derived from sedimentary rock (clastic). Vast areas of coastal wetlands in the US have been recharged with sediments but these systems have much lower tidal ranges (often less than 2 m) and saltmarsh sediments often consist largely of peat rather than clastic sediments, making it difficult to extrapolate to the very different conditions found in most regions of the UK.

Within the United Kingdom, there is little published in the specifically biological or ecological peer-reviewed literature concerning intertidal habitat creation and restoration. The creation of a mudflat at Seal Sands is an obvious exception (Evans *et al* 1997), as are analyses on the role of benthic invertebrates, particularly *Nereis* spp. and *Corophium* *volutator,* on the establishment of saltmarsh plants (Emerson 2000; Hughes 1999), and the role of micro-algal biofilms in stabilising sediments (Underwood 1997). There have however been a number of papers concerning non-biological processes on some of the UK managed retreat sites including geochemical changes (Orplands and the Blackwater - Macleod *et al* 1997; Emerson *et al* 1997, 1999, 2000; Humber - Jickells *et al* 2000), tidal exchange (Orplands - Emerson 1997), persistence of saltmarsh in unmanaged retreat sites (south-east UK - Burd 1994; Medway Estuary - French 1999; Essex – Crooks & Pye 2000) and policy related to managed retreat (Wash - Nunn 2000; Cley - Klein & Bateman 1998; generally – Crooks & Turner 1999; Ledoux *et al* 2000; Crooks & Ledoux 2000). It is perhaps not surprising that little has been published in the peer-reviewed literature on the biological aspects as sites at which habitat creation or restoration has been practised in the UK are generally less than five years old (Table 1.1).

Much of the biological information on the UK sites is held in the grey literature published by organisations such as the Environment Agency, English Nature and the National Trust. Table 1.1-1.3 summarises the sites in the UK where such work has been carried out and the processes and taxa that have been monitored. The major motivation of this work has been coastal protection and, because of the value of saltmarshes in dissipating wave energy, the emphasis has frequently been on the creation of saltmarshes rather than mudflats.

Within north-west Europe there are large areas of man-made marshes and mud flats. Within the Netherlands, there are over 17,000 ha of man-made saltmarshes, although these were created specifically for flood defence purposes rather than for any environmental benefit (Esselink 1998). This policy is changing and saltmarshes on the North Sea coasts of Germany, Belgium, the Netherlands and Denmark, which are of high conservation importance because of the large concentrations of wintering, passage and breeding waterfowl that they support, are now increasingly being managed for nature conservation purposes (Esselink 2000). Again little has been published in the peer-reviewed literature although the Sieperdaschor in the Netherlands is a notable exception (Stikvoort 2000, Castelijns *et al* 1997).

1.3 How is the success of creation or restoration schemes gauged?

What constitutes success in creating new intertidal habitat? We return to this question in more detail in Chapter 7 and almost inevitably the question will end up being tested in the courts at some point in the future. But it is helpful to examine the key elements of the question at this stage. A completely successful created habitat would be indistinguishable in all respects from the corresponding natural habitats. Intertidal habitats are often very dynamic in their characteristics, so persistence does not require complete immutability. In addition to this, a successfully created habitat will have biological, physical and chemical characteristics that are within the range as those characteristics found at equivalent natural habitats. Bird populations using the new habitat will be within the normal range for such habitats. Invertebrate populations in sediments, fish and crustacean populations moving into the area at high tide and plants growing on any saltmarsh habitats will show normal abundance and diversity. The landscape features of the area, such as saltmarsh creek and pan patterns, will be indistinguishable from those in natural environments nearby. Chemical processes in the created environment such as nitrogen cycling and biological processing of organic carbon will be within the range of patterns seen at nearby natural sites. If the habitat is being created in mitigation for the destruction of existing habitat there may be an even stricter requirement, that biological, physical and chemical characteristics are not only within the normal range of these parameters but closely match the characteristics of the habitat being replaced. So, for example, it may not be acceptable to create an intertidal sand flat in place of a mudflat, as the coarser grained sediments on the newly created habitat will contain much lower concentrations of organic carbon and will therefore support much lower densities of invertebrates which will in turn provide much reduced food supplies for over-wintering birds.

In the following sections we examine the extent to which mitigation and compensation schemes have been successful in creating persistent intertidal habitats with desirable physical characteristics; and the extent to which they have been successful in creating habitats with plant, invertebrate and bird populations similar to those on nearby natural environments.

Table 1-1 Monitoring characteristics of single-site evaluations of intertidal habitat restoration projects in the United Kingdom

United Kingdom sites	Name	Reference	Habitat	Tidal Regime	Sedimentary system	Type	Time of creation	Parameters evaluated					
								Topography	Sediments	Plants	Invertebrates	Fish	Birds
Essex	Orplands	Annual monitoring reports (Environment Agency)	SM	Macro	Muddy	Dike	1995	*	*	*			
	Tollesbury	Reading *et al* (1998,1999)	SM	Macro	Muddy	Dike	1995	*	*	*	*	*	*
	Northey Island	English Nature Research Reports 104 & 128	SM	Macro	Muddy	Dike	1991	*	*	*			
	Abbotts Hall		SM	Macro	Muddy	Sluice		?	?	?	?	?	?
	Havergate Island	Cooper (2000)	SM	Macro	Muddy	Dike	2000	*	*	*			*
	Trimley	Unpubl. Reps (EA)	SM/M	Macro	Muddy	Excavate Dredge & Dike	2001				*		*
	Horsey Island	Unpubl. Reps (EA)	SM/M	Macro	Muddy	Dredge	1998				*		*
	Cobmarsh Island	Unpubl. Reps (EA)	SM/M	Macro	Muddy	Dredge	1998				*		*
	Old Hall Point	Unpubl. Reps (EA)	SM/M	Macro	Muddy	Dredge	1998				*		*
	Tollesbury Wick	Unpubl. Reps (EA)	SM/M	Macro	Muddy	Dredge	1998				*		*
	Wallasea Ness	Unpubl. Reps (EA)	SM/M	Macro	Muddy	Dredge	1998				*		*
	Pewet Island	Unpubl. Reps (EA)	SM/M	Macro	Muddy	Dredge	1998						
Devon	Blaxton Meadow, Saltram (Plymouth)	Reading *et al* (1998, 1999)	SM	Macro	Muddy	Dredge	1995		*				
Teeside	Seal Sands	Evans *et al* 1998 Evans *et al* (unpubl.)	M	Macro	Muddy	Excavate/Sluice	1993				*		*
Hants	Thornham Point	Unpubl Reps. (Chichester Harbour Conservancy)	SM	Macro	Muddy	Dike	1990-1991	*		*			
	Chalkdock Point	Unpubl Reps. (Chichester Harbour Conservancy)	SM	Macro	Muddy	Dike	2000	*		*	*		
Somerset	Bleadon Marsh	No details available											

Key: *Habitat*: SM = Saltmarsh, L = Lagoon, M = Mudflat. *Type*: Dike = Breach in sea wall, Excavate = Digging out of site, Dredge = dumping of dredged materials onto site, Sluice = opening of sluice gate in sea wall.

Table 1-2 Monitoring characteristics of single-site evaluations of intertidal habitat restoration projects in the United Kingdom

Region	Name	Reference	Size (ha)	Habitat	Tidal Regime	Type	Age / Time of creation	Parameters evaluated Topography	Seds	Plants	Inverts	Fish	Birds
Hamford Water	Foulton Hall A	Burd (1994)	66	SM	Macro	Nat. breach	1896		*	*			
	Foulton Hall B	Burd (1994)	34	SM	Macro	Nat. breach	1921		*	*			
	Stone Marsh	Burd (1994)	30	SM	Macro	Nat. breach	1874		*	*			
	Walton Central Marsh	Burd (1994)	73	SM	Macro	Nat. breach	1338		*	*			
	Horsey Island	Burd (1994)	5	SM	Macro	Nat. breach	1953		*	*			
	Skippers Island	Burd (1994)	37	SM	Macro	Nat. breach	1953		*	*			
Colne Estuary	Fingringhoe Marsh A	Burd (1994)	70	SM	Macro	Nat. breach	1897		*	*			
	Fingringhoe Marsh B	Burd (1994)	8	SM	Macro	Nat. breach	1897		*	*			
	Aldboro Point	Burd (1994)	7	SM	Macro	Nat. breach	1921		*	*			
	Ferry Lane	Burd (1994)	6	SM	Macro	Nat. breach	1945		*	*			
	Barrow Hill	Burd (1994)	23	SM	Macro	Nat. breach	1953		*	*			
Blackwater Estuary	Sampson's Creek	Burd (1994)	4	SM	Macro	Nat. breach	1945		*	*			
	Northey Island	Burd (1994)	79	SM	Macro	Nat. breach	1897		*	*			
Crouch Estuary	Clementsgreen Creek	Burd (1994)	4	SM	Macro	Nat. breach	1897		*	*			
	Brandy Hole A	Burd (1994)	51	SM	Macro	Nat. breach	1897		*	*			
	Brandy Hole B	Burd (1994)	12	SM	Macro	Nat. breach	1897		*	*			
	North Fambridge A	Burd (1994)	27	SM	Macro	Nat. breach	1897		*	*			
	North Fambridge B	Burd (1994)	43	SM	Macro	Nat. breach	1897		*	*			
	Wallasea A	Burd (1994)	2	SM	Macro	Nat. breach	1953		*	*			
	Wallasea B	Burd (1994)	2	SM	Macro	Nat. breach	1953		*	*			
Thames Estuary	Canvey Point	Burd (1994)	23	SM	Macro	Nat. breach	1874		*	*			
Deben Estuary	Woodbridge	Burd (1994)	15	SM	Macro	Nat. breach	1953		*	*			
	Hemley	Burd (1994)	31	SM	Macro	Nat. breach	1953		*	*			
Somerset	Porlock Marsh	Balance 1994 Chown 2000a Chown 2000b		SM/L/M	Macro	Nat. breach	1996	*		*			*

Key: *Habitat*: SM = Saltmarsh, L = Lagoon, M = Mudflat. *Type*: Dike = Breach in sea wall, Excavate = Digging out of site, Dredge = dumping of dredged materials onto site.

Table 1-3 Monitoring characteristics of single-site evaluations of intertidal habitat restoration projects in the United States

	Reference	Size (ha)	Habitat	Tidal Regime	Sedimentary system	Type	Age / Time of creation	Sampling Freq		Parameters evaluated					
								No	Years	Topography	Seds	Plants	Inverts	Fish	Birds
US East Coast															
CT	Sinicrope *et al* 1990	20	SM	Micro	Muddy	Dike	10	2	2		*	*			
	Fell *et al* 1991						12	1	1				*		
	Peck *et al* 1994						13	2	2				*		
VA	Havens *et al* 1995	2.2	SM	Micro	Muddy	Excavate	5	2-3	1		*	*	*	*	*
NC	Moy & Levin 1991	0.24	SM	Micro	Muddy	Excavate	3	5	3		*		*	*	
NC	Levin *et al* 1996	9	SM	Micro	Muddy	Dredge	5	10	5				*		
NC	Rulifson 1991	2.15	SM	Micro	Muddy	Excavate	3	10	5					*	
	Craft *et al* 1991								2		*				
US Gulf Coast															
TX	Lindau & Hossner 1981	4.5	SM	Micro	Muddy	Dredge	2	4	3		*				
	Webb & Newlings 1985						4	3	5			*			
TX	Minello *et al* 1994	8	SM	Micro	Muddy	Dredge	5	3	``2						
US Pacific Coast															
WA	Shreffler *et al* 1990	3.9	SM	Micro	Muddy	Excavate	3		2	*				*	
	Shreffler *et al* 1992						3		2					*	
	Simenstad & Thom 1996						8		7		*	*	*	*	*
OR	Frankel & Morlan 1991	32	SM	Micro	Muddy	Dike	10		11	*	*	*			
CA	Chamberlain & Barnhart 1993	3.5	SM	Micro	Muddy	Dike	2	3-7	1.3					*	
CA	Langis *et al* 1991	4.9	SM	Micro	Muddy	Excavate	4	2-6	2		*				
	Scatolini & Zedler 1996						4	8	1		*		*		
	Zedler 1996						2	14	6		*	*	*	*	

Key: *Habitat*: SM = Saltmarsh, L = Lagoon, M = Mudflat. *Type*: Dike = Breach in sea wall, Excavate = Digging out of site, Dredge = dumping of dredged materials onto site

2. Why should lost intertidal habitat be replaced? The effects of habitat loss on bird populations

2.1 Introduction

It is usually assumed that loss of intertidal habitat will lead to a decline in waterfowl populations. The extensive literature on the effects of habitat loss on waterfowl populations consists mostly of theoretical studies with a few empirical studies. Empirical studies are rare for several reasons: (i) habitat loss is often small in comparison with the remaining habitat, (ii) waterfowl are extremely mobile and the fate of displaced birds is difficult to follow, (iii) studies need to be long-term and detailed (ie include observations on food supply and other factors which affect waterfowl usage of an area) and (iv) high natural variability in waterfowl fluctuations from year to year makes it more difficult to measure the effects on the population as a whole. The few studies that have taken place within the UK have tended to show either no effect or a local decrease in waterfowl numbers.

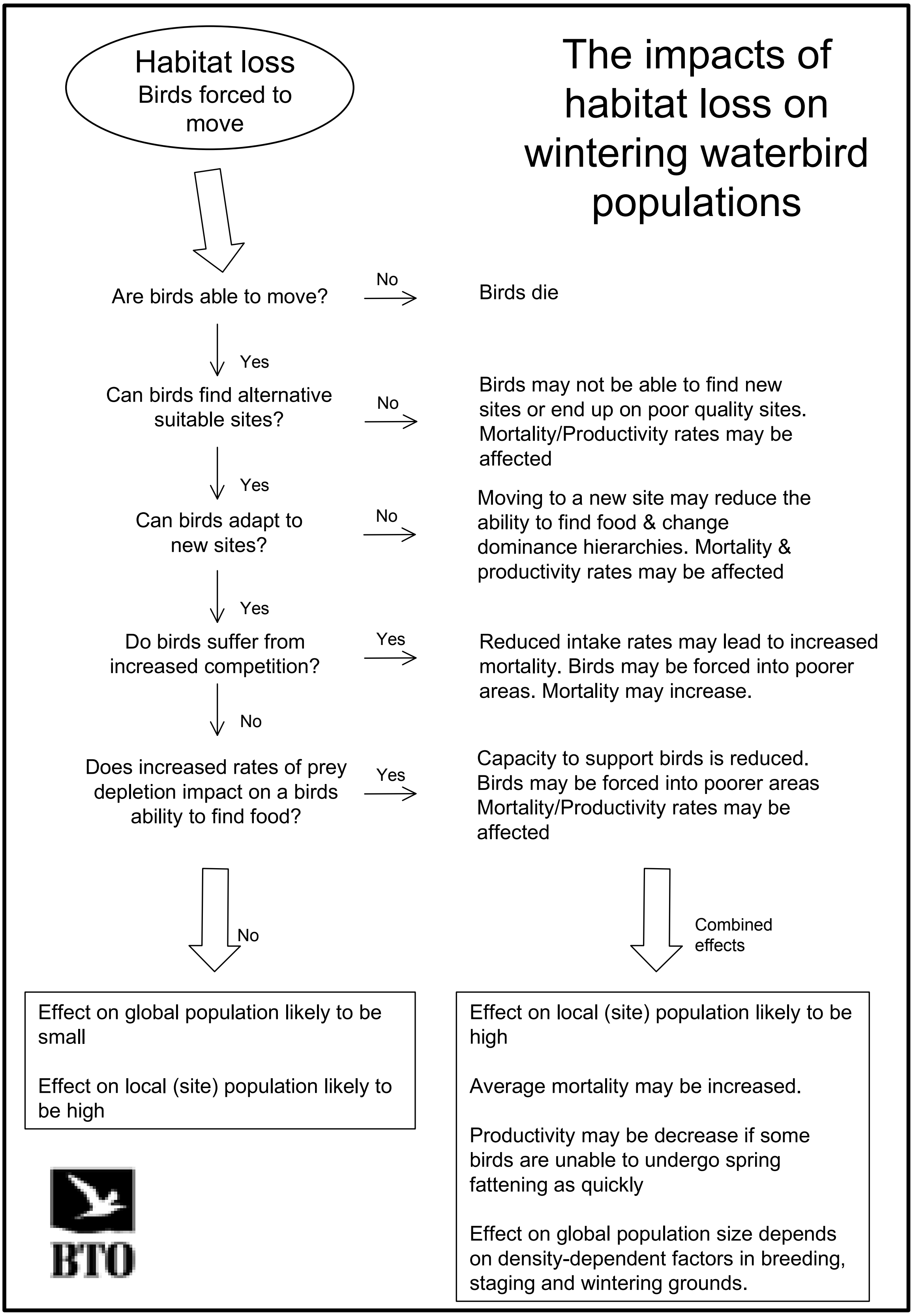
2.2 A brief summary of theoretical studies

2.2.1 A framework within which to evaluate the effects of habitat loss on waterfowl populations

The application of theoretical models have proved useful to understand the effects of habitat loss and changing management on waterfowl populations. These models, based on sound knowledge of the ecology of each particular species and situation, have provided reasonable estimates of real-world scenarios. The main issues addressed in these models are included in Box 2.1. The models have successfully predicted the usage of mudflat and saltmarsh by Brent Geese and Wigeon at Lindisfarne (Percival *et al* 1996,1998), the usage of saltmarsh and agricultural land by Brent Geese on the North Norfolk Coast (Rowcliffe *et al* 1995, 1998, 1999, 2001), the interaction between management and the Wigeon and Bean Geese populations in the Yare Valley (Sutherland & Allport 1994), the effects of habitat loss on Twite populations wintering on saltmarshes in south-eastern England (Atkinson 1998) and the usage of mudflats by Black-tailed Godwits at three different scales in southern England (Gill *et al* 2001). Sutherland (1986a) provides a useful overview of depletion and interference-based models and their application to determining the effects of habitat loss on bird populations.

These models are all based around a similar framework. Waterfowl are mobile and most use a network of sites of varying quality during the non-breeding season. In the framework, birds are individuals and have different competitive abilities. They distribute themselves around the network of sites based on certain criteria and the decision to settle on a site is likely to be determined by factors such as food supply, suitable roost sites, predation rates and *density* of and *interference* from conspecifics (see Box 2.2 for a glossary of terms in italics). These birds *deplete* the resources and are *free* to move between sites. If the birds behave in an *ideal* manner (Fretwell & Lucas 1970) then it would be expected that individuals distribute themselves across sites so that they can maximise their resource usage. This in turn means that birds will occur at the highest densities at the most favourable sites. The consequences of a bird's decision can also be translated into survival and if a bird fails to maintain an adequate intake rate, it can die. In this way effects of habitat change can be assessed at the population level.

Box 2.1 Issues determining how habitat loss affects wintering waterbird populations



Box 2.2 Depletion, interference, density-dependence and the ideal free distribution

Ideal Free Distribution, depletion, interference and density-dependence are four key concepts in the models that have been applied to determining the effects of habitat loss or degradation on waterfowl populations.

Interference: This is the decline in resource use resulting from the behaviour of other individuals around you (Sutherland 1996a). An example of this could be one Oystercatcher defending a patch of high quality mussels and forcing another bird to forage in a poorer quality area elsewhere or cases where food stealing occur. Indirect interference can also occur - high feeding densities of Redshank can cause the crustacean *Corophium volutator*, a favoured food item, to bury deeper in the mud making them less available to the birds (Selman & Goss-Custard 1988).

Depletion: Depletion is the permanent removal of resources from the system through predation and can be thought of as a non-reversible form of interference.

Density-dependence: This is simply the concept that a particular parameter varies with the density of the organisms being studied. For example, mortality may increase with increasing density due to greater effects of interference. One example of this would be Oystercatchers defending a mussel bed. No matter how many birds start the winter, the bed can only support a certain number of territories. As the density of birds at the beginning of the winter increases, mortality increases so that the same number of birds remain at the end of the winter. Winter mortality is therefore said to be density-dependent.

Ideal free distribution: The models applied to each situation often assume that the animals behave *ideally,* which means that they go to the patch where their rewards are highest, and that they are *free* to move where they want to and are not limited in terms of dispersal ability. The IFD is used as a useful base for theoretical studies.

Predicting the effects of habitat loss or degradation is easily incorporated into these frameworks. For example, loss of feeding habitat has the effect of displacing birds and increasing the density of foragers in the remaining feeding habitat. Degradation, however, changes the quality and not the area, eg by reducing the density of invertebrates available to shore birds in a site through a pollution incident. If the birds which fed in these sites are able to move to surrounding areas, and there is no increase in the effects of interference, then this will have the effect of depleting resources more quickly thus reducing the overall capacity of the site to hold as many birds for as long a period of time. If the increase in density causes birds to suffer a greater degree of interference then this may cause a reduction in the usage of the site. Interference can come in many forms and may be a product of increased competition between species, eg aggression between Oystercatchers on mussel beds or increased fighting amongst Knots at high densities or due to other factors such as anti-predator responses of prey.

Depletion and interference can both play important roles in determining the usage of a site and the effects of these two factors need to be understood before the effects of habitat loss can be predicted with a degree of certainty. The relative contribution of these two factors in determining the total usage of an estuarine site is poorly known for most species but one can predict that for species which form feeding flocks and show little aggression, such as Knot and Dunlin, depletion is likely to be the relatively more important factor, although interference should not be discounted as it can be extremely difficult to measure.

2.2.2 The concept of population size and the effects of habitat change

The framework and studies described earlier predicts total usage of a site but does not necessarily include the consequence for survival and recruitment at the global or flyway population level which, arguably, is the more important question. In terms of a whole population, the size of the population can be driven by density-dependent and also density-independent processes which can take place at different times during an organism's life cycle. Considering a simple scenario, Sutherland (1996b) showed that change in population size resulting from habitat loss can be determined by a simple relationship between the density-dependence occurring in the breeding and non-breeding grounds. For example, if the population of a species is limited by suitable nest sites then removing or reducing the quality of small amounts of wintering habitat may have little or no affect on the overall population size. By contrast, if the population size of a species is limited by the availability of winter feeding habitat, then loss of even small amounts of this habitat may impact directly on the overall population size. Using game theoretical models to estimate the strength of density-dependence in mortality and productivity, Sutherland predicted for the Oystercatcher *Haematopus ostralegus* that a loss of 1% of wintering habitat will result in a population decline of 0.69% in the population while a loss of 1% of breeding habitat will result in a population decline of 0.31% but adds the caveat that values for density dependence are based values from single sites. This approach can also be applied to calculating the effects of local habitat deterioration (or improvement) on total population size. Sutherland (1998) predicted that a decline in Oystercatchers on The Wash, England from 43,500 in January 1989 to 7,800 in January 1997 resulted in a decline in the total population of 24,633. As Sutherland points out this method is unlikely to give exactly the same result as a full model which incorporates many different biological and behavioural parameters but do provide a quick approximation of the effects of habitat loss. This does depend on the availability of estimates of breeding and wintering density-dependence.

If overall population size is determined by factors operating during the breeding season then density-dependent processes may still affect the distribution of birds across sites during the non-breeding season. The concept of carrying capacity has often been applied to birds wintering on estuarine systems. The definition given by Goss-Custard & West (1997) of a 'one in one out' scenario of carrying capacity based on the resources available and the enemies present is a useful theoretical concept but is an unrealistic measure. Although, theoretically possible, large natural variations in prey density, both seasonally and between years (Beukema *et al* 1993; Zwarts & Wanink 1993), disturbance, predation, the ability of species to switch between prey species (Beukema 1993) and the likelihood that individuals may not behave in an ideal manner may mean that this theoretical carrying capacity is unlikely to be reached. Thus, shorebirds and other waterfowl populations will tend to fluctuate below this level.

Goss-Custard & West (1997) also point out that carrying capacity is difficult to measure and the manner in which it is expressed is crucial to its understanding. For example, carrying capacity can be defined as the maximum number of birds that can be supported at the start or end of a winter or as total bird usage over a winter. All of these measures have drawbacks and each may be appropriate to some situations and not others (see Goss-Custard & West 1997 for a review). For example, for species where depletion is important, eg Black-tailed

Godwits, total bird usage is a more useful measure, whereas in a species such as Oystercatcher where interference is an important factor in determining population size the number of birds remaining at the end of a winter may be a more realistic measure to use.

Despite the large numbers of drawbacks, carrying capacity is often cited in studies investigating habitat loss or degradation as it is a simple concept to grasp. Although it may be useful, in some circumstances, to predict the consequences of habitat loss at whichever scale, it is essential to understand both the ecology and behaviour of the species being studied.

2.2.3 The relationship between shorebirds and their food

Despite the difficulties in determining the effects of habitat loss on waterfowl populations, there is strong evidence that the usage or density of shorebirds on intertidal sites is often determined largely by the availability of suitable food and, as such, any removal of foraging habitat will reduce the site's capacity to support waterfowl. One of the larger scale studies which has investigated the relationships between shorebirds and their food supplies is described in Box 2.2 but many other examples are to be found in the literature, eg spring usage of brackish lagoons by shorebirds in North Carolina was related to prey density (Weber & Haig 2000), Oystercatcher abundance was related to their shellfish prey (Meire 1996) and Knot abundance in the Wadden Sea was related to their prey, notably *Macoma* but also *Hydrobia* and *Cerastoderma* (Piersma *et al* 1993). The major prey items of some estuarine shorebirds and wildfowl have been extracted from the literature and are presented in Table 4.2.

There is less literature on the relationship between estuarine wildfowl and their food supplies. Shelduck in the Wash were related to densities of oligochaetes, dipterous larvae, *Hydrobia* and *Corophium* (Yates *et al* 1993). The usage of saltmarsh habitats by Brent Geese in North Norfolk was found to be determined by the food available on the marsh (Rowcliffe 1994). As birds arrived back from the breeding grounds, they first fed on algae until it was depleted, then moved on to saltmarsh grass *Puccinellia maritima* and then inland to feed on cereal crops. The population has undergone a large increase over the past 30 years and the move from saltmarsh to arable habitats occurs earlier as depletion of intertidal food resources on the saltmarsh in winter has taken place at a faster rate. Other examples include a strong relationship between the number of Twite in a site during late winter and the density of *Salicornia* seeds, their main prey item (Atkinson 1998).

Box 2.2 The relationship between estuarine shorebirds and their benthic prey in south-western England

In response to a proposed tidal barrage on the River Severn a study was set up to investigate the relationship between the densities of shorebirds and benthic invertebrates on six estuaries in south-west Britain. Comparisons were made at 40 sites within these six estuaries. Within most estuaries bird density was correlated with densities of one to three widely taken prey species (see Table 2.1 below).

Table 2-1 Bird-prey associations, marked by the filled in circles, in six south-western UK estuaries (from Goss-Custard *et al* 1991)

	Dunlin	Redshank	Grey Plover	Bar-tailed Godwit	Black-tailed Godwit	Oystercatcher	Curlew
Nereis diversicolor	•	•	•	•			•
Nepthys hombergii			•				•
Scolopos armiger				•			
Cirratulidae	•						
Corophium volutator		•					
Cardium edule						•	
Mytilus edulis						•	
Scrobicularia plana					•		

2.3 Examples of studies which have measured the effect of habitat loss on waterfowl populations

When placed in the framework described above, it is very difficult to accurately assess the effect of habitat loss on wintering waterfowl populations without detailed ecological studies. The effects depend on the quality, quantity and a bird's knowledge of other intertidal habitat and its ability to disperse and find new suitable habitat, the densities of birds elsewhere and the susceptibility to interference of a bird arriving at a new site. It is not surprising, given that most estuarine habitat loss tends to be small relative to the overall estuary, that many empirical studies fail to find an effect.

There are two main types of paper in the peer-reviewed literature which investigate waterbirds and habitat change. The first investigate the empirical relationship between the loss or degradation of the habitat and the response of the waterbird assemblage. Often the lack, or short-term nature, of pre-loss monitoring data, the large inter-annual variability in waterbird population size, different climatic conditions between winters and the dynamic nature of intertidal areas means that relationships are often impossible to determine. Also, many studies are superficial in that the relationships between the waterfowl species, their prey and other factors that will affect a species usage of the site are poorly understood and only numerical responses are considered.

The second, and rarer, type of study takes a demographic approach and investigates the demographic mechanisms through which habitat change affects population size. Two prime examples are those that were carried out on populations of Dark-bellied Brent Geese and Oystercatchers in the Netherlands. The Black-tailed Godwit example above (Box 2.3) also shows that where a bird decides to winter has important implications for its survival and also, possibly, its breeding success the following summer.

The few studies that have been published (Table 2.2) show that removal of habitat can act on waterfowl populations in different ways. The loss may be total (eg the loss of a favoured goose staging site, Ganter & Ebbinge 1997) forcing birds to move to different sites, or partial (eg removal of some mudflats on the Orwell Estuary) forcing birds into surrounding inter-tidal areas. The loss may also affect feeding time as many reclamations tend to be at the top end of the intertidal range. This will reduce feeding time available to birds as was observed on the Tees Estuary (Evans 1997).

The following case studies also include those that modify, rather than remove, intertidal habitats. These include, for example, the change in the nature of the sediments at Fagbury Flats when coarse material spread from the reclaimed area onto the mudflat or changes in the tidal range or flow characteristics in an estuary. Also, the resulting habitat may either lose its waterfowl interest totally or there may be change in the waterfowl assemblage, as was the case in Nordstrand Bay (Hotker 1997) where some, albeit different, intertidal habitats remained.

Box 2.3 Black-tailed Godwit usage of areas is not only strongly related to the prey available but also wintering in areas of differing quality can have implications for survival and productivity.

The Black-tailed Godwit is a good example of a species whose wintering numbers are regulated by depletion of their invertebrate prey. Black-tailed Godwits winter on the south and east coast estuaries of England and there is a strong relationship between the total number of bird days (ie total usage) and the amount of prey available at those sites (Figure 2.1 from Gill *et al* 2001). Depletion models based around the predator-prey relationships identified in this study predicted the actual usage at both small (mudflat), medium (estuary) and large (national) scales indicating that knowledge of the initial prey densities and the relationship between intake rate and prey density was sufficient to explain the total usage of the godwit's estuarine sites.

This population of godwits, which breed in Iceland, has been increasing since at least the 1970s. As numbers, and hence density, have increased, this scenario can be used as a useful parallel to habitat loss where increases in density also occur. When the population of godwits was low, birds were concentrated on south coast sites. Following the increase, and as the population increased, a classic 'buffer effect' has been observed. Food supplies, and hence intake rates, are generally poorer on east coast estuaries. As the population has increased, birds have been increasing at a faster rate on poorer quality, ie east coast sites, indicating that some density dependent factors are in operation. This has also been shown to have important consequences for population regulation, as the birds on poorer quality sites were demonstrated to have significantly poorer survival rates compared with birds from better quality, ie south coast sites. This difference in survival rate (94% on south coast sites and 87% annual survival on east coast sites) is a major difference as 50% of south coast individuals will die 11 years after reaching adulthood compared with just five years for east coast birds. Furthermore, wintering on a poorer site may also have implications for breeding success as the birds wintering on south coast sites arrived back in the Icelandic breeding grounds earlier than east coast birds. Early arrival in the breeding grounds is often associated with obtaining better territories and hence productivity.

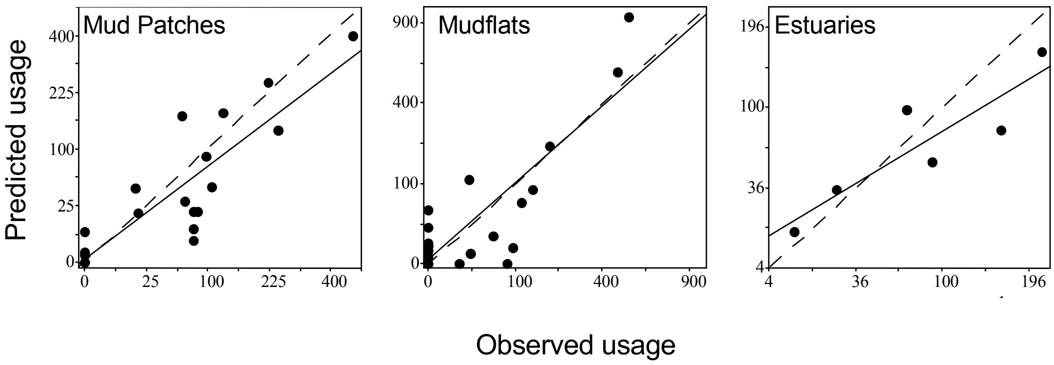


Figure 2.1 Predictive depletion models, based on knowledge of the food supplies available and also the relationship between birds and their food, accurately predict how Black-tailed Godwits use different size feeding areas ranging from small mudflats to whole estuaries (from Gill *et al* 2001)

Case Studies Group 1: Studies which have investigated the numerical response of waterbirds to habitat loss

One of the larger European studies of habitat loss affecting waterbirds was undertaken on the impact of the removal of 33 km^2 of saltmarsh, mud flats, sand flats and some tidal channels in Nordstrand Bay in the German Wadden Sea (Hotker 1997). The replacement area, the Beltringharder Koog, contained saltwater tidal lagoons and extensive fresh-water habitats to compensate for the lost habitats. The tidal range of the lagoons was reduced to 40 cm, which exposed only 140 ha of mud flat at low tide. Benthic invertebrates tended to occur at lower densities here compared with the Wadden Sea. Semi-quantitative predictions of the effects of the habitat loss were made by knowledge of the ecology of each species, the plans for the embanked area and data from other reclaimed sites. Of the important waterfowl that used the area prior to the reclaim, only the herbivorous species Wigeon and Barnacle Goose showed increases in the reclaimed area and, even with the remaining intertidal habitats, shorebirds showed a large decline. Only Redshank used these areas for feeding at high tide and the reduction in tidal range, reduced invertebrate densities and enclosure by tall embankments rendered the remaining area unsuitable for shorebirds. As a result of the embankment, the numbers of Brent Geese, Shelduck and most shorbird species decreased in the area surrounding the embankment compared with pre-embankment numbers. Most species elsewhere in the German part of the Wadden Sea increased or were stable over the same period. Hotker concluded that 'the lost feeding opportunities due to the land claim could not be compensated in the immediate surroundings of the study site'.

This study emphasised the fact that it is difficult, if not impossible, to measure the effects of localised habitat loss on wider populations, due to changing environmental conditions, the relatively small size of the land claim and also the transitoriness of bird populations. It also concluded that the attempts to compensate for the loss of habitat was not successful, although it did restore the freshwater transition habitats that had been largely lost due to previous reclaims. The proposed compensatory measures were insufficient to account for the loss of mudflats. This highlights that compensation measures must ensure that habitats of similar or better quality and quantity to those that are lost are provided as compensation. Even in the case of such large scale habitat loss, it was not possible to determine the effects of habitat loss on the larger population in the Wadden Sea.

Other smaller scale studies have only investigated local habitat loss; two well-known UK examples include the loss of mudflats at Fagury Flats on the River Orwell and the loss of intertidal habitats on the Tees Estuary.

The Tees estuary has undergone extensive reclamation and is one of the most heavily industrialised estuaries in Britain. Habitat loss was at such a scale that Evans *et al* (1998, 2000) recorded changes in some shorebird species despite large annual fluctuations in numbers using the whole estuary. By 1971, only 15% of the 2,500 ha intertidal area remained and 60% of the main feeding area was enclosed by a porous slag wall. This area was gradually infilled with dredgings from 1973 and 1974 and by spring 1974 only 140 ha remained.

Birds could respond in different ways and at the time of the research it was unclear whether the same number of birds would settle but use the area for less time, or whether there would be a reduction in the number settling on the estuary. In fact, there was little change in the seasonal pattern of abundance but there was a reduction in the number of birds settling on the estuary in the autumn. Evans predicted that there would be insufficient benthic resources to support the number of birds which were using the estuary prior to the development and predicted that those feeding on *Nereis* and *Hydrobia* (the two common macroinvertebrates at the site) would undergo the greatest reductions. There were clear declines in Grey Plover, Curlew and Dunlin which feed to various extents on these two species of invertebrate. Redshank numbers also declined and changes in numbers were related to changes in *Corophium* numbers on Seal Sands. Shelduck, however, increased in 1975 and 1976 probably due to the increased amounts of fine muddy sediments which provide a good feeding habitat. As these have compacted, the number of Shelduck have declined. As time has progressed there have been other changes. For example, an influx of sand has led to increased numbers of Bar-tailed Godwit and Ringed Plover.

This study is one of the clearest examples of declines in waterfowl following habitat loss. As reclamation took place at the upper end of the tidal range, there was not only habitat loss but also a reduction in the feeding time available to the birds. However, it was not possible to determine the fate of the birds that were displaced by the habitat loss.

On the River Orwell, the construction of the Trinity Terminal at the Port of Felixstowe in 1985-86 and the enclosure and reclamation of a strip of land at Fagbury Flats led to a reduction in the amount of intertidal habitat on the estuary. Discharge of sandy sediment from the Trinity Terminal onto part of Fagbury Flats also led to a change in the nature of the sediments in the area, thus modifying the remaining area. In total out of the 34 ha of intertidal habitats, only 15 remained after the reclamation. The changes in waterfowl numbers on the Flats are presented by Evans (1995). For most shorebird species there were declines in the number of birds settling on the new area compared with those that occurred before the reclaim. However, due to changes in the substrate (from the sediment discharge from the new area) these changes were not necessarily in proportion to the loss of habitat. Similar local declines have been reported from the Forth estuary in Scotland and in Denmark (Laursen *et al* 1983; Mclusky 1992)

The loss of saltmarsh may also affect local wintering populations. Effects of saltmarsh loss were not shown on the population of Redshank breeding on marshes in the UK although changing grazing management was implicated in the change in Redshank numbers (Norris *et al* 1997). However habitat loss has been implicated in the number of wintering Twite in south-east England. The English breeding population of Twite winters exclusively on the saltmarshes of south-eastern England. These marshes have undergone a large reduction in area during the last 30 years due to a mix of factors such as anthropogenic influences (eg reclaim and dredging), climate change, sea-level rise and isostatic change. On some estuaries over 50% of the total area was lost over a 30 year period. The greatest loss took place at the seaward end of the saltmarsh which supported the pioneer communities dominated by *Salicornia*.

The distribution of Twite at the end of the winter is strongly determined by the density of *Salicornia* seeds remaining. This relationship is not found in autumn as food is super-abundant at that time before the removal of seed by storms, tide and other predation. Atkinson (1998) showed using depletion models that the decrease in the pioneer saltmarsh and consequent loss of *Salicornia* during the 1970s and 1980s was sufficient to explain the decline in the Twite population. The current regime of managed retreat sites and those sites described by Burd where natural breaches had formed were mostly unsuitable for Twite due to the nature of the resultant saltmarsh vegetation. As most sites were flat and had poor drainage the resulting vegetation was a rank *Puccinellia* and *Halimione* sward, unsuitable for this species.

Case Studies Group 2: Studies which have measured the subsequent effects of habitat loss on demographic parameters

Although studies which evaluate the consequences of habitat change in terms of demographic parameters are ideal, very few are published in the literature. Ganter *et al* (1997) investigated the consequences of the loss of a Brent Goose spring staging site on subsequent dispersal, fecundity and survival. No significant differences in the fecundity or the survival of the displaced and control birds could be found although there was a tendency towards lower breeding success and lower survival in displaced birds. It appeared that most of the birds managed to find suitable alternative sites. Ganter *et al* also point out that, even if the individuals that were displaced suffered no significant adverse effect, the increased density on remaining sites may reduce the fitness parameters of the population as a whole.

The other well-known study is that of the effects on the Oystercatcher population of the loss of approximately 30% (approximately 170 km^2) of the intertidal habitats in the Oosterschelde in south-western Netherlands (Lambeck 1991; Meire 1996). The removal of intertidal habitats caused general declines in those species dependent on these areas (Schekkerman 1994). Up to 10,000 Oystercatchers were unable to establish themselves on the remaining intertidal area and displaced birds were more likely to be found in poorer quality areas and disappeared more often than birds which were not displaced. This study highlights the problem for individual birds moving to previously unknown areas and indicated that, unlike the previous goose example, moving may have consequences for survival. The mechanism for this is unknown but is likely to be related to either living in a poor quality area (as in the godwit example) or due to increased interference from other, already established birds.

Although, arguably, not directly removing intertidal habitat, over-fishing of mussel beds on the Wash, England, led to a collapse in the mussel fishery. The number of mussel beds reduced from 20 to one during the 1980s and 1990s and Oystercatchers on the Wash underwent a similar decline from the mid 1980s from approximately 40,000 birds to 10,000 over a 10-year period (Atkinson *et al* 2000). Additionally, in three years, very high winter mortality was seen. In normal winters, winter mortality (October to March) was approximately 2% but during these three winters was 10-15 times greater than the normal rate. Although the removal of the mussel beds was in itself not a problem for the Oystercatchers as there was other prey (notably cockles *Cerastoderma* and *Macoma*) available, it was not until the cockle stock collapsed that Oystercatchers died in large numbers. Cockle populations are normally extremely variable, whereas mussels tend to be more stable. The loss of the mussel beds removed a buffer for when cockle stocks were low, thus causing the very large Oystercatcher mortality.

As in the above example the removal of habitat may not have immediate consequences. For example, during normal winters an area may support low densities of birds but during cold weather may act as a refuge for populations of birds which normally winter elsewhere. One of the most spectacular examples of this is the ‘cold-rushes’ seen in the Wadden Sea. Camphuysen *et al* (1996) describe Oystercatcher cold-rushes between 1972 and 1996. These involve movements ranging from hundreds to tens of thousands of birds along the Dutch coast to areas in northern France. These occur during cold weather and the areas in France act as a cold-weather refuge. Mortality was increased during these cold-rushes, both because of the effects of cold weather and also the influences of French hunters killing birds in these refuge areas. It is therefore important to understand that removal of intertidal habitat is more likely to result in increased mortality during time when birds are highly stressed.

2.4 Conclusion

All of the studies detailed above and in Table 2.2, indicate that habitat loss can have local effects and that wider effects on population size are difficult to measure. However the studies which have investigated the population consequences have shown that forcing birds to move elsewhere may increase mortality rates by forcing them into poorer areas or increasing their susceptibility to interference from other foragers. To fully understand the impacts of habitat loss on waterfowl populations, it is essential that in future situations a sufficient amount of pre-loss monitoring is carried out which should include detailed studies of the relationship between the birds and their prey and also an understanding of their dispersal and subsequent survival through studies of individually marked animals. The effects of habitat loss may also not be immediate and it is important to understand the role of areas as cold-weather refuges. Also, the removal of areas may cause birds to become reliant on a different, perhaps extremely variable, food source. When this prey declines in number there may not be sufficient alternative food sources available to sustain the population.

Table 2-2 Studies which have empirically measured the effects of intertidal habitat loss on waterbirds

Site	Type of habitat loss	Conclusions	Reference
Orwell Estuary	Removal of saltmarsh and mudflat to accommodate expansion of the Port of Felixstowe. In 1988-89, 19 ha out of 34 ha of intertidal areas on Fagbury Flats (parts of the Orwell Estuary) was enclosed and filled. Part of the remaining flats were degraded due to sediment overspill from the infilling process.	Local declines in shorebirds were observed but the small area enclosed (c 3% of total estuary intertidal area), the small number of birds involved and high natural interannual variation in shorebird numbers made it difficult to detect the effects of the local decline on total Orwell numbers.	Evans 1997
Tees Estuary	By 1971, only 15% of the total area present at the beginning of the 19th Century was present due to reclaim for industrial development.	Removal of intertidal habitat led to (a) loss of feeding area and (b) loss of feeding time for some species. The effect on individual species depended on diet, the time required for feeding, presence of other feeding sites during high tide and interactions with other species.	Evans 1978/79 Evans 1997
Wadden Sea	Embankment of saltmarsh area favoured by spring-staging Brent Geese.	Displaced birds appeared more often in less preferred areas. No detectable changes in productivity or survival among displaced birds	Ganter & Ebbinge 1997
Wadden Sea	Short-term effect of reclamation on numbers and distribution of waterfowl at Hojer, Danish Wadden Sea. 11 km^2 of saltmarsh removed through diking.	Overall shorebirds decreased by 80% and ducks 60%. Although only 10% of the tidal flats were removed 90% of the mudflats were removed. These tend to support higher invertebrate biomass than remaining sandier area. The time allowed for feeding was reduced by 1.5 hours. Species preferring muddier areas (Grey Plover, Redshank & Dunlin) all declined in numbers in the local area.	Laursen *et al* 1983
Nordstrand Bay	Response of migratory coastal bird populations to the land claim in the Nordstrand Bay, Germany.	Local declines in some species. Overall most species showed increase in the whole Wadden Sea.	Hotker 1997

Site	Type of habitat loss	Conclusions	Reference
Forth Estuary	The impact of land-claim on macrobenthos, fish and shorebirds on the Forth estuary, eastern Scotland	Effect of loss of 20% of mudflat habitat led to statistically significant declines in 2 out of 9 species (Dunlin and Bar-tailed Godwit). 5 out of 9 species declined by > 20% (Oystercatcher, Turnstone, Bar-tailed Godwit, Knot, Dunlin) No change in Wigeon & Redshank.	Mclusky 1992
Delta area, SW Netherlands	Effects of a substantial reduction in intertidal area on numbers and densities of shorebirds. A storm-surge barrier and secondary dams were built in the Oosterschelde estuary (The Netherlands) resulting in a 30% decrease of intertidal area.	Total numbers of Oystercatchers in the unaffected parts of the Oosterschelde did not increase after the loss of feeding grounds in the Krammer-Volkerak. Up to 10,000 Oystercatchers failed to establish themselves in the remaining intertidal area (also see Lambeck 1991)	Meire 1996
Delta area, SW Netherlands	Changes in abundance, distribution and mortality of wintering Oystercatchers after habitat loss in the delta area, SW Netherlands.	Oystercatchers which were displaced from areas which were lost occurred more frequently on poorer quality areas and disappeared more often than birds which were not displaced.	Lambeck 1991
Delta area, SW Netherlands	Removal of 170 km^2 of intertidal habitat between 1982 and 1987 in the Oosterschelde (SW Netherlands) as a result of large-scale coastal engineering works.	Open water species remained stable or decreased. Those dependent on intertidal areas decreased and intertidal bird density decreased slightly. Those birds displaced by engineering works were not able to settle in the remaining areas.	Schekkerman 1994

3. Success of mitigation and compensation schemes at creating intertidal habitat

3.1 Introduction

There have been very few studies that have looked in depth at the geomorphological characteristics of restored mudflats and saltmarshes. Yet, it is the geomorphology of the intertidal wetland system that defines the range and degree of environmental functions that the wetland undertakes (Zedler 2000). Given adequate sediment supply and suitable hydrodynamic conditions, often a marsh will slowly develop and succeed naturally from a mudflat, its morphology most likely reflecting this slow and gradual evolution.

Globally, a range of techniques has been employed in an attempt to restore or create saltmarsh and intertidal flats. These techniques include managed realignment of flood defences involving either a simple breach with maintenance of an outer wall or total bank retreat. At several sites in the UK and adjacent shores of north-west Europe, attempts have been made to enhance natural sedimentation within sedimentation fields (brushwood fenced areas). At several locations around the world marshes and mudflats have been created rapidly utilising dredge material (See US east coast review by Streever (2000) for example). In many cases the need to dispose of dredge material has been a principle driving force for the placement of material on the intertidal zone. Elsewhere, the use of dredge material has been driven by the need to rapidly provide natural flood defences or because marsh mitigation is required in a shorter time period than that taken for natural marsh formation.

The following sections will:

1. outline the basic scientific understanding of mudflat and saltmarsh development (section 3.2);
2. briefly describe the most common methods for creating saltmarsh and mudflat systems and outline the pros and cons of each (section 3.3);
3. draw together lessons learned from several relevant case studies within the UK, the Netherlands and United States (section 3.4).

3.2 Background to intertidal flat and saltmarsh development

Most basically, a saltmarsh is an intertidal mud or sand flat that has been colonised by salt-tolerant (halophytic) vegetation. Thus, saltmarshes and mudflats are a linked continuum of intertidal habitats, much in the same way that beaches and dune fields are linked in sandier systems. The colonisation by vegetation not only differentiates a marsh from a mudflat but also facilitates increased sediment accumulation on the marsh surface, which becomes raised relative to that of the adjacent mud or sand flat and takes on a morphology of its own. (Box 3.1).

Box 3.1 Modelling marsh accretion

To the geomorphologist, seeking to model intertidal evolution, a saltmarsh may be thought of as a high intertidal environment comprised of repeatedly flooded, vegetated platforms, dissected by blind-ending tidal creeks that widen seaward (Allen 1997). Quantification of saltmarsh vertical accretion rates, under varying conditions of sediment supply and sea-level rise has been modelled within a number of studies of minerogenic (sediment-dominated) saltmarshes from a number of large (meso-macro) tidal coastlines (Randerson 1979; Krone 1987; Allen 1990a,b, 1995, 1997; French 1991, 1993) and, less commonly, from studies of organogenic (vegetation-dominated) microtidal marshes (Churma *et al* 1992).

The basis of these models is essentially a mass balance calculation reflecting the change in surface elevation over time, relative to the moving tidal frame in response to mineral and organic sedimentation rates, and the rate of rising sea-level. By specifying the sediment supply, tidal and sea-level regimes and initial elevation, the mass balance equation may be integrated over time to reflect the growth of a young marsh from an intertidal flat or the response of a well established wetland to sea-level fluctuations. There are still several gaps in the models that have yet to be accounted for. These include: spatial variability in sedimentation; predicting the impact on sea level rise on organic sedimentation (root remains within sediment etc) and; long-term consolidation of sediments.

Though lacking topographical and spatial detail, the outcome from these vertical accretion models is upheld by observations describing sedimentation rate to be strongly related to the amount of time the marsh is flooded, and to decline rapidly and non-linearly as the marsh surface gains elevation relative to the intertidal frame. A state described as reaching 'maturity' is attained when the marsh surface reaches a level some distance lower than the highest astronomical tide (H.A.T.), in equilibrium with the rate of sea-level rise. The models also predict that for minerogenic marshes, such as those found in the UK, marsh growth is very sensitive to the balance between sediment supply and the rate of sea-level rise. On the north Norfolk coastline, French's (1993) model supported corrected empirical data (Pethick 1980, 1981) describing these marshes to reach maturity after about 300 years.

By extending these models it is possible to look at the effect of increased rates of sea-level rise (or reduced sediment influx) on marsh existence. French (1993), for example, re-ran his model to simulate the effect on marsh status of a range of possible sea-level change scenarios (1.1-15mma^{-1}) and found that only under the more extreme predictions did ecological drowning take place by the year 2100. Under more moderate conditions (c. 4mma-^{1}) the marsh surface stabilised at a low elevation, which would cause a change in marsh halophyte community structure. It must be borne in mind that an addition effect of increased water level, if uncompensated by an adequate supply of sediment, will lead to an increase in wave attack, an erosion of intertidal sediments and movement upward and shoreward in estuarine habitats. This erosion may be more significant than ecological drowning in many estuaries, particularly those confined by flood embankments.

The factors that control the morphology of an intertidal flat and saltmarsh are complex, involving dynamic interactions between erosive or reshaping forces of waves and tides, the resistive forces inherent within sediment cohesion and, finally, the binding action of plants and micro-organisms. The biotic factors often vary seasonally (Patterson & Black 1999). Erosion rates are lower on vegetated marshes than bare intertidal flats because of the binding actions of plant roots and because of the effect of vegetation in baffling flow. In this way, mudflats tend to respond rapidly to wave and tidal conditions whereas marshes tend to respond to larger events such as significant storms. Moreover, the sediments on the marshes are drier than on intertidal flats because of the elevation difference.

In terms of vegetation cover, numerous studies have shown that vegetation establishment on saltmarshes is principally determined by one key factor: surface elevation relative to the tidal frame (the position between highest and lowest tidal levels) (Davy 2000). It is possible to estimate empirically the elevation at which primary vegetation (such as *Spartina* spp., *Salicornia* spp. and *Puccinella maritima*) will begin to colonise a mudflat or sandflat surface and the elevations at which low- mid- and high-marsh communities will succeed these. However, the complex physical and ecological interactions that influence marsh development are very poorly understood and thus there is little capacity to predict how developing vegetation community structure on the marsh surface will change with time, as the marsh matures.

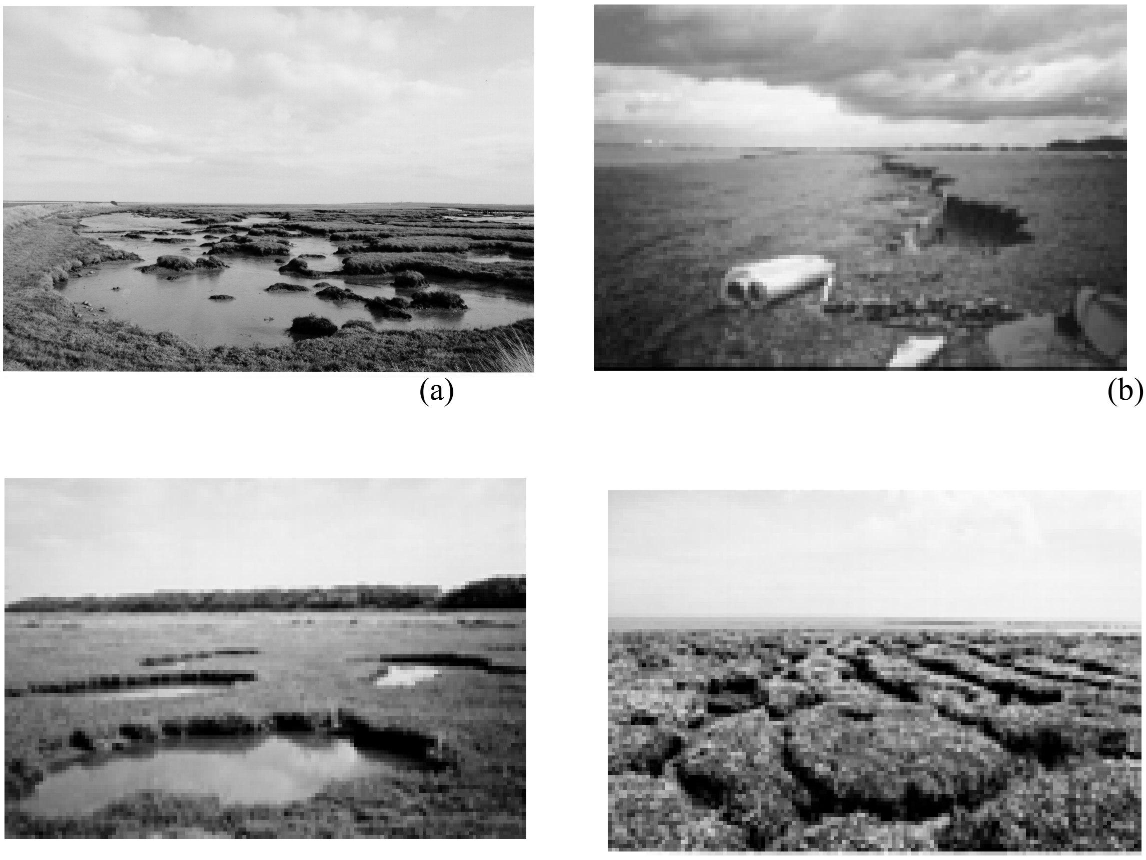
Creeks are a very important component of a saltmarsh, providing an extension from the intertidal mudflat and wider estuary into the interior of the marsh system. They are vital in exchanging water, sediment, nutrient (carbon, nitrogen etc.) between estuarine waters and the marsh interior. Creeks provide habitat for invertebrates, shelter for birds and conduits for fish and mobile invertebrates foraging within the creeks or on the marsh surface during highest tides. However, many of these functions are, as yet, very poorly quantified on European marshes.

Around Britain, and NW Europe more generally, saltmarsh morphologies are highly variable with creek networks and salt pans occurring in low or high densities (see Figure 3.1 for examples). Generally speaking, however, studies are beginning to show that the density of creek systems is related to a number of hydrological laws reflecting the volume tidal flow through creeks on and off the marsh surface (see Box 3.2; Figure 3.2; Steel & Pye 1997; Zeff 1999). In the UK the density of creeks, and hence the environmental functions that the marsh undertakes, is both regionally and locally variable, determined by factors such as tidal range and sediment properties. Similarly, naturally regenerated marshes (ie those created historically through breaches in sea walls during storm events) often possess creek systems that are similar, but not necessarily identical in morphology of marshes in that region (Box 3.3: Sedimentology).

Box 3.2 Understanding saltmarsh morphology

By contrast to the success of modelling vertical accretion response to sea-level movement, relatively little is known about the factors governing marsh and creek networks morphology (which are fundamental to the flux of water sediment and nutrients) and how they respond to changing flooding events. It is, however, recognised that marsh surface morphology can vary widely between regions, though is often intra-regionally consistent (Pye & French 1993). Salt-pans may be abundant or non-existent. Creeks may range in morphology from linear, through dendritic to complex, with varying degrees of density, frequency and regularity in branching. Similarities have been drawn between the planiform character of creek networks and fluvial networks and have thus been interpreted as reflecting a tidal drainage function (Frey & Basan 1985; French & Stoddart 1992). However, unlike fluvial creeks, flow of water through saltmarsh creeks is bi-directional. Maximum flood tide flow velocities often occur as the tide reaches the elevation of the marsh whilst maximum ebb velocities are often attained at a late stage on the ebbing tide (Bayliss-Smith *et al* 1978; French & Stoddard 1992).

In one study, based on a morphometric analysis of 13 marshes from around coastal England and Wales, a conceptual three stage model of creek development as marshes progress from a mudflat towards maturity has been suggested by Steel and Pye (1997). It is proposed that initially, as the young marsh develops, its primary creek system morphology is inherited from that of the ebb-flow drainage features of the mudflat. Gradually, with vertical accretion of the marsh surface, an increase in ebb-flow erosion induces headward extension of the creek network until a maximum density is reached at about mean high water level. Finally, as the marsh approaches maturity, reduced flow requirements lead to abandonment and infilling of smaller creeks. This study, and another in New Jersey on microtidal *Spartina* marshes (Zeff 1999) also describe the characteristics of creek systems planiform structure (order, density, bifurcation ratio and sinuosity) and cross sectional channel form (width: depth ratio, hydraulic geometry) to be related to marsh size.



(a) (b) (c) (d)

Figure 3-1 Example of the wide range of saltmarsh morphologies

The morphology of natural saltmarshes is diverse. Creek densities and the ratio of mudflat to vegetated saltmarsh may be very high, as in Essex and Kent and Poole Harbour (Example a: Tollesbury Marsh, Essex) or may be very low, as in the Severn Estuary (Example b; Littleton Warth, Gloucestershire). The occurrence of salt pans on saltmarshes is locally and regionally variable often associated with, but not exclusive to, older marshes with lower surface slope, poor surface drainage and low creek density. Salt pans appear to be medium to long-term features of some marshes and possess a rich ecological diversity (Example c: Warham Marsh upper terrace, Norfolk). Ridge and runnel features (Example d: Rumney Wharf, South Glamorgan) appear morphologically similar to creek networks but their genesis and function reflect wave energy attenuation rather than tidal flow scour.

Scientific studies around the world are beginning to identify the importance of a varied morphology in supporting biodiversity and other important functions. Many features of the marshes and mudflats are very difficult, if at all possible, to replicate. Currently the means to assess wetland functioning in the UK are not available. Because of this it is open to question whether restored mudflats and saltmarshes offer the same, lesser or enhanced environmental benefits as their natural counterparts.

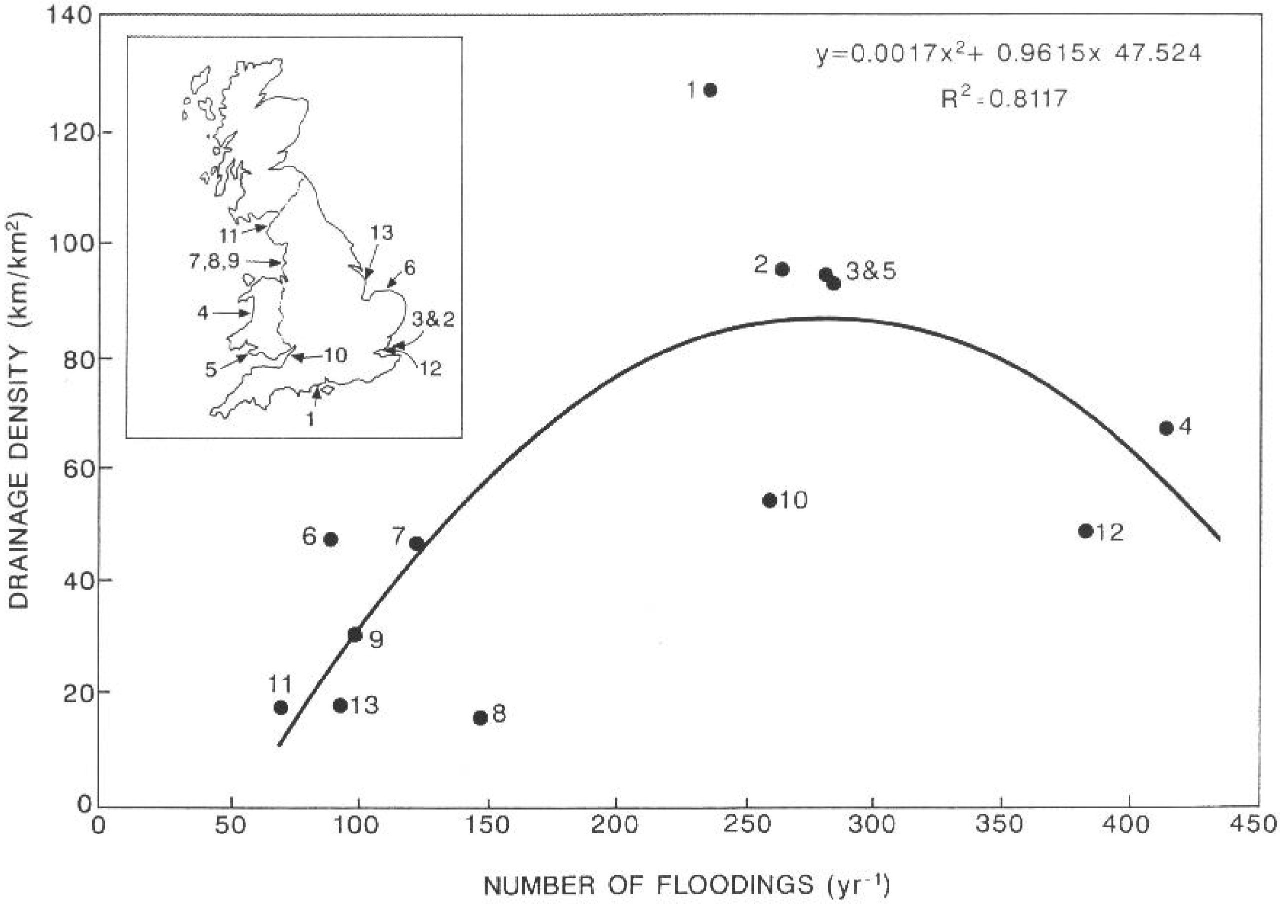


Figure 3-2 Changing creek density as saltmarsh accretes within the tidal frame (Steel & Pye 1997)

Largely related to the volume of water flowing across a marsh, tidal flow plays a major role in determining creek density on saltmarsh and mudflats. As a young saltmarsh grows above a mudflat creek density increases and then declines again reflecting the changing importance of creeks in transferring water and sediment to and from the marsh interior. Accompanying this general trend other factors such as sediment type, land surface slope and land management act to cause local and regional variations in creek density and form. Any intertidal restoration action should seek to tidal channels and creek with a natural range of forms and densities.

Box 3.3 Sedimentological influences on saltmarsh morphology

Beyond direct hydraulic controls there is evidence that sedimentology influences drainage character, causing marshes with extremely high or low creek densities (Crooks & Pye 2000).

The nature of saltmarsh sediments is determined by the availability of material and prevailing hydrodynamic conditions at the site of deposition. Estuarine marshes commonly consist of fine-grained sediments, derived from flocculated clays deposited within low energy settings. Even in many sheltered conditions, sand laminae may be abundant within the alluvium, which can greatly increase the subsurface flow of water. Marshes consisting largely of sandy material are not uncommon, and may develop from the colonisation of sand flats in open coastal areas enjoying waters with low suspended sediment concentrations (embayment margins of the Irish Sea for example). Hydraulic conductivity within sandy marshes is much greater than muddy marshes, and equally drainage through sandy levee will be greater than through more fine grained sediments distant from channels. In comparing marshes of similar size, creeks formed in sandy sediments tend to be wider and shallower than those in muddy sediment. Similarly, marshes forming in sandy systems are prone to periods of rapid accretion or erosion dependent, for instance, on fluctuations in intertidal channel proximity (Scot & Gray 1987).

It is also recognised that storm regenerated marshes in SE England are subject to poor drainage (Burd 1994; French 1996; French *et al* 1999). Poor drainage is enhanced by the lack of calcium carbonate within the formerly land-claimed saltmarsh sediments, which, if drained for agricultural purposes, are vulnerable to dispersion and formation of an overconsolidated horizon with very low hydraulic conductivity (Crooks 1999; Crooks & Pye 2000). On restoration of saltmarsh the impact of this reduced drainage is to reduce sediment resistance to erosion as well as supporting a vegetation cover which is different from that of nearby natural marshes (Crooks *et al* in press).

3.3 Methods available for restoring intertidal flat and saltmarshes

3.3.1 Coastal realignment

Coastal realignment is the landward relocation of the actively maintained line of flood defence and the restoration of intertidal habitat through reactivation of the coastal floodplain. In restoring intertidal habitat by realignment, there are many physical elements that must be carefully considered when choosing the design of a scheme. These include: tidal prism, estuarine morphology, tidal hydraulics, site history, surface elevation, surface gradient, sediment characteristics and accretion processes and wave climate (Burd 1995).

Depending on requirements and local situation, the original outer wall may be removed (bank retreat), breached (breach retreat) or retained with water levels controlled via a sluice or spillway. Bank retreat is a possible option if aiming to restore a new intertidal profile that is better able to respond to the physical processes defining wider estuarine morphology. However, removal of the seawall will expose the site to wave action and may result in prevention or limitation in the reestablishment of intertidal habitat, particularly if the site surface lies below the level of low tide. By maintaining extensive lengths of the outer seawall during breach retreat the re-establishing intertidal habitat will be protected largely from externally generated waves. The size of the breach required is dependant on the tidal prism of the site and can be calculated from Equation 1 (Box 3.1). Breach retreat may be used as an intermediary stage towards full bank retreat in providing a temporary shelter for the establishing intertidal habitat, while the outer wall degrades. Restoring intertidal habitat by permitting tidal flooding via a sluice has very limited impact (positive or negative) upon the wider estuary but because of constrained flow does not aid the creation of a natural marsh. From a hydrological perspective a fully functional wetland habitat should require no direct management and, as such, bank or breach retreat are preferred options.

Since 1995 there have been a handful but growing number of experimental coastal realignment sites around southern Britain. All of these have involved a breach retreat reflecting the preference to provide a sheltered environment from wave action.

3.3.2 Enhanced sedimentation

Where ongoing intertidal erosion is a flood defence problem, groynes and sedimentation fields (fenced areas) have been extensively used, notably in Essex and locally elsewhere in the UK, in an attempt to enhance mudflat accretion and hence reduce severity of wave erosion. Several different types of structures have been employed, including single rows of groynes, rows of groynes linked with shore-parallel sections, larger sedimentation fields with and without artificial drainage channels (called ‘grips’) and offshore breakwaters with or without marginal groynes (Pye & French 1993).

Two pairs of Schleswig-Holstein–type sedimentation fields were constructed at Marsh House and near Deal on the Dengie Peninsula. Each structure was approximately 400 m square and was enclosed by groynes constructed of double rows of wooden stakes (0.5 m apart) infilled with brushwood fences and secured by coated wire. On the inner side of the groynes, sediment excavated from a shallow ditch parallel to the groyne was heaped to stiffen the structure and to make the lower levels impermeable to water. Each square was divided into two 200 m wide sedimentation fields by constructing shore-normal (perpendicular, as opposed to parallel to the shore) earth groynes. At Marsh House, further wave protection was provided at the site in 1986 by the placement of 16 redundant Thames lighters (barges) along at the lower foreshore.

At Deal Hall, Pye and French (1993) reported significant accretion within the sedimentation fields. Similarly, at Marsh House, the elevation of the mudflat within the sedimentation field has been raised and is maintaining a constant equilibrium elevation, above that of the adjacent natural mudflat and below the level for saltmarsh development. This level is subject to seasonal variation and the site is prone to reworking during large storms. Elsewhere across Essex the success of such sedimentation fields has been mixed, with several local failures. After some 16 years since the first trials in Essex, the sedimentation field technique is believed only to be successful if the local sedimentary trend is accretionary. Where the trend is towards erosion, the fields have been ineffectual (Mark Dixon pers. comm.).

Sedimentation fields have also been used on the continental mainland, most notably in the sandy estuaries such as the Dollard in the Netherlands (Esselink 2000). Here, in areas that naturally favour long-term sediment accumulation, the rate at which saltmarshes have extended over intertidal flats has been enhanced.

3.3.3 Foreshore recharge and beneficial use of dredge material

The recharge of fine-grained cohesive (muddy) areas is more complex than for sand or shingle foreshores. While sands and gravels rapidly form well-drained deposits on placement, the cohesive nature of a muddy sediment means that it may take some time after placement before the sediments reach a ‘balance’ with the local tidal and wave conditions.

Reaching this balance involves complex de-watering and consolidation processes as well as biological actions from micro-organisms and invertebrates. Because of this, previous and most current mainstream recharge techniques with mud cannot be a rapid process but instead involve a collection of a slurry which is confined behind a protective bund until consolidation takes place. If placed upon the intertidal zone without confinement the slurry is lost through gravity flow down slope.

The method of placing the slurry into the intertidal storage site is most commonly via a hydraulic pipeline connected to a vessel. This system enables placement of material over possible distances of several kilometres. Numerous dredges of this type currently exist to meet the needs of channel dredging. Some have additional features, such as cutterheads capable of following the natural contours of the basin bottom without damage to natural or man-made seals, and capable of dredging forwards or backwards. A fluidising system is generally needed to create a slurry from subtidal muds. Unloading facilities are unnecessary since the dredged material is usually pumped out of a pipeline on to the containment area.

Because of consolidation and dewatering, a number of problems exist in terms of creating intertidal habitat with fine-grained sediment by hydraulic pipeline. To address these problems a number of experimental techniques have been applied in the UK. In particular, Harwich Harbour Authority are pioneering a hydraulic method of pump transportation utilising dredge material of high density which is then reworked in to a slurry without additional water and transported via the pipeline to the containment site. The advantages of this method is that the degree of dewatering required is reduced as are issues related to slurry flow. Early trial result are discussed in section 3.4. Alternative approaches, such as direct side release from barges, involve the mass placement of material on the lower foreshore from where the material is spread slowly over several days by wave and tidal currents. This placement technique has also been on trial on the Essex coast but requires further investigation to determine whether mudflats can be maintained in this way.

3.4 Summary of monitoring data from case studies

While the importance of sediment accretion in saltmarsh development and generally to address sea-level rise issues has been recognised for 20 years or more, the importance of other geomorphological features such as creek development and impacts of drainage are only just becoming widely acknowledged. It is because of this, and because in the UK many restoration actions have been primarily led by flood defence requirements rather than to replicate a lost habitat, basic monitoring has commonly involved only a vegetation survey accompanied by some form of sediment budget assessment. Most commonly, and most simply, the sediment budget has often been derived from repeated monitoring of a limited number of marker stations on the restoration site. On experimental restoration sites (notably coastal realignment sites at Tollesbury, Orplands and Abbots Hall and the dredge beneficial use sites at north Shotley and Trimley) monitoring has been more extensive. Aspects monitored have included: surface topographical change, physical and chemical soil characteristics, vegetation coverage and invertebrate recolonisation. In addition, monitoring of the intertidal zone seaward of the sites has been undertaken, both to assess the impacts of the breach on local hydrology and sedimentation and also to provide a reference ‘yardstick’ against which the progress of the restoration could be compared.

Given that these sites are experimental, and that a number of independent agencies have been responsible for undertaking the work, no coherent monitoring and assessment plan has yet been drawn up. The situation is little different in other parts of the world. In the United States, for example, regulatory requirements demand stringent monitoring programmes to assess functionality but review assessments have found that these actions focus on simpler aspects, such as vegetation coverage, while neglecting more complex issues of geomorphology and hydrology (Streever 2000, Zedler 2000).

3.4.1 Unmanaged retreat: geomorphological development following bank failure

There have been a number of historic sea defence failures around the southern regions of Britain (Table 1.2). These provide an indication of the way intertidal habitat will develop if sites are created through managed realignment without significant management. Most striking is the comparison of the marshes naturally restored in the south-east of England with those in the Severn Estuary.

Most of the natural marshes in south-east England are of the ‘estuarine fringe’ type (terminology of Pye & French 1993) lying seaward of flood embankments. Other natural types are locally represented, with ‘mid-estuarine island’ marshes having formed around the low-lying island of London Clay in the Blackwater and Medway, ‘back-barrier’ marshes behind large shingle spit complexes, as at Colne Point and Shell Ness on the Isle of Sheppey, and ‘open coast marshes’ on the Dengie Peninsula. Unlike sandy marshes in north-west England and the muddy marshes of the Severn Estuary, the marshes in south-east England do not display clear series of marsh terraces and it is uncommon to find more than two marsh levels juxtaposed. By further contrast with the Severn Estuary, most of the natural marshes are highly incised by creek networks and have a much higher proportion of bare mud relative to vegetated surface. Many individual creeks terminate in sub-circular basins or coil back on themselves, for example Old Hall Marsh, Tollesbury and Northey Island in the Blackwater Estuary (Figure 3.3). Natural open coast and outer estuarine marshes at Dengie and Foulness are an exception to this regional morphological generalisation. Dengie marsh, in particular, is characterised by a low density of linear creeks which often extend the full width of the marsh (Figure 3.3).

All recognised unmanaged realignments in the south-east UK have taken place within the estuaries, away from the open coast. The restored sites are highly incised either by a dense network of sinuous creeks or, by a rectilinear system reflecting the underlying former